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$PM_{2.5}$-related health and economic loss assessment for 338 Chinese cities

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ABSTRACT

China is in a critical stage of ambient air quality management after global attention on pollution in its cities. Industrial development and urbanization have led to alarming levels of air pollution with serious health hazards in densely populated cities. The quantification of cause-specific $PM_{2.5}$-related health impacts and corresponding economic loss estimation is crucial for control policies on ambient $PM_{2.5}$ levels. Based on ground-level direct measurements of $PM_{2.5}$ concentrations in 338 Chinese cities for the year 2016, this study estimates cause-specific mortality using integrated exposure-response (IER) model, non-linear power law (NLP) model and log-linear (LL) model followed by morbidity assessment using log-linear model. The willingness to pay (WTP) and cost of illness (COI) methods have been used for $PM_{2.5}$-attributed economic loss assessment. In 2016 in China, the annual $PM_{2.5}$ concentration ranged between 10 and 157 μg/m^3 and 78.79% of the total population was exposed to > 35 μg/m^3 $PM_{2.5}$ concentration. Subsequently, the national $PM_{2.5}$-attributable mortality was 0.964 (95% CI: 0.447, 1.355) million (LL: 1.258 million and NPL: 0.770 million), about 9.98% of total reported deaths in China. Additionally, the total respiratory disease and cardiovascular disease-specific hospital admission morbidity were 0.605 million and 0.364 million. Estimated chronic bronchitis, asthma and emergency hospital admission morbidity were 0.986, 1.0 and 0.117 million respectively. Simultaneously, the $PM_{2.5}$ exposure caused the economic loss of 101.39 billion US$, which is 0.91% of the national GDP in 2016. This study, for the first time, highlights the discrepancies associated with the three commonly used methodologies applied for cause-specific mortality assessment. Mortality and morbidity results of this study would provide a measurable assessment of 338 cities to the provincial and national policymakers of China for intensifying their efforts on air quality improvement.

1. Introduction

The rapid industrial revolution, urbanization and economic development in China have led to severe ambient air pollution in its cities. High levels of $PM_{2.5}$ (particulate matter with aerodynamic diameter < 2.5 μm) are now a serious environmental, social and economic burden in China which has attracted great public attention (Ferreri et al., 2017; Xue and Zhang, 2018; Wang et al., 2018; H. Chen et al., 2017; L. Chen et al., 2017; Cai et al., 2017; S. Li et al., 2016; L. Li et al., 2016). Several previous studies have concentrated on the negative relationship between the ambient $PM_{2.5}$ and acute, as well as chronic health effects on the different age group in China. These studies indicated that exposure to $PM_{2.5}$ can cause lung cancer (LC), ischemic heart disease (IHD), asthma, acute bronchitis, cardiovascular disease, mental and behavioral disorders, nervous system breakdown and other health complications (Guan et al., 2016; Lin et al., 2016; Pope et al., 2011; Korek et al., 2015; Shi et al., 2016; P. Yin et al., 2017; L. Zhao et al., 2017; Ho et al., 2018; T. Li et al., 2018; T. Liu et al., 2018).

The mortality attributable to $PM_{2.5}$ was about 7.1% of global mortality in the year 2010 (Evans et al., 2013). In the recent Global Burden of Disease (GBD) study, it was indicated that $PM_{2.5}$ was responsible for 4.09 (95% Confidence Interval (CI): 3.62, 4.58) million premature deaths in 2016, in which IHD, chronic obstructive pulmonary disease (COPD), lower respiratory infections (LRI) and LC contributes 38.51, 19.23, 15.97 and 6.83% respectively (Abajobir et al., 2017). According to this study, outdoor $PM_{2.5}$ exposure is the fourth leading risk factor in China (http://www.healthdata.org/china). In the absence of more stringent policies, the $PM_{2.5}$-related annual premature deaths are

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projected to be 1.374 million in 2030 (Lanzi et al., 2018). Past studies have shown large differences in the estimation of annual $PM_{2.5}$-related death, ranging between 0.91 and 3.03 million in China (Chen et al., 2013). The reason for the diverse result in the estimation of $PM_{2.5}$-related premature death is discussed in Section 3.3.

The $PM_{2.5}$-related health burden directly affects economic activity as measured in the national accounts and gross domestic product (GDP). Several studies have estimated the economic loss due to excess mortality and morbidity based on the PM_{10} concentrations in China (Wang and Mauzerall, 2006; Hou et al., 2010; Tang et al., 2014; Matus et al., 2012; X. Zhao et al., 2016; Gao et al., 2015; S. Li et al., 2016; L. Li et al., 2016). Some studies have focused on the economic loss assessment specifically using $PM_{2.5}$ data (Y. Xie et al., 2016; Lu et al., 2017). However, the focus has primarily been either on the city level or province level (Zhang et al., 2010; Huang et al., 2012; H. Yin et al., 2017; P. Yin et al., 2017; Jiang et al., 2015). Epidemiological studies have shown that $PM_{2.5}$ fraction of PM_{10} is primarily responsible for increased mortalities while the coarse fraction of PM_{10} contributes majorly to respiratory morbidity. Lanzi et al. (2018) estimated that in 2015, the economic loss from outdoor $PM_{2.5}$ pollution was 0.85 trillion US$ in 2015 and will reach 2.26 trillion US$ in 2030. According to the World Bank (World Bank, 2017), the welfare loss in China from ambient $PM_{2.5}$ grew from 0.13 trillion US$ to 1.59 trillion US$ from 1990 to 2013.

A major objective of this study is to provide a comprehensive health risk assessment followed by an economic impact assessment of $PM_{2.5}$ in China. Secondary objective includes a comparative analysis of cause-specific mortality outcomes, based on three prevalent health risk assessment methods, and estimation of potential health benefits for future scenarios under implementation of various standards in China for the year 2030. The study is based on newly available ground level $PM_{2.5}$ data in 338 cities of 31 Chinese provinces during 2016. The systematic approaches for $PM_{2.5}$-related health impacts and corresponding economic cost assessment are shown in Fig. S1 (Supplementary material). The methodological difference of mortality estimation is highlighted in this study for an appropriateness decision by practitioners.

2. Methodology

2.1. Ground-level monitoring of $PM_{2.5}$ and exposed population

The Department of the Environment continuously operates and maintains the national air quality monitoring network of China, an effort that began in 2012. Moreover, for the first time, $PM_{2.5}$ was included in this ambient air quality standard (GB 3095-2012). The hourly $PM_{2.5}$ concentrations from 1st January 2016 to 31st December 2016 in 338 cities in 31 Chinese provinces were obtained from the China National Environmental Monitoring Center. The mass concentration of $PM_{2.5}$ was measured by the method of the Filter Dynamic Measurement System (FDMS) in coordination with the method of Tapered Element Oscillating Microbalance (TEOM). The instruments which measured $PM_{2.5}$ concentration in each site were tested by using at least 3 samples based on HJ 618-2011, according to the regulations published by the Ministry of Environmental Protection of the People's Republic of China. Obtained data were refined to remove any inconsistency which might arise due to monitoring operation disruptions like routine maintenance activities, communication failures and power outages at monitoring sites. The data was transformed into standard z-scores and removed when the following conditions were met: (1) < 12 h of available data in a day; (2) an absolute z score was larger than 4 ($|z_t| > 4$); (3) the increment of the z score between the current time and a previous time larger than 6 ($z_t - z_{t-1} > 6$) (He et al., 2017; Feng et al., 2017). (4) have an increment twice the value of the city-wide average of the increment for all monitoring stations (city $(z_t - z_{t-1})$) (i.e., $(z_t - z_{t-1})$/city $(z_t - z_{t-1}) > 2$) (Song et al., 2017a, 2017b). After the screening, > 80% of the hourly data could be used for further calculation. The seasonal and yearly $PM_{2.5}$ concentrations were all calculated based on the filtered hourly data.

The age-specific population data have been sourced from the National Bureau of Statistics of China and the United Nation's projected age-specific population data for the year 2030 (NBSC, 2017; UN, 2017). Analyzed 338 cities are home to 1308 million people (UN, 2017), about 95% of Chinese population, in which 0.22, 3.65, 17.33 and 78.79% of the population is exposed to 10–15 μg/m^3, 16–25 μg/m^3, 26–35 μg/m^3 and > 35 μg/m^3 $PM_{2.5}$ concentration respectively.

2.2. Health impact assessment

In this study, we estimated the long-term health effects attributable to $PM_{2.5}$ exposure with the use of the epidemiological relative risk (*RR*), which can link the concentration of $PM_{2.5}$ to negative health effects in 31 Chinese provinces during the year 2016. For the $PM_{2.5}$-caused health impacts, the equation below was implemented which has been used in past studies (Evans et al., 2013; Lelieveld et al., 2015; M. Liu et al., 2017a, 2017b; Zheng et al., 2017):

$$HI = [(RR - 1)/RR] \times B \times EP \tag{1}$$

where *HI* is the health impacts of a specific-disease; *B* is the baseline disease-specific health impacts rate and *EP* is the population exposed to $PM_{2.5}$ with a specific age group. Here, the term [(RR − 1)/RR] is defined as the population attributable risk-potential reduction in the incidence of morbidity or mortality when an entire population would be exposed to pollution with reference concentration (Zhang et al., 2017).

In the past, three *RR* estimation functions were developed for ambient $PM_{2.5}$ exposure. For the $PM_{2.5}$-related mortality assessment study, the integrated exposure risk (IER) and log-linear (LL) exposure-response functions have mostly been used, although, the non-linear power law (NLP) function was also developed in the recent year. However, for $PM_{2.5}$-attributed morbidity studies, only log-linear (LL) function has been used in the past literature.

2.2.1. Integrated exposure risk (IER) function

Burnett et al. (2014) developed an IER function for relative risk estimation which was used to estimate global $PM_{2.5}$-related mortality under GDB study in 2010. The IER function incorporates data from cohort studies of ambient air pollution and tobacco smoke to describe the exposure-response relationship throughout the full distribution of ambient $PM_{2.5}$, including the extremely high levels that appear in China. The cause-specific *RR* was calculated through Eq. (2);

$$RR_{IER}(C_a) = \begin{Bmatrix} 1 + \alpha(1 - \exp^{-\gamma(C_a - C_0)^{\delta}}), & if \;\; C_a > C_0 \\ 1, & else \end{Bmatrix} \tag{2}$$

where C_a is the annual average ambient $PM_{2.5}$ concentration; C_0 is the threshold concentration below which no additional health impacts are calculated; and α, γ and δ are parameters used to describe the different shapes of the relative risk curve among various diseases (Burnett et al., 2014; Jiang et al., 2015).

This method remains widely applied for $PM_{2.5}$-related mortality estimation after 2010 (Table A1). Following the GBD approach, four leading causes of deaths attributable to $PM_{2.5}$ [cerebrovascular disease (stroke), ischemic heart disease (IHD), chronic obstructive pulmonary disease (COPD) and lung cancer (LC)] among adults (≥25 years), was calculated.

2.2.2. Log-linear (LL) function

The relative risk was estimated by LL-function (Eq. (3)), were drawn from the studies conducted by Pascal et al. (2013), Lelieveld et al. (2013) and Hubbell et al. (2009). This function has mostly been used for the estimation of health impacts in high $PM_{2.5}$ polluted regions (Lelieveld et al. (2013). The LL function remains the most widely applied method at present for studies in China (H. Yin et al., 2017; Lin et al., 2017; L. Chen et al., 2017; Zheng et al., 2015).

In this method, the cause-specific RR was calculated through Eq. (3);

$$RR_{LL} = \exp[ER \times (C_a - C_0)] \tag{3}$$

where *ER* is the exposure-response coefficients obtained from the recent epidemiological studies in China, which means the incidence change of certain health impact per μg/m^3 of $PM_{2.5}$ increment.

Recently, H. Yin et al. (2017) and P. Yin et al. (2017) carried out a most comprehensive cohort study, focusing on the long-term $PM_{2.5}$ exposure related cause-specific mortality among Chinese men, used in the present study.

The *ER* coefficient can be derived by the following expression: $ER = \ln(RR)/\Delta C$ (Lelieveld et al., 2013) where ΔC is the change in pollution concentration.

The $PM_{2.5}$-related morbidity health endpoints have been selected based on the availability of *ER* coefficient, baseline incidence rate (B), and the economic cost of each health outcomes. After a considerable literature review, the five health endpoints covered are morbidities related to $PM_{2.5}$-related exposure, hospital admission due to respiratory disease (RHA), hospital admission due to cardiovascular disease (CHA), chronic bronchitis (CB), asthma attack (AA) and emergency room visits (ERV) for respiratory disease. Health impact-specific ER coefficients are listed in Table 1, with priority for studies performed in China to maximize regional accuracy.

2.2.3. Non-linear power law (NLP) function

Chowdhury and Dey (2016) developed NLP function for four diseases (stroke, IHD, COPD and LC), based on the past cohort studies across the globe-spanning exposure from ambient, household air pollution, active and second-hand smoking. The cause-specific RR at any given $PM_{2.5}$ exposure was calculated through Eq. (4):

$$RR_{NPL} = 1 + \alpha \times (C_a - C_0)^{\beta} \tag{4}$$

where α and β are the two constant, having the different value for each cause-specific mortality.

The $PM_{2.5}$-exposure related premature mortality among infants (< 5 years) (acute lower respiratory infection, ALRI) was not calculated in the present study, as their contribution was very low, around 0.32–0.82% of total $PM_{2.5}$-related estimated deaths (Wang et al., 2018; Maji et al., 2018). The modeling was performed with the R statistical and graphics software, version 3.4.3 (R Core Team, 2017).

2.3. Baseline incidence rate and threshold concentration

The national mortality-specific baseline incidence rates were derived from GBD study 2016 (http://www.healthdata.org/results/data-visualizations). The baseline mortality value for stroke, IHD, COPD and LC have been fixed for all cities in China at 130.95, 126.01, 64.13 and 43.20 per 10^5 population respectively. In addition, the baseline incidences rate for the five morbidity health endpoints attributed to $PM_{2.5}$ pollution is listed in Table 1.

Table 1
Relative risk factor and baseline incidence rates for the $PM_{2.5}$-related morbidity health endpoints.

Health impacts	RR (95% CI) (10 μg/m^3)	BIR per 10^5 population	Reference
Respiratory disease hospital admission	1.022 (1.013–1.032)	550.9 (NBSC, 2016)	Li et al. (2013)
Cardiovascular disease hospital admission	1.013 (1.007–1.019)	546.0 (NBSC, 2016)	Qiu et al. (2013)
Chronic bronchitis	1.029 (1.014–1.044)	694.0 (NBSC, 2016)	Li et al. (2013)
Asthma attack	1.021 (1.015–1.028)	940.0 (NBSC, 2016)	Ko et al. (2007)
Emergency room visits for respiratory disease	1.01 (1.005–1.016)	204.5 (NBSC, 2016)	Zhang et al. (2017)

In the past studies, the theoretical $PM_{2.5}$ threshold concentration varied between 5.8 and 8.8 μg/m^3 (Héroux et al., 2015; Burnett et al., 2014; Apte et al., 2015), but in the recent GDB study, the concentration range below which no adverse health outcomes are reduced to 2.4–5.9 μg/m^3, therefore, current study used 5.9 μg/m^3 as the threshold value (Gakidou et al., 2017). The morbidity assessment study was performed using WHO AQG level of 10 μg/m^3 as reference concentration (WHO, 2006) and as used in the past studies conducted for China (Lin et al., 2017; H. Yin et al., 2017; P. Yin et al., 2017; Zhang et al., 2017).

2.4. Economic valuation estimates

In the absence of a market for human lives, the monetization of mortality relies on non-market valuation methods (OECD, 2012). A standard method for estimating the monetary cost of a positive welfare effect, e.g., a reduction in mortality risk, is to create a hypothetical market for the mortality risk considered and derive based on the value of statistical life (VSL) (Braathen et al., 2010; Xie, 2011). The VSL is calculated in survey studies assessing individuals 'willingness to pay' (∂WTP) for a small reduction of mortality risk ∂R. Thus, for a small value of ∂R, the $VSL = \partial WTP/\partial R$ (Persson et al., 2001; Huang et al., 2012). Whereas, the morbidity related economic cost is estimated mainly using WTP as well as the cost of illness (COI) method. The COI method calculates the cost of a disease in terms of medical treatment cost, hospitalization costs, and productivity loss (Hoffmann et al., 2012).

WTP of health endpoints assumed to increase as individual income increases (Hammitt and Robinson, 2011). In the past, most of the studies estimated health cost based on the average per capita gross domestic product (GDP) in China whereas the per capita GDP for 31 Chinese provinces are highly different. Health costs for Beijing or a national average for China has primarily been estimated in previous studies. As the city and province-specific health costs are not available for China, we derived the different health costs (WTP and COI) for 31 Chinese provinces in the year 2016, using following benefit transfer method (Giannadaki et al., 2018; Lu et al., 2017; Matus et al., 2012): -

$$HC_{i,2016} = HC_{C/B,k} \times \left(\frac{G_{i,k}}{G_{C/B,k}}\right)^{\beta} \times (1 + \%\Delta P_i + \%\Delta G_i)^{\beta} \tag{5}$$

where $HC_{i,\,2016}$ is the adjusted health cost for province *i* in 2016; $HC_{C/B,\,k}$ is the base value of health cost for China or Beijing in the year *k*; $G_{i,\,k}$ is the GDP per capita in purchasing power parity (PPP) terms in province *i* in the year *k*; $G_{C/B,\,k}$ is the GDP per capita in PPP terms in China or Beijing in the year *k*; β is the income elasticity of health cost equals to 0.8 as recommended by OECD (OECD, 2016). $\%\Delta P_i$ and $\%\Delta G_i$ are the percentage increase/decrease in consumer price index (CPI) and GDP per capital in the province *i* during the year *k* to 2016.

The total economic health burden in a province *i* can be evaluated by multiplying the number of disease-specific health impacts (HI_i) attributable to $PM_{2.5}$ in province *i* with the corresponding $HC_{i,\,2016}$ value using the Eq. (6).

$$Economic\ \ Burden = HI_i \times HC_{i,2016} \tag{6}$$

The details of the economic cost of $PM_{2.5}$-caused specific health damage together with the sources from where the estimates are taken in 31 provinces for total economic loss analysis are shown in Table 2 (see details Supplementary material Table S1).

Table 2
Economic costs (US$) per case of different health endpoints in Beijing or China.

Health impacts	Economic cost	Approach	Study location	Reference
Mortality	132,000	WTP	Beijing	Guo et al., 2010; Matus et al., 2012; Maji et al., 2017; H. Yin et al., 2017; P. Yin et al., 2017
Respiratory disease	792.90	COI	China	Gao et al., 2015; Zeng et al., 2015; Zhou et al., 2014; Zhang et al., 2017; Maji et al., 2017
Cardiovascular disease	1205.93	COI	China	Gao et al., 2015; Zeng et al., 2015; Zhou et al., 2014; Zhang et al., 2017; Maji et al., 2017
Chronic bronchitis	7000	WTP	Beijing	Guo et al., 2010; Matus et al., 2012; Maji et al., 2017; H. Yin et al., 2017; P. Yin et al., 2017
Asthma attack	7	WTP	Beijing	Guo et al., 2010; Matus et al., 2012; Maji et al., 2017; H. Yin et al., 2017; P. Yin et al., 2017
Emergency	35.32	COI	China	Gao et al., 2015; Zeng et al., 2015; Zhou et al., 2014; Zhang et al., 2017; Maji et al., 2017

3. Results and discussion

3.1. Distribution of $PM_{2.5}$

The spatial distributions of annual mean $PM_{2.5}$ concentrations in 338 Chinese cities are illustrated in Fig. 1a. The annual average $PM_{2.5}$ ranges from 10 μg/m^3 (Altay in Xinjiang province) to 157 μg/m^3 (Kashgar in Xinjiang province) with an average of 46 ± 18 μg/m^3. As many as 232 cities cannot meet the Chinese National Ambient Air Quality Standard (CNAAQS) (35 μg/m^3) and no city was able to meet the WHO guidelines (10 μg/m^3) in 2016. $PM_{2.5}$ concentration is higher in the densely populated Northern regions than those observed in lower population region in the South. $PM_{2.5}$ also tend to be higher in the inland region compared to the coastal regions. The highest $PM_{2.5}$ concentration is observed in the Beijing-Tianjin-Hebei (BTH) region including two megacities of Beijing (72 μg/m^3) and Tianjin (69 μg/m^3); Hebei (69 μg/m^3) province, and Henan (72 μg/m^3) province.

In winters, $PM_{2.5}$ concentrations in China remained very high compared to summers, highlighting a variability trend with seasons (Fig. 1). Reduced pollutant dispersion associated with adverse meteorology as well as emissions associated with heating through fossil fuel and biomass may justify high $PM_{2.5}$ levels in winter. In addition, the role of the summer in Asian monsoon in bringing clean air for pollutant dilution along with humid aerosols and efficient atmospheric convection remains crucial for lower summer $PM_{2.5}$ levels specifically in Eastern China. (Sun et al., 2014). As a result, the $PM_{2.5}$ minima are observed in summer.

3.2. $PM_{2.5}$-attributed premature mortality

The long-term exposure to $PM_{2.5}$ is associated with increased mortality in adults (≥25 years) from stroke, IHD, COPD and LC were estimated based on the health impact function (Eq. (1)). As the IER, LL and NPL relative risk estimation function generate different relative risk value, therefore different results have been obtained in this health risk assessment study. For three relative risk estimation functions, the total $PM_{2.5}$-related premature mortality (i.e. the sum of four type of mortality) in one million Chinese population is shown in Fig. S2. A summary of methodology specific and cause-specific mortality attributable to $PM_{2.5}$ is shown in Fig. 2. Based on IER function, the total premature mortality attributed to $PM_{2.5}$ in China was 0.964 (95% CI: 0.447, 1.355) million, whereas the premature mortality was 1.258 (95% CI: 1.053, 1.420) and 0.770 (95% CI: 0.235, 1.267) million in LL and NPL function. In the total mortality, stroke contributes 48.76% (IER) [LL: 41.02%, NPL: 24.27%]; IHD contributes 31.09% (IER) [LL: 28.73%, NPL: 20.64%]; COPD contributes 10.89% (IER) [LL: 18.07%, NPL: 27.69%] and LC contributes 9.26% (IER) [LL: 12.17%, NPL: 27.40%]. High uncertainty was observed for NPL function based estimated premature mortality. The total deaths were about 30.46% higher in LL than IER whereas 25.30% lower in NPL than IER.

The total annual premature mortality attributed to $PM_{2.5}$ (IER) was about 9.98% of total deaths in China during year 2016. The $PM_{2.5}$-stroke caused 470 (95% CI: 161, 614) thousand and $PM_{2.5}$-IHD caused 300 (95% CI: 208, 458) thousand premature deaths which were 26.3% and 17.4% of cause-specific deaths in China. COPD and LC contributed 105 (95% CI: 50, 155) thousand and 89 (95% CI: 28, 128) thousand premature deaths and those are about 12% and 15.1% of cause-specific deaths in China during 2016 (Fig. 3).

Fig. 4 illustrates the total $PM_{2.5}$-related premature mortality in cities of 31 provinces of China. The provinces which have the highest premature mortality were Anhui (ANH) [52 (95% CI: 24, 72) thousand], Guangdong (GUD) [58 (95% CI: 26, 85) thousand], Hebei (HEB) [64 (95% CI: 31, 87) thousand], Henan (HEN) [82 (95%: 39, 110) thousand], Hunan (HUN) [50 (95% CI: 23, 70) thousand], Jiangsu (JIS) [61 (95% CI: 28, 86) thousand)], Shandong (SHD) [77 (95% CI: 36, 105) thousand] and Sichuan (SIC) [61 (95% CI: 28, 86) thousand] and they contributed 5.3, 6.0, 6.6, 8.5, 5.2, 6.3, 7.9 and 6.3% of total mortality respectively (Fig. 4).

The western, northwestern and southern parts of China had lower $PM_{2.5}$ concentrations and the cities with lower population density had lower premature mortality. Cities with high levels of pollution and population density, such in Beijing and in North China had the largest number of $PM_{2.5}$-related premature death, while the highest per capita attributable deaths are all located in heavy-pollution industrial regions such as in Liaoning Province, in Hebei province and in Henan Province. In BTH region, cities with high $PM_{2.5}$-related deaths were Beijing (18.58 thousand), Tianjin (9.98 thousand), Shijiazhuang (9.89 thousand) and Baoding (10.70 thousand). The premature deaths of the megacities Shanghai and Chongqing were about 17.74 and 10.74 thousand. The city, Guangzhou (8.18 thousand), Harbin (8.12 thousand), Zhengzhou (8.25 thousand), Nanyang (8.30 thousand), Wuhan (8.18 thousand), Suzhou (8.83 thousand), Linyi (8.64 thousand) and Chengdu (11.79 thousand) had high $PM_{2.5}$-related premature deaths in 2016. The city-specific premature death per 100,000-person-years and city-specific death per years attributed to $PM_{2.5}$ has been shown in Fig. 5 and Fig. S3 (see Supplementary material).

3.3. Uncertainty and variations in previous studies

From the year 2010, various studies on mortality attributed to $PM_{2.5}$ in China have been compiled in Table A1 (see Appendix A) for a comparative analysis and there are very large differences in the results. The difference in $PM_{2.5}$-related deaths is mainly due to the different methods for ground-level $PM_{2.5}$ estimation and the modeled resolution in a study. The past studies mainly relied on chemical transport models, satellite data or the kriging model performed on ArcGIS (van Donkelaar et al., 2010; Lim et al., 2012; Evans et al., 2013; Brauer et al., 2015; Fang et al., 2013; Lelieveld et al., 2015; Apte et al., 2015; Liu et al., 2016). These models included huge rural areas where measurements were not available and often lacking the validation with ground measurements (Chen et al., 2013).

The reported $PM_{2.5}$-related premature deaths in 2010 were 1.08 million in the GDB 2010 study (Lim et al., 2012); 1.27 million by Apte et al. (2015); 1.36 million by Lelieveld et al. (2015); 1.26 million by R. Xie et al. (2016) and Y. Xie et al. (2016) (excluding ALRI) and 1.02 million by H. Zhao et al. (2017) and L. Zhao et al. (2017) (excluding ALRI). In these studies, Lim et al. (2012) estimates of $PM_{2.5}$ using TM5 model and satellite data with 0.1° × 0.1° grid cell resolution and linked to available measures of $PM_{2.5}$ from ground-based monitoring, whereas Lelieveld et al. (2015) analyzed $PM_{2.5}$ using ECHAM5/MESSy

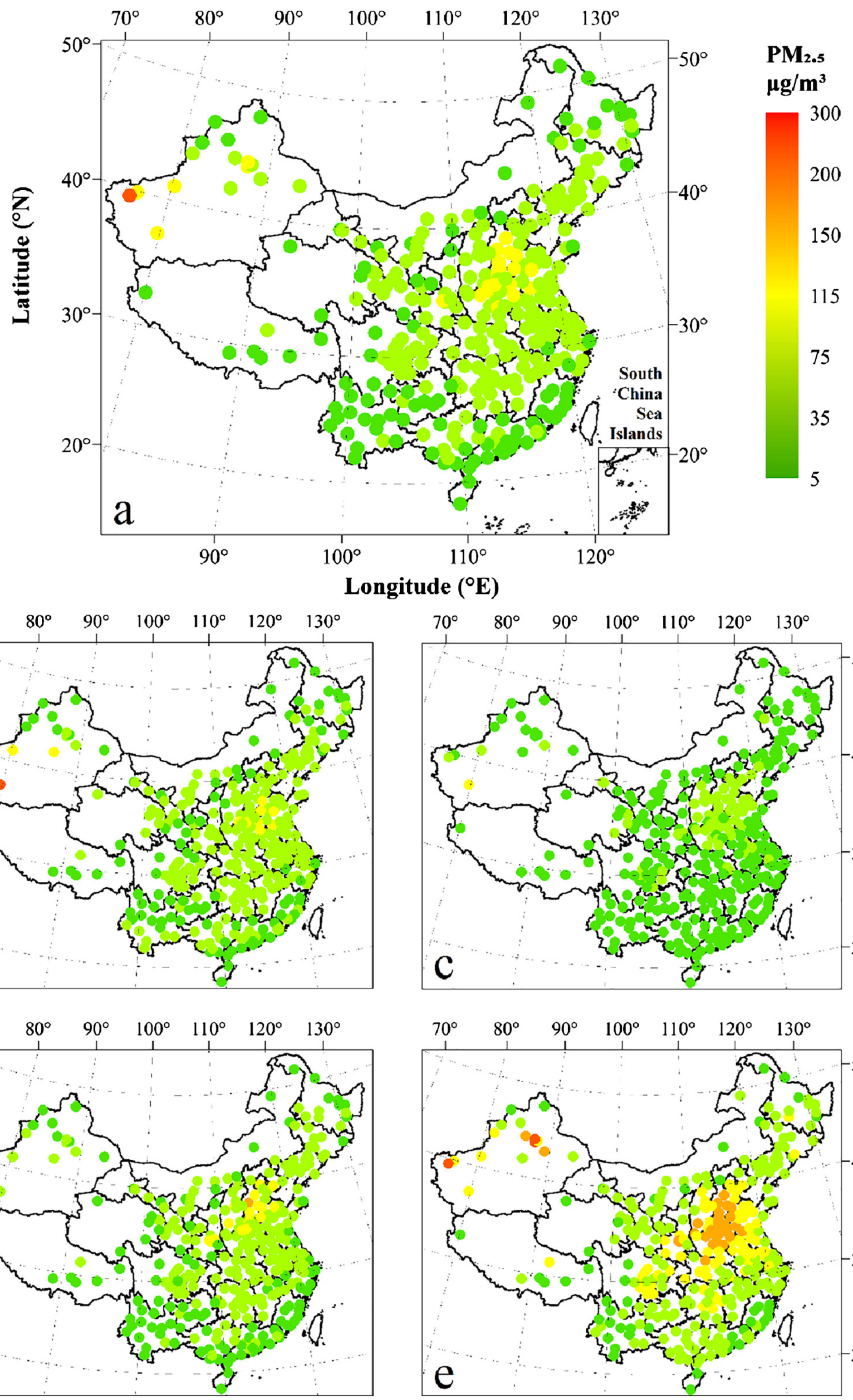

Fig. 1. (a) The averaged $PM_{2.5}$ concentrations (μg/m^3) in the 338 cities in China during the year of 2016 and during the (b) spring, (c) summer, (d) autumn and (e) winter.

atmospheric chemistry (EMAC) model using the grid size approximately 1.1° × 1.1°. Both estimations were based on the older IER model (Lim et al., 2012). On the other side Apte et al. (2015) used same age-specific mortality data, however, they used updated IER function (IHME, 2016) and satellite data with the combination of GEOS-Chem chemical transport model for ground-level $PM_{2.5}$ estimation (0.1° × 0.1° grid cell resolution). R. Xie et al. (2016) and Y. Xie et al. (2016) and H. Zhao et al. (2017) and L. Zhao et al. (2017) used the same IER function and baseline cause-specific death rate (IHME, 2016) although the total $PM_{2.5}$-related deaths were different. R. Xie et al. (2016) and Y. Xie et al. (2016) applied satellite-derived ground-level $PM_{2.5}$ concentration at a spatial resolution of 0.1° × 0.1° developed by van Donkelaar et al. (2015), and H. Zhao et al. (2017) and L. Zhao et al. (2017) used satellite instruments and conversion factors between aerosol optical depth and

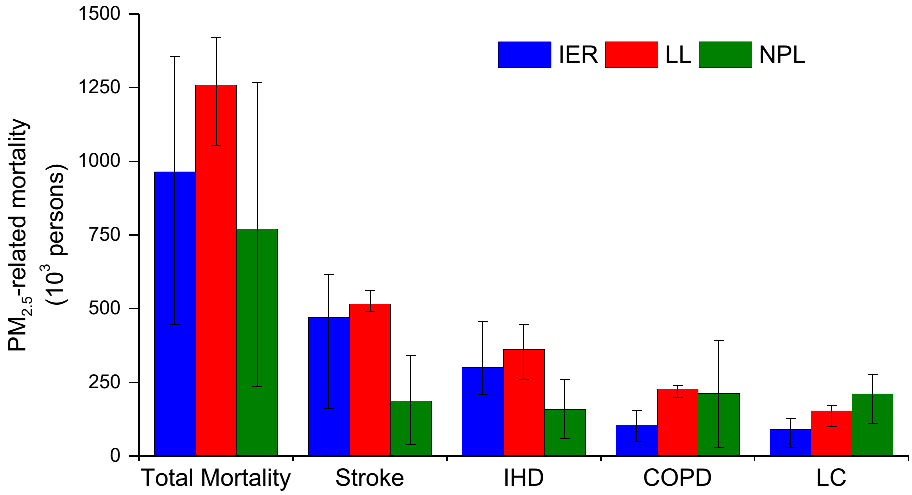

Fig. 2. Methodology-specific and cause-specific mortality attribute to $PM_{2.5}$ in China.

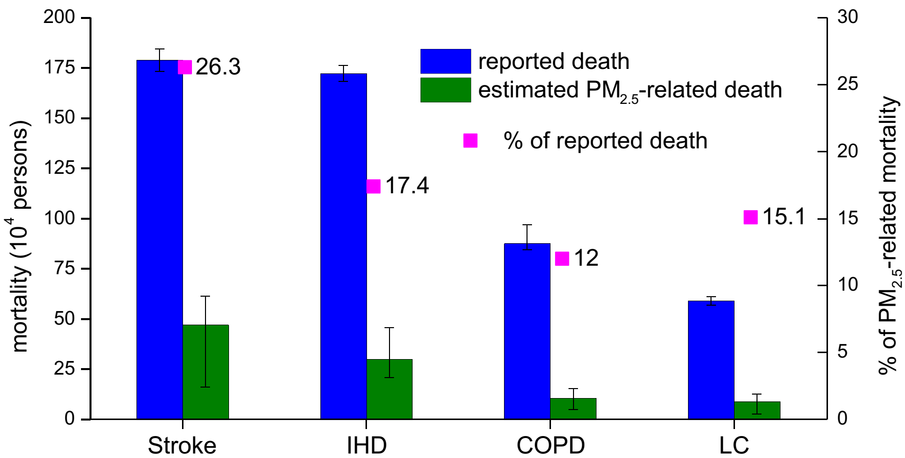

Fig. 3. Reported and $PM_{2.5}$-related disease-specific mortality in 338 cities in China.

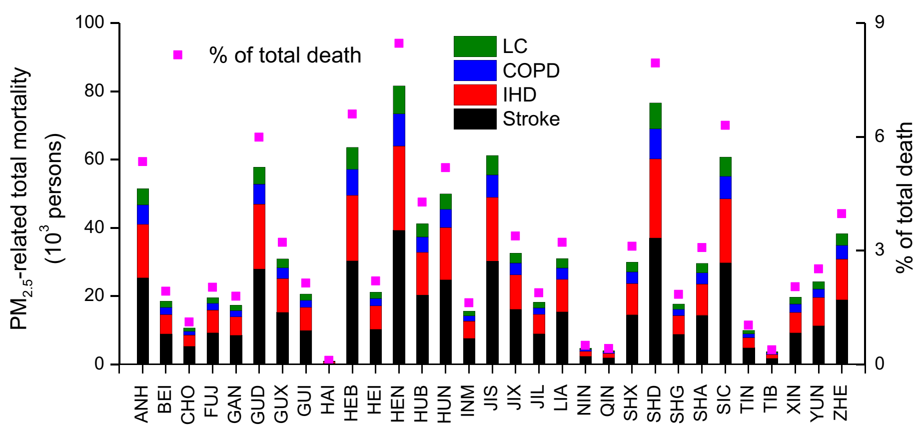

Fig. 4. $PM_{2.5}$-related disease-specific mortality in 31 provinces in China.

$PM_{2.5}$, simulated by the GEOS-Chem chemical transport model at a resolution of 0.5° × 0.667°.

The use of different population density, age distribution and baseline mortality rate are likely to be responsible for the difference in the results. For example, the $PM_{2.5}$-related premature deaths in 2015 were reported 1.52 million by Song et al. (2017a); 1.85 million by Zhang et al. (2017) (excluding ALRI) and 1.13 million by Feng et al. (2017) (excluding ALRI). These studies used the same IER function and ground monitoring data set for $PM_{2.5}$ although the $PM_{2.5}$-related total deaths were different because of the diverse use of the population density and cause-specific incidence death rate.

The variation in relative risk function is also a major source of uncertainty in $PM_{2.5}$-related mortality estimation (Burnett et al., 2014; Fang et al., 2016; H. Chen et al., 2017; L. Chen et al., 2017). The premature mortality calculated based on IER function had slightly higher value than NLP function and the use of LL function leads to a much

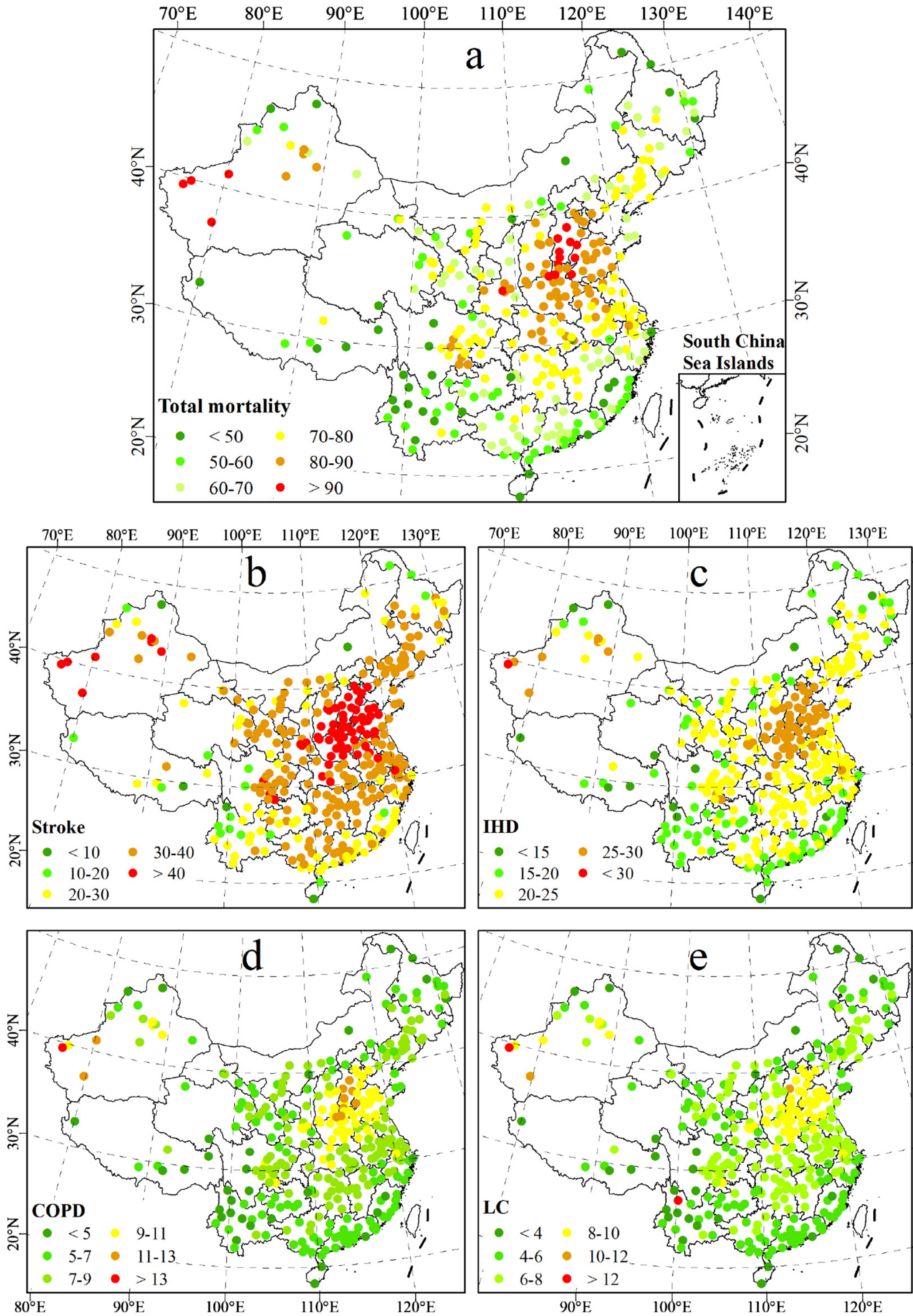

Fig. 5. Mortality per 100,000-person attribute to $PM_{2.5}$ in China 338 cities in 2016 (a) Total mortality, (b) Stroke, (c) Ischemic Heart Disease (IHD), (d) Chronic Obstructive Pulmonary Disease (COPD) and (e) Lung Cancer (LC).

higher value of $PM_{2.5}$-related deaths. For example, $PM_{2.5}$-related premature deaths were estimated to be 1.37 million by Liu et al. (2016) using IER function for the year 2013 whereas Fang et al. (2016) reported 3.03 million deaths using LL function, in 74 Chinese cities in 2013. Applying LL function, H. Chen et al. (2017) and L. Chen et al. (2017) estimated that the combination of all air pollution was responsible for 6.5% to 25.7% of total deaths in China.

Variations in the estimations of $PM_{2.5}$-related mortality due to differences of relative risk function were significant, until the development of IER function by Burnett et al. (2014). After its use in GDB study, IER function remains a widely applied method for $PM_{2.5}$-related mortality estimation. However, the IER incorporated $PM_{2.5}$-related mortality risk from cohort studies in Western Europe and North America, where $PM_{2.5}$ exposures were usually lower than China. Direct evidence of the magnitude of mortality risk over the complete global exposure range is therefore lacking in existing knowledge base (H. Yin

Table 3
Excess $PM_{2.5}$-related mortality of different health outcomes in different scenarios in China in 2030 (10^3 persons).

	2016 scenario	IT-1 (35 μg/m^3)	IT-2 (25 μg/m^3)	IT-3 (15 μg/m^3)	AQG (10 μg/m^3)
Stroke	470.2	429.8	324.8	141.5	50.9
IHD	299.8	287.5	253.3	184.8	126.1
COPD	150	90.8	69	42.3	18.9
LC	89.3	76	59.9	36	16.8
Total	964.3	884.1	706.9	404.6	212.6
Avoidable excess premature mortality	–	80.2	257.4	559.7	751.6

et al., 2017; P. Yin et al., 2017). The personal exposure monitoring in a city indicated high space-time variation in $PM_{2.5}$ concentrations, which is usually not captured by the fixed air quality monitoring stations (Menon and Nagendra, 2018). In addition, the short-term exposure to higher concentrations during haze can have high health impacts, which needs to be considered for the health risk-based air quality management (Gao et al., 2015).

3.4. Potentially avoidable $PM_{2.5}$-related mortality in different scenarios in 2030

The government of China wants to achieve the national standard of $PM_{2.5}$ by 2030. Table 3 shows the potential health benefits of controlling $PM_{2.5}$ emissions under different $PM_{2.5}$ concentration scenarios for the year 2030. The number of potentially avoidable mortalities in these scenarios increased with the declining $PM_{2.5}$ concentration. The least health benefits were observed by restricting the targets to IT-1, only 80.2 thousand deaths can be avoided (8.3% compared with 2016), although that is the primarily targeted by the Government of China. The total potential mortality benefits in scenarios where the $PM_{2.5}$ concentrations meet the IT-2, IT-3 and AQG are 257.4, 559.7 and 751.6 thousand, which represent 26.7, 58.0 and 77.9% of the base year. As the total population and aged population (≥25 years) will increase by 2030, the mortality benefits will not increase proportionally to the targeted reduction in $PM_{2.5}$.

HEI (2016) predicted that 0.28 million premature deaths will be avoided in 2030 by continued actions to control air pollution. However, even with these reduced future $PM_{2.5}$ levels, as the Chinese population continues to grow and age, the $PM_{2.5}$-related death will increase to 1.3 million in 2030.

3.5. Morbidity due to $PM_{2.5}$

Five morbidity indicators associated with acute exposure of $PM_{2.5}$ were estimated for all Chinese cities under investigation for the year 2016. The total respiratory disease and cardiovascular disease-specific hospital admissions were 0.61 (95% CI: 0.37, 0.86) million and 0.36 (95% CI: 0.20, 0.52) million. The $PM_{2.5}$-related hospital admissions due to chronic bronchitis or to asthma attacks as well as emergency hospital visits were 0.99 (95% CI: 0.50, 1.44) million, 1.00 (95% CI: 0.70, 1.28) million and 0.12 (95% CI: 0.06, 0.18) million respectively. The provinces with the highest morbidity cases were Anhui, Hebei, Henan, Jiangsu, Shandong, Sichuan and they contributed about 5.25, 8.69, 10.52, 6.18, 9.55 and 6.09% of total morbidity cases respectively. The magnitude of $PM_{2.5}$-caused specific health damage in 31 provinces has been shown in Table S2 (see Supplementary material).

3.6. Economic loss

Across 338 Chinese cities, we apply the WTP and COI method to calculate the economic loss attributable to particulate matter pollution, as shown in Table S1. Compared to mortality and morbidity, the economic health burden is influenced mainly by the GDP per capita and consumer price in each year in a province (Zhang et al., 2017). The economic cost for each province and for each health endpoints is estimated based on Eq. (5) (OECD, 2014). Applying the mortality and morbidity estimation (Table S2) and economic costs per case of different health endpoints, we estimated the total economic cost of health effects due to $PM_{2.5}$ pollution by Eq. (6). The sum of the economic loss in all the cities due to the mortality and morbidity in 2016 was 101.39 [range(r): 47.33, 142.79] billion US$, which corresponds to 77.52 US$ per capita per year, which was 0.91% (r: 0.42, 1.27) of the total GDP in China (World Bank, 2017). In the total economic loss, premature deaths accounted for 94.76 (r: 43.88, 133.18) billion US$, approximately 93.46% of the total loss. The high $PM_{2.5}$-economic loss was observed for the province Guangdong, Hebei, Henan, Jiangsu, Shandong, Sichuan and Zhejiang. The losses were about 7.19, 5.85, 7.13, 8.16, 8.37, 5.25 and 5.86% of the total economic loss in China for these provinces respectively. The magnitude of $PM_{2.5}$-economic loss in 31 provinces has been shown in Table S3 (see Supplementary material). Economic loss due to $PM_{2.5}$-related health damage mainly depends on the mortality health cost and mortality level in a province. For example, $PM_{2.5}$-related mortality in Beijing (18.59 thousand) was lower than Guangxi (31 thousand), however the economic loss in Beijing [4.75 (r: 2.24, 6.65) billion US$] was higher than Guangxi [2.63 (r: 1.19, 3.78) billion US$], as the mortality cost in Beijing (0.187 million US$) was higher than Guangxi (0.081 million US$).

Zhang et al. (2017) reported that at the national level, China's $PM_{2.5}$-associated welfare loss was 248 billion US$ in 2015, which was 3.1% of China's GDP. Giannadaki et al. (2018) estimated that the economic cost for the 1.33 million $PM_{2.5}$-attributed deaths in China in 2010 was about 1.3 trillion US$ which is 10% of the GDP per capita. Such an estimate of economic loss remains unjustifiably very high because the VSL was taken as about 0.98 million US$.

The cities with high $PM_{2.5}$-related-mortality are mainly located in the heavily polluted and densely populated regions like BTH region and North China. This highlights the urgency for China to control air pollution in these regions. Still, a considerable number of premature deaths were seen in other regions with a lower average concentration of the $PM_{2.5}$. For example, some provinces with relatively low $PM_{2.5}$ concentration but high population density in Shanghai and Guangdong had large numbers of premature deaths. This information is important for making priority emission-abatement policies for different regions. For the highly populated regions, policies and air quality standards may need to become stricter to avoid more $PM_{2.5}$-related premature mortality. In recent years, the Chinese government has taken measures to control the pollution through the "Air Pollution Prevention and Control Action Plan" was initiated in 2013 (State Council, 2013) and by this plan, China wanted to achieve the city-specific targeted $PM_{2.5}$ concentration in 2017. Because of the intensive planning, $PM_{2.5}$ concentration has seen continuous improvement in air quality in most of the cities (Song et al., 2017b; Cai et al., 2017; Liu et al., 2018). However, it is still common for $PM_{2.5}$ concentrations to exceed relevant standards by large margins (Clean Air Asia, 2016). The $PM_{2.5}$-related premature mortality will likely decline if the emission reduction targets are reached. HEI (2016) predicted that with continued actions to control air pollution, levels will decline substantially by the year 2030, and 0.28 million premature deaths will be avoided. Although, combined with the population grows and people live longer, the $PM_{2.5}$-related premature mortality is likely to remain heavy irrespective of the planned improvement of air quality (HEI, 2016). To reduce the health effects of $PM_{2.5}$, China needs continuous efforts in pollution reduction. Measures and improvement should be systematically implemented based on the available scientific evidence.

3.7. Limitations and assumptions

Several methodological uncertainties are involved in the approach that limits the applicability of absolute numbers as concurrent deaths. In particular: - (1) The total population in a city exposed by the same average $PM_{2.5}$ concentration in the city, although the personal exposure monitoring in a city indicated high space-time variation (Jyoti et al., 2017). (2) The IER model, which integrates secondhand smoking, active smoking, and household air pollution cohort. Thus, there is inherent uncertainty involved in assessing health risks in different countries (Pope et al., 2018) (3) Air is a mixture of multiple pollutants which have synergistic effects on human health (Konishi et al., 2014) and this may underestimate the values in absolute terms. (4) It has been considered that the toxicity of $PM_{2.5}$ depends on mass concentration only, not on chemical composition. However, there is considerable evidence that the chemical composition, size distribution and sources of $PM_{2.5}$ also influence health effects (Pope et al., 2018). (5) The simple assumption in this study is that cause-specific baseline mortality rates remain constant in future, but non-communicable diseases have increased rapidly, due to urbanization, changing lifestyles, and aging (Cohen et al., 2017). (6) Short-term $PM_{2.5}$-mortality due to haze events have not been considered in the present study (Gao et al., 2015).

4. Conclusion

In this study, we estimated annual premature mortality and morbidity attributed to ambient $PM_{2.5}$ exposure and corresponding economic loss in 338 cities across 31 provinces of China in 2016. In the past, three exposure-response functions were developed by various research groups to estimate $PM_{2.5}$-related health effects, results of these three methodologies, the integrated exposure risk (IER) and log-linear (LL) and non-linear power law (NLP) has been compared over 338 cities in a most comprehensive evaluation. The economic costs for each case of different health endpoints vary based on socio-economic conditions across the country, therefore the health cost, for each of the 31 provinces, was adjusted with province level per capita GDP.

The estimated annual premature deaths of 0.964 (95% CI: 0.447, 1.355) million using IER function are lower than the estimates based on LL risk function [1.258 (95% CI: 1.053, 1.420) million] but higher than the estimations based on NPL risk function [0.770 (95% CI: 0.235, 1.267) million]. It is difficult to conclude which risk function is more appropriate for China in absence of any robust health and cohort study data. However, based on the mean estimate values, NPL lies in outcomes significantly overlap the uncertainty range with IER estimates, but not with LL outcomes. Most accepted IER approach uses evidence from exposures to household air pollution by solid fuel, second-hand smoke and active cigarette smoking to explore mortality exposure–risk relationships in high $PM_{2.5}$ concentration. Recently calculated *RR* values from a cohort study in high $PM_{2.5}$ concentration region in China mismatch with the RR values specified in IER approach. The China-specific cohort study helps to narrow the evidence gap for RR values in high concentration range but does not fit the IER, suggesting that the IER may be underestimating effects at the high $PM_{2.5}$ levels in China (Pope et al., 2018). It is crucial to initiate more China-specific cohort studies spanning across provinces and $PM_{2.5}$ exposure levels to improve *RR* estimates and validation to identify a fitting health impact assessment model.

The external costs of $PM_{2.5}$ pollution associated mortality and morbidity have been estimated to be 101.39 billion US$, which is 0.91% of the total GDP in China. If China manages to meet the national air quality standard in all the cities, annual premature deaths will be reduced by 8.3% in 2030. Coal and household biomass-burning emissions control must become a priority for Chinese pollution control authorities for $PM_{2.5}$ exposure reductions. A systemic solution to deal with air pollution requires source-specific control planning which can effectively be implemented across all provinces for a breathable air quality.

Appendix A

Table A1
Various studies on mortality attributed to $PM_{2.5}$ in China after the year 2010.

Reference	Study year	$PM_{2.5}$ data set	Baseline mortality rate	RR function	Total mortality (million)
Miao et al., 2017	2006	WRF-Chem model	National Bureau of Statistics of China (NBSC)	LL	1.7
Fu et al., 2015	2008	Satellite	WHO	LL	0.53 (lung cancer)
GDB 2010 (Lim et al., 2012)	2010	Satellite	WHO Global Database	IER	1.08
OECD, 2014	2010	Satellite	WHO		1.28
Apte et al., 2015	2010	Satellite	WHO	IER	1.27
Lelieveld et al., 2015	2010	EMAC- model	WHO	IER	1.36
R. Xie et al., 2016; Y. Xie et al., 2016	2010	Satellite	GDB dataset	IER	1.26
OECD, 2016					0.91
Giannadaki et al., 2016	2010	EMAC- model	WHO	IER	1.33
H. Zhao et al., 2017; L. Zhao et al., 2017	2010	Satellite	GDB dataset	IER	1.02
Wang et al., 2018	2010	Satellite	GDB dataset	IER	1.27
WHO, 2016	2012	Satellite	WHO		1.03
M. Liu et al., 2017a, 2017b; Y. Liu et al., 2017	2012	Satellite	NHFPCC, 2005–2013	IER	1.25
Liu et al., 2016	2013	Ground monitoring	OSPC, 2012; NBSC, 2014	IER	1.37
Fang et al., 2016	2013	Ground monitoring	NBSC, 2015	LL	3.03
GDB 2013 (IHME, 2016)	2013	Satellite	WHO	IER	0.91
Hu et al., 2017	2013	CMAQ model	WHO	IER	1.30

(continued on next page)

Table A1 (*continued*)

Reference	Study year	$PM_{2.5}$ data set	Baseline mortality rate	RR function	Total mortality (million)
Tian et al., 2017	2013	Satellite		IER	1.07
Rohde and Muller, 2015	2014	Ground monitoring	WHO	IER	1.6
Song et al., 2017a, 2017b	2015	Ground monitoring	NBSC and GDB dataset	IER	1.52
Zhang et al., 2017	2015	Ground monitoring – GIS	Matus et al., 2012	IER	1.85
Feng et al., 2017	2015	Ground monitoring – GIS	National Health and Family Planning Commission of China	IER	1.13
Zheng et al., 2017	2015	WRF–CMAQ model	Ministry of Health of China	IER	1.10
GDB 2015 (Cohen et al., 2017)	2015	Satellite	WHO	IER	1.11
Y. Xie et al., 2016	2015	Ground monitoring - GIS	Ministry of Health of China	LL	2.62
GDB 2016 (Abajobir et al., 2017)	2016	Satellite + GEOS-Chem	WHO	IER	1.08
Feng et al., 2017	2016	Ground monitoring	National health and family planning commission of China	IER	1.09

Appendix B. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envint.2018.09.024.

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