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Spatial and temporal trends in the mortality burden of air pollution in China: 2004–2012

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Abstract

While recent assessments have quantified the burden of air pollution at the national scale in China, air quality managers would benefit from assessments that disaggregate health impacts over regions and over time. We took advantage of a new 10 × 10 km satellite-based $PM_{2.5}$ dataset to analyze spatial and temporal trends of air pollution health impacts in China, from 2004 to 2012. Results showed that national $PM_{2.5}$ related deaths from stroke, ischemic heart disease and lung cancer increased from approximately 800,000 cases in 2004 to over 1.2 million cases in 2012. The health burden exhibited strong spatial variations, with high attributable deaths concentrated in regions including the Beijing–Tianjin Metropolitan Region, Yangtze River Delta, Pearl River Delta, Sichuan Basin, Shandong, Wuhan Metropolitan Region, Changsha–Zhuzhou–Xiangtan, Henan, and Anhui, which have heavy air pollution, high population density, or both. Increasing trends were found in most provinces, but with varied growth rates. While there was some evidence for improving air quality in recent years, this was offset somewhat by the countervailing influences of in–migration together with population growth. We recommend that priority areas for future national air pollution control policies be adjusted to better reflect the spatial hotspots of health burdens. Satellite-based exposure and health impact assessments can be a useful tool for tracking progress on both air quality and population health burden reductions.

Keywords

Air pollution; $PM_{2.5}$; Health burden; Temporal; Spatial; China

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1. Introduction

Over recent decades, rapid economic development has led to worsening air quality in China (Wang et al., 2014; Wu et al., 2012; Zhao et al., 2013a, b), which now ranks as one of the most polluted countries in the world (Van Donkelaar et al., 2010; Verstraeten et al., 2015). As a leading modifiable risk factor for non–communicable diseases, air pollution has attracted considerable interest from both the research and policy communities (Krewski et al., 2009; Laden et al., 2006; Pope et al., 2009; Pope and Dockery, 2006; Shang et al., 2013; West et al., 2016; Yang et al., 2013). However, assessments of health impacts, and their trends over time and space, have been hampered by the limited availability of relevant air monitoring prior to 2013 in China. Initial health impact studies used surrogate measures such as ambient PM_{10} concentrations measured at fixed monitoring stations to estimate health outcomes attributable to China's air pollution (Cheng et al., 2013; Hou et al., 2012; Matus et al., 2012; Zhang et al., 2008). However, PM_{10} is a less robust health–related exposure metric than $PM_{2.5}$ (USEPA, 2012), and central–site monitoring data may lead to exposure uncertainty related to spatial variability of PM, population density and demographic characteristics (Steinle et al., 2013).

In recent years, methods have been developed for estimating ground–level $PM_{2.5}$ concentrations based on satellite derived aerosol optical depth (AOD), providing a promising alternative for estimating exposure to outdoor $PM_{2.5}$ and associated health impacts, with more extensive spatial and temporal coverage (Brauer et al., 2012, 2015; Ma et al., 2014; Yao and Lu, 2014). For example, taking advantage of a global $PM_{2.5}$ dataset derived from AOD, the Global Burden of Disease (GBD) project reported that outdoor air pollution in China caused 1.2 million premature deaths and 25 million disability adjusted life years (DALY) losses in 2010 (Yang et al., 2013). A subsequent study reported a lower mortality burden of 0.35–0.50 million premature deaths, based on a different exposure methodology (Chen et al., 2013a). Both of these $PM_{2.5}$-based health assessments reported results only for a single year and for China as a whole.

Recently, as the Chinese government's demands for air quality management broadened from understanding the scope of the problem to targeting interventions, identification of spatial and temporal trends in health burdens attributable to air pollution is becoming more and more important. A recent study evaluated the temporal trend and spatial distributions of PM_{10}–related health impacts in China based on monitoring data from 2001 to 2011, contributing valuable insights into this question (Cheng et al., 2013). However, because PM_{10} trends may not track those of $PM_{2.5}$, spatial and temporal trends derived from PM_{10} data can be misleading (Cheng et al., 2013; Ma et al., 2016). Therefore, more accurate and refined information of the spatial–temporal characteristics of $PM_{2.5}$ effects are needed to support future policy interventions.

To address this need, we used a new 10 km resolution satellite derived $PM_{2.5}$ dataset in conjunction with fine scale population data to develop novel estimates of $PM_{2.5}$ related health damage in China from 2004 to 2012 at the subnational scale (Burnett et al., 2014; Ma et al., 2016; Yang et al., 2009). The implications of observed temporal trends and spatial distributions of impacts for future policy directions are explored.

2. Methods and data

While several air pollutants are known to have adverse health impacts, we focus here on outdoor $PM_{2.5}$ as the indicator of risk as it is widely regarded as the single best metric of air pollution–related risk to public health (USEPA, 2012). Regarding outcomes, we selected the premature deaths caused by stroke (International Classification of Diseases Revision 10 codes/ICD-10: I60–I69), ischemic heart disease (IHD, ICD-10: I20–I25), and lung cancer (LC, ICD-10: C33–C34) because of strong evidence for causal effects between these outcomes and $PM_{2.5}$ from prior high–quality epidemiological studies and sufficient cause-specific mortality data to estimate outcome–specific effect sizes per unit of exposure (Burnett et al., 2014; NHFPCC, 2005–2013). We excluded chronic obstructive pulmonary disease (COPD) deaths because the mortality data for COPD were not available. Annual cause-specific baseline mortality rates were taken from the China Health Statistical Yearbook.

Population weighted exposure (PWE) to $PM_{2.5}$ was calculated using $PM_{2.5}$ concentrations derived from satellite AOD at 10 km resolution and population distribution maps at 1 km resolution (Ma et al., 2016; Yang et al., 2009). Briefly, we developed and validated a model for estimating $PM_{2.5}$ based on collection 6 AOD retrieved by the US National Aeronautics and Space Administration (NASA) Moderate Resolution Imaging Spectroradiometer (MODIS), meteorology data, land use data and China's ground-monitoring $PM_{2.5}$ data in 2013. Historical daily $PM_{2.5}$ concentrations (2004–2012) were then estimated with this model using historical AOD data as inputs, assuming that the daily relationship between $PM_{2.5}$ and AOD was constant for the same Day of Year (DOY) in each year. This model was calibrated to minimize bias in the monthly and seasonal estimates on $PM_{2.5}$ and has been successfully applied in recent analyses (Chen et al., 2016; Liu et al., 2016b; Ma et al., 2016). The population distribution maps we adopted in this study also have been successfully applied in prior studies (Huang et al., 2014; Liu et al., 2013). In regard to relative risk per unit of exposure for each mortality outcome, the ideal one for the study population should be the results from Chinese cohort studies. However, to our knowledge, there are no published Chinese cohort studies directly examining the impacts of $PM_{2.5}$ on mortality so far. Most of existing Chinese cohort studies used the concentrations of SO_2, NO_2, TSP, or PM_{10} as exposure surrogates, which were poorer predictors of mortality compared with $PM_{2.5}$ (Cao et al., 2011; Liu et al., 2016a; Zeng et al., 2016; Zhou et al., 2014). In the few published $PM_{2.5}$ cohort studies (Guo et al., 2016; Qian et al., 2016), only the impacts of $PM_{2.5}$ on lung cancer incidence and preterm birth are discussed. Under this circumstance, the integrated exposure response (IER) model developed as part of the GBD study seems to be the most suitable choice for the estimates of mortality burdens attributable to $PM_{2.5}$ in China. Even compared with the RRs estimated based on Chinese TSP cohort and the conversion rate of $PM_{2.5}$/TSP, the RRs evaluated by the IER model can yield sensible results in the risk analysis over the range of concentrations that prevail in China (Burnett et al., 2014). 5 $\mu g/m^3$ of annual $PM_{2.5}$ was assumed as the counterfactual theoretical–minimum–risk exposure (Burnett et al., 2014). It should be noted that because the IER model was developed for adults, the age ranges for population and baseline incidence rates used in existing studies using the IER model are usually aged 25 and over or aged 30 and over (Lelieveld et al.,

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2015; Silva et al., 2016). However, the data of population and cause-specific baseline incidence rates with detailed age structures are not available in China at such a long time scale and fine spatial resolution. So we used the aggregated all-age data for population and baseline incidence rates in this study. While this might induce some bias, we think the biases are acceptable because most of baseline mortalities from IHD, stroke and lung cancer occur among adults (NHFPCC, 2005–2013). Using these inputs, we computed annual excess deaths attributable to outdoor $PM_{2.5}$ exposure as follows (Cheng et al., 2013; Zhang et al., 2008).

$$ED_{i,j,t}=\left(1-\frac{1}{RR_{i,j,t}}\right)*I_{i,j,t}*P_{i,j,t}=AF_{i,j,t}*I_{i,j,t}*P_{i,j,t}$$

where $ED_{i,j,t}$ is the excess deaths caused by air pollution for stroke, IHD or LC; $AF_{i,j,t}$ is the attributable fraction (AF), defined as the fraction of the disease burden attributable to $PM_{2.5}$;$I_{i,j,t}$ is the annual all-age incidence rate of the mortality end point, obtained from China Health Statistical Yearbook (2005–2013) compiled by the Ministry of Health of China according to ICD-10 codes (NHFPCC, 2005–2013); $P_{i,j,t}$ is the permanent all-age population aggregated from population density map at 1 km resolution provided by Chinese Academy of Science (Yang et al., 2009); $RR_{i,j,t}$ is the relative risk of premature mortality due to $PM_{2.5}$ obtained from the integrated exposure response (IER) model (Burnett et al., 2014) and *i*, *j*, and *t* are the grid, the prefecture and the year, respectively. Annual excess deaths caused by air pollution were estimated at 10 km resolution and also aggregated for 339 prefectures, 31 provinces and the whole of China, for each year from 2004 to 2012.

3. Results

During the period 2004–2012, over 93% of people in China lived in areas where $PM_{2.5}$ exceeded China's National Air Quality Standard for GradeII of 35 μg/m^3 (Fig. 1). Population weighted exposure (PWE) averaged between 67.1 and 76.7 μg/m^3 for China as a whole (Table S1). To visualize how regional impacts of $PM_{2.5}$ are distributed across the concentration range, we plot the distribution of population as a function of ambient $PM_{2.5}$ in 2004 (Fig. 1A) and 2012 (Fig. 1B). The distributions were bimodal, with one peak in the range 45–60 μg/m^3 and another around 80 μg/m^3. Compared with 2004, the population exposure shifted to greater extremes of $PM_{2.5}$ concentrations in 2012.

Total excess premature deaths due to stroke, IHD, and LC attributable to outdoor $PM_{2.5}$ exposure in China increased rapidly from 807 (95% CI: 328–1057) thousand cases in 2004 to 1250 (95% CI: 559–1672) thousand cases in 2012. Specifically, excess deaths due to stroke, IHD, and LC increased from 589 (95% CI: 209–736), 124 (95% CI: 86–194) and 93 (95% CI: 33–127) thousand cases in 2004 to 761 (95% CI: 274–947), 321 (95% CI: 224–498) and 169 (95% CI: 61–227) thousand cases in 2012 (Fig. 2), with the annual average growth rate of 3.25%, 12.63%, and 7.75%, respectively. In 2012, 43.9%, 29.0% and 27.9% of deaths caused by stroke, IHD and LC, respectively, in China were attributed to exposure to outdoor $PM_{2.5}$.

At the provincial level, the highest ambient $PM_{2.5}$ concentrations were observed in the Beijing–Tianjin Metropolitan Region, Henan, and Shandong, with PWEs of 102.4, 107.0, and 101.1 μg/m^3 in 2012, respectively. The Yangtze River Delta and Hunan experienced moderately high $PM_{2.5}$ concentration (~70–90 μg/m^3). The provincial variations of population–$PM_{2.5}$ distribution led to significant spatial heterogeneity in air pollution related excess deaths (Fig. S1–S2). Eight provinces - Henan (HeN), Shandong (SD), Hebei (HB), Jiangsu (JS), Sichuan (SC), Guangdong (GD), Anhui (AH) and Hunan (HuN) - contributed over 50% of the national excess deaths. Although the trend over time of excess premature deaths became flat or slightly declined in a few provinces after 2007, increases were seen in most provinces over the nine–year time window (Fig. S1–S2). The annual average growth rate for total excess premature deaths caused by stroke, IHD and LC varied from 0.74% (the lowest in Hainan) to 9.46% (the highest in Beijing).

Maps of mean $PM_{2.5}$-attributed deaths at 10 km resolution exhibited strong spatial gradients (Fig. 3). As expected, deaths were concentrated in regions including the Beijing–Tianjin Metropolitan Region, Yangtze River Delta, Pearl River Delta, Sichuan Basin, Shandong, Wuhan Metropolitan Region, Changsha–Zhuzhou–Xiangtan, Henan, and Anhui, which had heavy air pollution, high population density, or both. Over time, there was no significant expansion of high–excess deaths centers (Fig. 3A vs. B). In other words, more rapidly increasing excess deaths were mostly located in areas that already experienced high excess deaths in 2004.

4. Discussion

Using a satellite-derived reconstruction of $PM_{2.5}$ concentrations at 10 km resolution across China from 2004 to 2012, we observed significant variations of $PM_{2.5}$ related health burdens over time and across provinces. These new results add useful spatial and temporal dimensions to previous national scale, point-in-time health burden estimates for China (Chen et al., 2013a; Yang et al., 2013). There were several important differences in our analysis as compared to the GBD study. Whereas the GBD study based their estimates on the "registered population", here we used the higher and more relevant "permanent population", which captures health impacts related in–migration to more polluted areas. This led to slightly higher estimates here than in the GBD study (Fig. 2). Additional discrepancies in stroke are attributed to the different data sources of the annual incidence rate of mortality. Our results generally confirmed the results found in the 2010 GBD study and added additional spatial and temporal richness.

This study fills an important gap in historical air pollution related health risk assessment in China during the decade preceding the onset of national $PM_{2.5}$ monitoring. Though national excess premature deaths showed increasing trends from 2004 to 2012, some fluctuations were found in individual years after 2007. These fluctuations may reflect the intersection of three interacting trends: air quality change, in-migration induced by urbanization, and population growth. Regarding air quality, due to the combined effects of the global financial crisis starting in September of 2008 and stricter energy conservation and emissions reduction policy implemented in late 2006, national average $PM_{2.5}$ levels experienced a decline from 2007 to 2009 (Lo and Wang, 2013; Ma et al., 2016). Since 2010, $PM_{2.5}$ levels rose slightly due to the economic recovery but soon showed evidence of further decline due to the onset of new air pollution regulations. It could be concluded that without significant demographic changes, China's efforts on air pollution control would have led to reductions in the associated health damage by 2012. Unfortunately, these efforts were offset by the countervailing influence of in–migration driven by urbanization. The temporal variation of national average PWE (Table S1) illustrates the interaction between air quality change and in–migration on excess deaths. Before 2008, PWE rose at a faster rate (3.2 μg/m^3 per year) than the increase of $PM_{2.5}$ levels (1.97 μg/m^3 per year), which indicates that during the period of 2004–2008, in–migration patterns caused by rapid urbanization (Fig. S3) showed a strong trend to more polluted urban areas and hence intensified the health damage (Bai et al., 2014; Gong et al., 2012). Since 2008, the intersection between decreased $PM_{2.5}$ level and in–migration to more polluted areas led to fluctuations of PWE. As for population growth, it also played an important role in the increase of health damage.

We also observed significant spatial variations in PM-related mortality burdens. To better understand the spatial characteristics of health damage, we removed the effects of population size by calculating the attributable fractions (AF) of 31 provinces (Fig. 4). Generally, the provinces with higher AF are burdened with more deaths per thousand attributable to air pollution, holding all other independent factors constant. In the provinces with increasing AF over time, exposure to air pollution is becoming a more and more dangerous risk factor to public health. As shown in Fig. 4, although there were fluctuations for several provinces in individual years, most Chinese provinces showed increased attributable fractions during the period of 2004–2012, which supports the findings from GBD Study 2010 that China has undergone an epidemiological transition from communicable to non–communicable diseases in the past decade (Yang et al., 2013). However, that's not the whole story. There were still ten provinces (Fig. 4) including Hainan (HN), Guizhou (GZ), Guangdong (GD), Guangxi (GX), Fujian (FJ), Yunnan (YN), Shanghai (SH), Hunan (HuN), Jiangxi (JX), and Zhejiang (ZJ) that experienced a decline of AF from 2004 to 2012. All of these provinces are located in southern China, with abundant precipitation and high forest coverage, which is beneficial for the deposition of airborne particles (Guo et al., 2014; Hofman et al., 2014; Li et al., 2014). Also, the lack of winter heating in these southern regions leads to improved air quality and health (Chen et al., 2013b). Lastly, the sea–land breeze may be conducive to atmospheric pollutant dispersion in some coastal regions (Li et al., 2014).

Our findings on spatial characteristics of health burdens have important implications for air pollution control policies in China. In recent years, China's Ministry of Environmental Protection (CMEP) has adopted a "spatial–differentiation" air pollution control strategy that imposes stricter controls in regions where air quality problems are thought to be most severe. In the 12th Five–Year Plan (2011–2015) on Air Pollution Control in Key Regions, thirteen regions were designated as priority areas for air pollution prevention and control in 2012 (Chen et al., 2013a). To examine whether the designated priority areas covered the spatial hotspots of air pollution related health damage, we overlaid our $PM_{2.5}$ related mortality maps with the boundary map of the priority control regions (Fig. 3). It can be seen that most centers with high excess deaths were located within priority control regions. Compared with non-priority areas, these priority control areas suffered more serious air pollution and higher mortality burdens per squared kilometer (Table S2 and Fig. 3). From 2004 to 2012, air pollution levels and related mortality burdens in the priority control areas increased before 2008 and then showed slight improvements in recent years (see Table S2). The trends are similar to non-priority control areas, which suggest that the improvements may be attributed to the implementation of nationwide energy conservation and emissions reduction policy in the 11th Five–Year Plan rather than the designation of priority control areas in 2012 (Lo and Wang, 2013; Ma et al., 2016). However, from the historical experience we can expect that the differentiated and stricter emission control strategies in the priority control areas will bring more improvements of air quality and related health in the future compared with other areas. In addition, we found that Henan and Anhui Province are two regions which are suffering serious air pollution and related health damage but haven't yet received enough regulatory attention. From the perspective of health protection, we suggest that in the future, CMEP should expand the scope of the priority areas to include these two regions in the list.

While informative for policy decision making, our analysis has several limitations that should be kept in mind. The first one is related to the satellite-derived $PM_{2.5}$ dataset used in this study. Satellite data are sometimes missing due to the cloud and snow cover, potentially biasing estimates of $PM_{2.5}$ (Ma et al., 2016). To address the problem, we previously carried out a sensitivity analysis for each grid to test how many AOD-derived $PM_{2.5}$ estimations are needed to represent the true mean $PM_{2.5}$ concentrations. We found that a minimum of 6 valid satellite observations in a month or 11 in a season were sufficient to appropriately represent a monthly or seasonal average (Ma et al., 2016). In the current study, we extended the validation to the annual level, which showed that a minimum of 33 data points can predict the annual average $PM_{2.5}$ well (R2 = 0.83, slope = 1.039). Based on this criterion, we found few missing annual average $PM_{2.5}$ estimates, and those were mainly concentrated in western China where the population density is lower than 1 person per square kilometer (Fig. S4). Therefore, missing satellite observations were unlikely to significantly alter the estimates of health burdens in this study. Another limitation relates to spatial scale. While our $PM_{2.5}$ estimates were developed at a relatively fine scale of 10 × 10 km, this is still too coarse to capture fine patterns of pollution related to proximity of local sources emitting pollution at ground level, such as from traffic, cooking and heating with solid fuels, etc. (Chafe et al., 2014).

Another limitation that affects this study is related to the IER model we adopted. To test the effects of exposure response functions on the results, a sensitivity analysis has been conducted. Specifically, the RRs from IER model were replaced by the RRs projected based on a published Chinese TSP cohort study and the conversion rate of $PM_{2.5}$/TSP (Cao et al., 2011). Using the RRs in Cao et al. (2011), IHD and stroke mortality decreased by 2–10% and 7–14% while lung cancer mortality increased by 36–43% (Table S3). These differences are mainly caused by the non-linear shape of the IER function (Silva et al., 2016), which indicated that the estimates of mortality burdens are sensible to the exposure response function used in the assessment process. Fortunately, the estimates using the RRs in Cao et al. (2011) were still in the uncertainty range of the estimates using IER model. And the general spatial characteristics of mortality burdens were not significantly altered even though some grid-by-grid differences have been found (Fig. S5).

In addition, we glossed over the source classes responsible for $PM_{2.5}$ related mortality burdens in this study, which may also bias the estimates of mortality burdens. As we know, the coal-dominated energy structure in China determined that a large share of the fine particle pollution were from coal combustion sources. At the same time, however, coal combustion $PM_{2.5}$ posed a IHD risk roughly five times higher than for $PM_{2.5}$ mass in general, on a per microgram/cubic meter $PM_{2.5}$ basis (Thurston et al., 2016). Therefore, the assessment based on $PM_{2.5}$ mass in general in this study may underestimate the mortality burdens. More importantly, the underestimations may vary across different areas within China, smallest for the areas where coal combustion $PM_{2.5}$ accounted for a small fraction and highest for the areas where coal combustion sources contributed the most. This trend could further alter the spatial distributions of mortality burdens within China. Because little is known about the variations of source contributions within China, we cannot directly quantify the spatial effects at this stage. However, some qualitative judgments can be inferred from the observed evidence on the variations of $PM_{2.5}$ constituents because these are largely caused by the variations of source contributions (Thurston et al., 2016). Observations from representative cities across China suggest that compared to the western region, the fraction of the secondary aerosols and elemental carbon is larger in the east of China (Yang et al., 2011). The secondary aerosols especially sulfate particulate are highly related to the coal combustion source; and elemental carbon is a key trace constitute to identify the traffic source, which was shown to be source with the second largest IHD risk (Thurston et al., 2016). If all above is true, the hotspots of health burdens in the east of China identified by our study (Fig. 3) would be further intensified after considering the toxicity of $PM_{2.5}$ from different sources. Certainly, these speculations remain to be tested by future research on source appointment and source-specific toxicity of $PM_{2.5}$. However, it is clear that clean air efforts in China should focus on key source classes like coal combustion sources and traffic related sources to maximize public health benefits.

5. Conclusions

This study provides a refined estimation of historical health damage attributable to outdoor air pollution in China at 10 × 10 km resolution from 2004 to 2012. To our knowledge, this is the first study to quantify the health consequences of China's $PM_{2.5}$ pollution at both a long time scale and sub–national level. Though some limitations exist, this work contributes valuable insights into the strong spatial and temporal variations of air pollution related health burdens in China. The findings of this study provide important implications for urban planners and policy makers regarding priority areas of national joint air pollution prevention and control in China. Moreover, open publication of our health burden datasets also makes it possible to further explore the determinants of these health impacts.

Supplementary Material

Refer to Web version on PubMed Central for supplementary material.

Acknowledgments

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Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.envint.2016.10.003.

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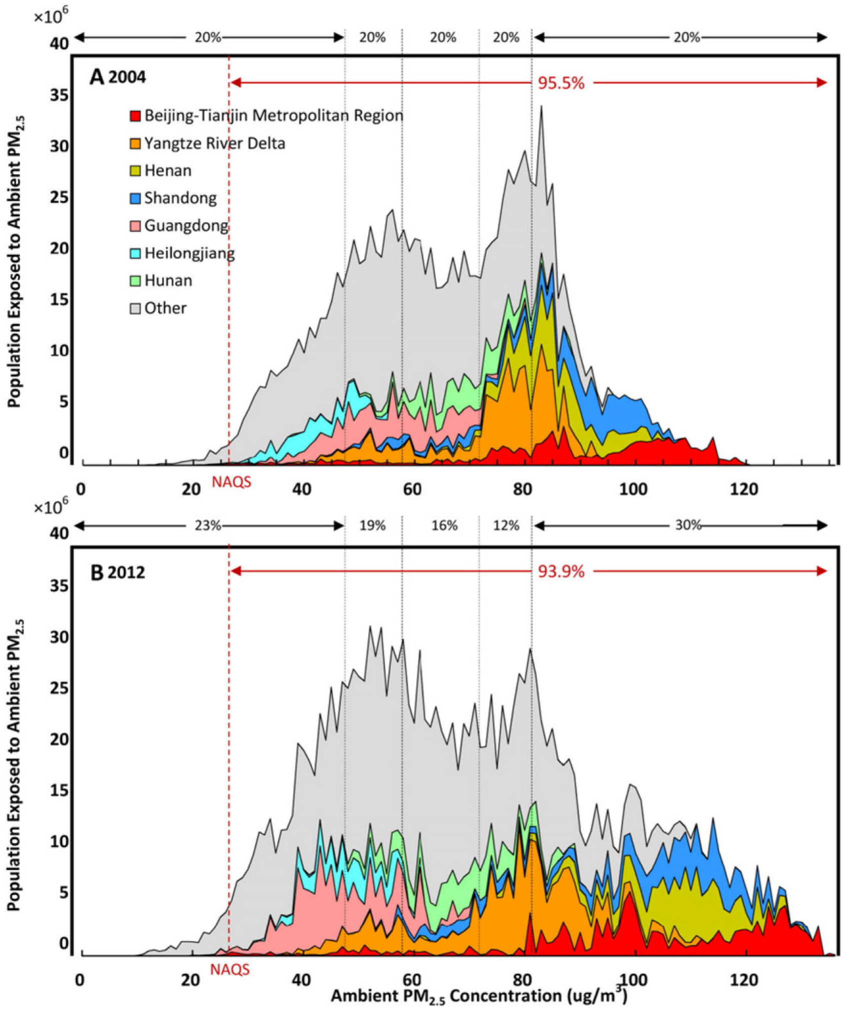

Fig. 1.

National and provincial distribution of population as a function of ambient $PM_{2.5}$ concentration in 2004 (A) and 2012 (B). Red dash vertical lines (in both plots) indicate the annual concentration limits in China's National Air Quality Standard for GradeII. Grey dash vertical lines (in both plots) demarcate boundaries of population quintiles that apportion the $PM_{2.5}$ concentration distribution into 5 bins with equal number of exposed population in 2004.

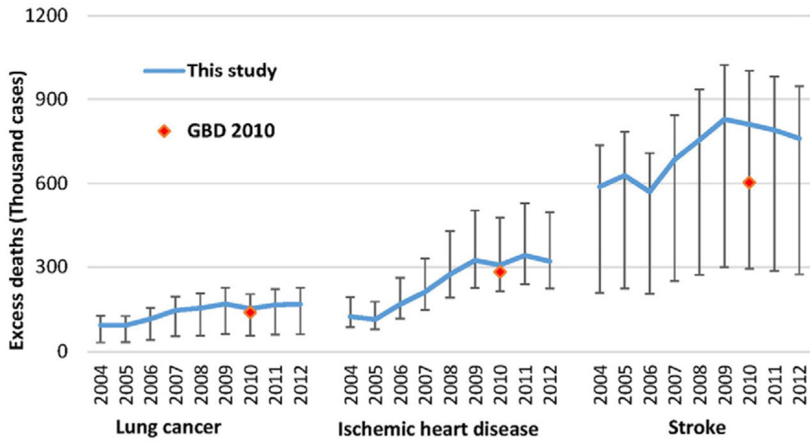

Fig. 2.

National excess deaths attributable to air pollution from 2004 to 2012. The grey error bars indicate the 95% confidence intervals of excess deaths. The error bars of excess deaths are determined by the 95% confidence intervals of relative risk, which are asymmetrical and in different directions for different outcomes at the higher end of $PM_{2.5}$ concentration.

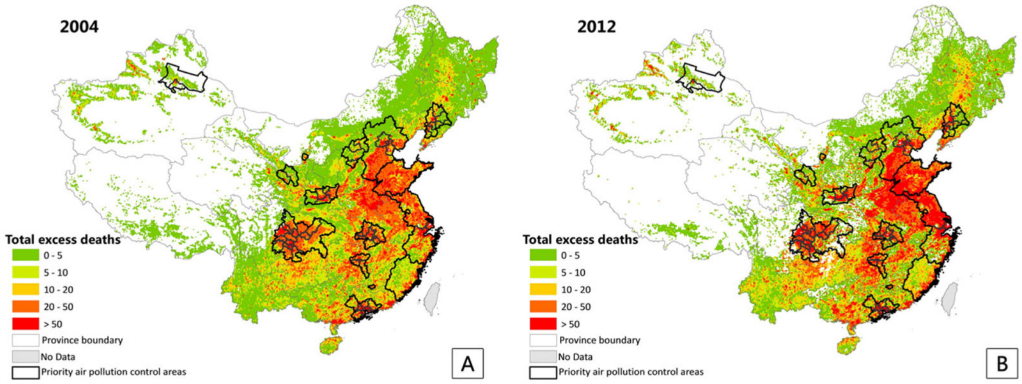

Fig. 3.
Spatial distributions of total excess deaths attributable to $PM_{2.5}$ exposure in 2004 (A) and 2012 (B) (cases per 100 km^2). Total deaths here are the sum of deaths caused by lung cancer, IHD and stroke. Black lines show the boundary of priority areas of national joint air pollution prevention and control.

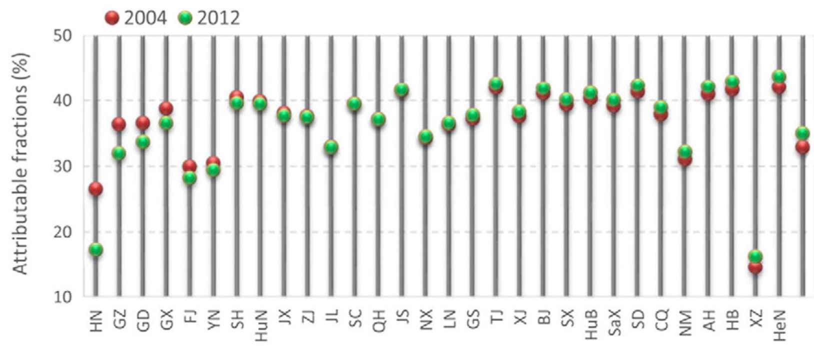

Fig. 4.

Averaged attributable fractions of total deaths for 31 provinces of China. Total deaths here are the sum of deaths caused by lung cancer, IHD and stroke.