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Health impacts and economic losses assessment of the 2013 severe haze event in Beijing area

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HIGHLIGHTS

- Health impacts of the 2013 Beijing haze event are estimated.
- Health-related economic losses are also calculated.
- The $PM_{2.5}$ concentrations in January 2013 might cause 690 deaths.
- This haze event might lead to 253.8 million US$ losses.

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ABSTRACT

Haze is a serious air pollution problem in China, especially in Beijing and surrounding areas, affecting visibility, public health and regional climate. In this study, the Weather Research and Forecasting-Chemistry (WRF-Chem) model was used to simulate $PM_{2.5}$ (particulate matters with aerodynamic diameter $\leq$2.5 μm) concentrations during the 2013 severe haze event in Beijing, and health impacts and health-related economic losses were calculated based on model results. Compared with surface monitoring data, the model results reflected pollution concentrations accurately (correlation coefficients between simulated and measured $PM_{2.5}$ were 0.7, 0.4, 0.5 and 0.6 in Beijing, Tianjin, Xianghe and Xinglong stations, respectively). Health impacts assessments show that the $PM_{2.5}$ concentrations in January might cause 690 (95% confidence interval (CI): (490, 890)) premature deaths, 45,350 (95% CI: (21,640, 57,860)) acute bronchitis and 23,720 (95% CI: (17,090, 29,710)) asthma cases in Beijing area. Results of the economic losses assessments suggest that the haze in January 2013 might lead to 253.8 (95% CI: (170.2, 331.2)) million US$ losses, accounting for 0.08% (95% CI: (0.05%, 0.1%)) of the total 2013 annual gross domestic product (GDP) of Beijing.

1. Introduction

Air pollution is a severe public health problem associated with industrialization and urbanization. The World Health Organization (WHO) Global Burden of Disease (GBD) project links over 3.2 million premature deaths worldwide in 2010 to ambient particulate matter (PM) pollution; furthermore, ambient PM pollution ranked 4th in health risk factors in East Asia in 2010 (Lim et al., 2012). Previous epidemiological studies have revealed robust associations between PM pollution and health effects (Dockery and Pope, 1994; Seaton et al., 1995) and recent studies provide strong evidence for the causal relationships between long and short term exposure to $PM_{2.5}$ and cardiovascular effects, mortality and likely causal relationships between long and short term exposure to $PM_{2.5}$ and respiratory effects (Burnett et al., 2014; U.S. Environmental Protection Agency, 2012). The evidence providing causal determination between $PM_{10-2.5}$ (aerodynamic diameter $\geq$2.5 μm but $\leq$10 μm), mortality and morbidity is not as strong as that for proving $PM_{2.5}$'s health impacts (U.S. Environmental Protection Agency, 2012). $PM_{2.5}$ may seriously affect human health because it is able to penetrate deeper into the lungs with small size, and various chemicals are absorbed on its surface (Pope and Dockery, 2006).

In January 2013, a severe and long-lasting haze episode occurred over eastern and northern China. According to the monitoring data by the Chinese Academy of Sciences (CAS), downtown Beijing's daily mean $PM_{2.5}$ concentrations exceeded 75 μg/m^3 (the Grade II daily $PM_{2.5}$ standard in

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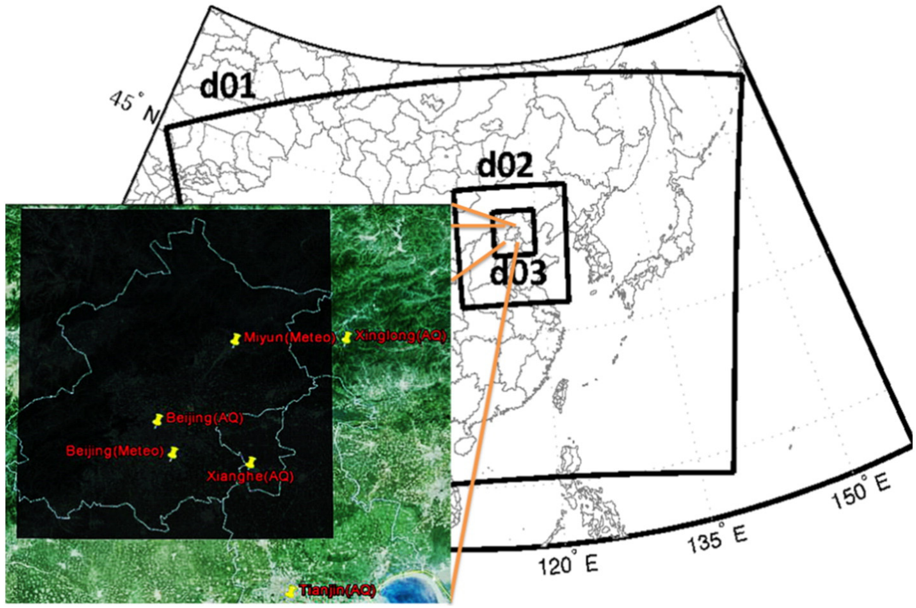

Fig. 1. WRF-Chem domain settings, study region and geographic locations of meteorological and air quality measurements.

China) for 70% days in January (He et al., 2014; Wang et al., 2013). As the political and cultural center of China, Beijing's air quality influences a large populace and is often prominently featured in global news and media channels. Extreme high population density increases the population weighted effects of air pollutants when air pollution events occur. To provide the basis for air pollution control, human exposures to air pollutants and health-related economic losses are needed to be quantified.

Most previous studies used monitoring data to estimate human exposures to PM (An et al., 2013; Hou et al., 2012; Zhang et al., 2008). While individual monitors have low measurement uncertainty, they do not provide or provide less information on spatial variability of PM; furthermore, monitors may not be operated continuously in time or may suffer from missing data due to mechanical and quality assurance failures. Air quality models provide an alternative method of establishing population exposure to outdoor PM, with the benefit of complete spatial and temporal coverage. For example, Marlier et al. (2012) includes the results from GEOS-Chem model to quantify health effects from fire emissions in Southeast Asia; The chemical transport model ATMoS was used to quantify health impacts of particulate pollution in Delhi (Guttikunda and Goel, 2013) and Hyderabad, India (Guttikunda and Kopakka, 2013). These studies have been based on annual average air pollution, but there are far fewer studies that focus on the evaluation of acute air pollution episodes, which have well documented impacts on mortality. Historical examples (summarized by Henschel et al., 2012) include 60 deaths in the Meuse Valley of France in 1930 (Godlee, 1991), 18 deaths in the small town of Donora, Pennsylvania in 1952 (Godlee, 1991), and 4000–12,000 excess deaths in London in 1952 (Godlee, 1991; Bell and Davis, 2001). In addition, time-series analysis shows that short-term increase in hospital admission rates is related to $PM_{2.5}$, especially for heart failure hospitalization, which increased 1.28% for every 10 μg/m^3 increase in daily mean $PM_{2.5}$ concentrations (Dominici et al., 2006).

In this paper, we aim to quantify the burden of $PM_{2.5}$ during the 2013 severe haze on both mortality and morbidity in Beijing area. Health-related economic losses are also calculated and the findings of this study can provide scientific basis for implementation of air pollution control strategies.

2. Data and methodology

Our integrated assessments include three steps: (1) define the study region and simulate $PM_{2.5}$ concentrations; (2) estimate human exposure and health impacts, including both mortality and morbidity; and (3) quantify the economic losses of those impacts. These three steps are described in detail here.

2.1. WRF-Chem settings and defining the study region

The 2013 haze event simulation was performed with WRF-Chem model. The model considers the interactions between meteorology and chemistry. Domain settings (definitions of simulation regions) in this study are the same as those of Jing-Jin-Ji modeled area of Yu et al. (2012). Three domains with two-way nesting were employed (Fig. 1) and grid resolutions are 81 km × 81 km, 27 km × 27 km and 9 km × 9 km from outer to inner domains. Beijing was set to be the center of the innermost domain. Gas-phase chemical mechanism CBMZ (Zaveri and Peters, 1999) coupled to the 8-bin sectional MOSAIC model with aqueous chemistry (Zaveri et al., 2008) was used. We used monthly 2010 Multi-resolution Emission Inventory for China

Table 1
β values and daily IRs.

Health endpoints	β values (per 10 μg/m^3, 95% CI) (%)	References	IRs (‰)	References
Mortality	0.53 (0.37,0.69)	Chen et al. (2011)	0.022377	BMBPH (2012)
Respiratory hospital admission	2 (1.33,2.67)	Aunan and Pan (2004), Zhang et al. (2007)	0.051925	BMBPH (2012)
Cardiovascular hospital	1.17 (0.5,1.83)	Aunan and Pan (2004), Zhang et al. (2007)	0.093509	BMBPH (2012)
Outpatient visits—internal medicine (15+)	0.57 (0.32,0.82)	Xu et al. (1995), Zhang et al. (2007)	2.92083	Zhang et al. (2007)
Outpatient visits—pediatrics (0–14)	0.65 (0.23,1.07)	Xu et al. (1995), Zhang et al. (2007)	0.811925	Zhang et al. (2007)
Acute bronchitis	9.17 (3.15,15.18)	Jing et al. (2000), Zhang et al. (2007)	0.140377	Zhang et al. (2007)
Asthma	2.1 (1.45,2.74)	Xie et al. (2009)	0.215982	Zhang et al. (2007)

Table 2
Unit losses for various health endpoints in Beijing (per case, US$) (Huang and Zhang, 2013).

Endpoints	Mortality	Acute bronchitis	Asthma	Clinic visit	Hospitalized
Cost per case (US$)	273,513.36	407.03	299.61	83.86	2761.04

(MEIC) (http://www.meicmodel.org/) as model anthropogenic emissions inputs. Spatial resolution of this emission inventory is 0.25 × 0.25°. Biogenic emissions were predicted hourly by the MEGAN algorithm (Guenther et al., 2006). Meteorological and chemical initial and boundary conditions were obtained from National Centers for Environmental Prediction (NCEP) Final Analysis and MOZART-4 (Model for Ozone and Related chemical Tracers, version 4) forecasts (Emmons et al., 2010), respectively. The study period is January 2013 and the last five days of December, 2012 were simulated as spin-up time to overcome the impacts of initial conditions. The study region is defined with the black rectangle (between 115.5°E and 117.54°E longitude and 39.42°N and 41.12°N latitude) shown in Fig. 1. Simulated $PM_{2.5}$ concentrations from the innermost domain were interpolated to the study region and the interpolated $PM_{2.5}$ concentrations were used to estimate human exposures and health impacts.

2.2. Estimating human exposures and health impacts

Cases of mortality and morbidity can be calculated using the Poisson regression model, shown as follows (Guttikunda and Kopakka, 2013):

$$\Delta E = \sum_{i=1}^{\#grids} \Delta POP * IR * \left(1 - \frac{1}{e^{(\beta * \Delta C)}}\right) \quad (1)$$

where

ΔE number of estimated cases of mortality and morbidity
ΔC the incremental concentration (use WHO 24-hour mean $PM_{2.5}$ guideline as reference)
ΔPOP the population exposed to the incremental concentration ΔC in grid i
IR incidence rate of the mortality and morbidity end points
β the concentration–response function, defined as the change in number cases per unit change in concentration per capita.

Simulated daily mean $PM_{2.5}$ concentrations were used to estimate exposure level using function (1). In the calculation, WHO 24-hour mean $PM_{2.5}$ guideline value (25 μg/m^3) was used as reference to obtain the incremental concentration. The gridded population data we used are from the Gridded Population of the World (GPW) future estimates for 2015 dataset. The resolution of this dataset is 2.5 arcmin for each grid. According to the results of the sixth population census (http://data.stats.gov.cn), 8.6% residents are between 0 and 14 years old and 8.7% residents are above 65 years old. For children, adults and old people, concentration–response functions are different. Concentration–response function and incidence rate are important in health impacts evaluation and they have variations for different countries and regions.

In this study, β values and IRs were cited from published findings. Some studies using PM_{10} for exposure assessment were also included and the following conversion relation was used: $PM_{2.5} = PM_{10} \times 0.6$ (Kan and Chen, 2004; Teng et al., 1999). The short-term β values and daily IRs are summarized in Table 1. The β values represent the increase in daily mortality and morbidity cases corresponding to a 10 μg/m^3 increase of $PM_{2.5}$ concentration. Chen et al. (2011) estimated the short-term association between $PM_{2.5}$ and daily mortality in Beijing using time-series method. For $PM_{2.5}$'s acute effects on respiratory and cardiovascular hospital admission, the β values were collected from the meta-analysis results in China (Aunan and Pan, 2004). Xu et al. (1995) investigated the association of air pollution with daily outpatient visits in Beijing and Jing et al. (2000) provided the relation between air pollution and new occurrence of chronic bronchitis in Benxi, China using statistical regression model. The β value for asthma was also obtained using meta-analysis method by Xie et al. (2009). Mortality and hospitalization IRs were cited from BMBPH (2012). Due to the limited data available, other IRs were cited from Zhang et al. (2007). These IRs were converted from annually to daily with the assumption that the cases took place equally in each day (Xie et al., 2014).

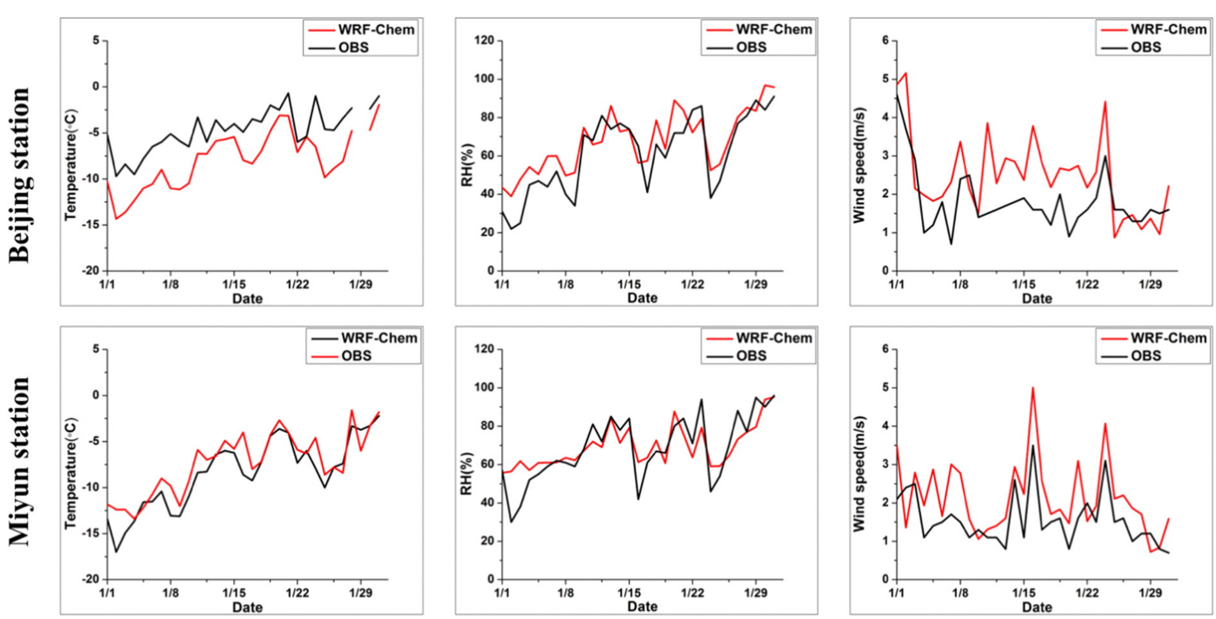

Fig. 2. Observed (black lines) and simulated (red lines) daily meteorological variables at Beijing and Miyun stations. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

Table 3
Performance statistics for meteorological predictions at Beijing and Miyun stations.

	Bejing			Miyun		
	MB	ME	RMSE	MB	ME	RMSE
Temperature (°C)	−3.2	3.2	3.6	−1.1	1.4	1.8
RH (%)	6.3	9.7	11.1	0.9	7.3	9.9
Wind speed (m/s)	0.7	0.9	1.0	0.6	0.7	0.9

2.3. Economic valuation of the health impacts

We further evaluated the economic losses of the health impacts associated with the high $PM_{2.5}$ concentrations during haze. Here we used the unit economic losses of related health endpoints for Beijing area borrowed from Huang and Zhang (2013), listed in Table 2. The unit economic cost of mortality was assessed using Value of a Statistical Life (VSL) method, which indicates how much people would be willing to pay (WTP) for a reduction of death. For the case that some morbidity endpoints cannot be valued from WTP literatures, the Cost of Illness (COI) method was used. COI considers both health expenditures and loss of labor productivity. It is shown as below (Huang and Zhang, 2013):

$$C_i = \left(C_{pi} + \mathrm{GDP}_p * T_{Li}\right) * \Delta E_i \quad (2)$$

where:

C_i	total economic cost for end point i
C_{pi}	health expenditures for end point i
GDP_p	daily GDP per capita
T_{Li}	loss of labor time due to end point i
ΔE_i	health impacts of end point i.

3. Model evaluation

In this section, surface meteorological and air quality measurements are used to evaluate model performance. Fig. 2 shows the time series of simulated and observed daily mean temperature, relative humidity

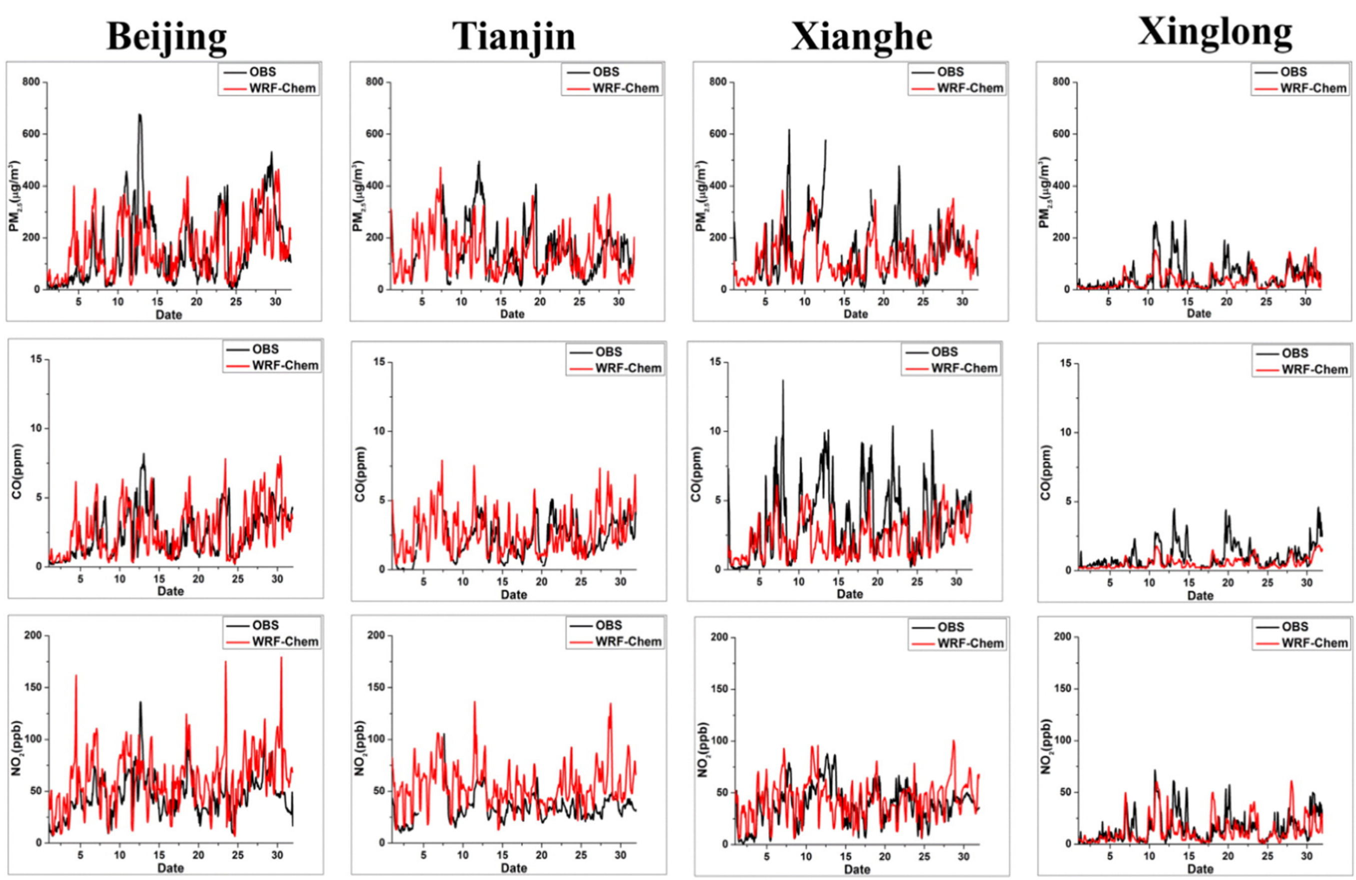

Fig. 3. Observed (black lines) and simulated (red lines) hourly $PM_{2.5}$, CO and NO_2 at Beijing, Tianjin, Xianghe and Xinglong stations. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

Table 4
Performance statistics for $PM_{2.5}$ at four stations.

Stations	Observation mean (μg/m³)	Model mean (μg/m³)	R	MB (μg/m³)	ME (μg/m³)	MNB (%)	MNE (%)	NMB (%)	NME (%)	MFB (%)	MFE (%)
Beijing	158.5	160.9	0.7	2.4	70.8	40.7	64.2	1.5	44.0	19.2	48.9
Tianjin	158.2	144.5	0.4	−13.7	64.4	17.4	53.7	−8.7	44.6	2.6	44.8
Xianghe	152.5	138.6	0.5	−13.9	55.6	13.7	43.0	−9.1	40.1	1.0	39.7
Xinglong	50.8	38.0	0.6	−12.8	20.6	−8.9	39.1	−25.2	54.3	−21.7	44.1

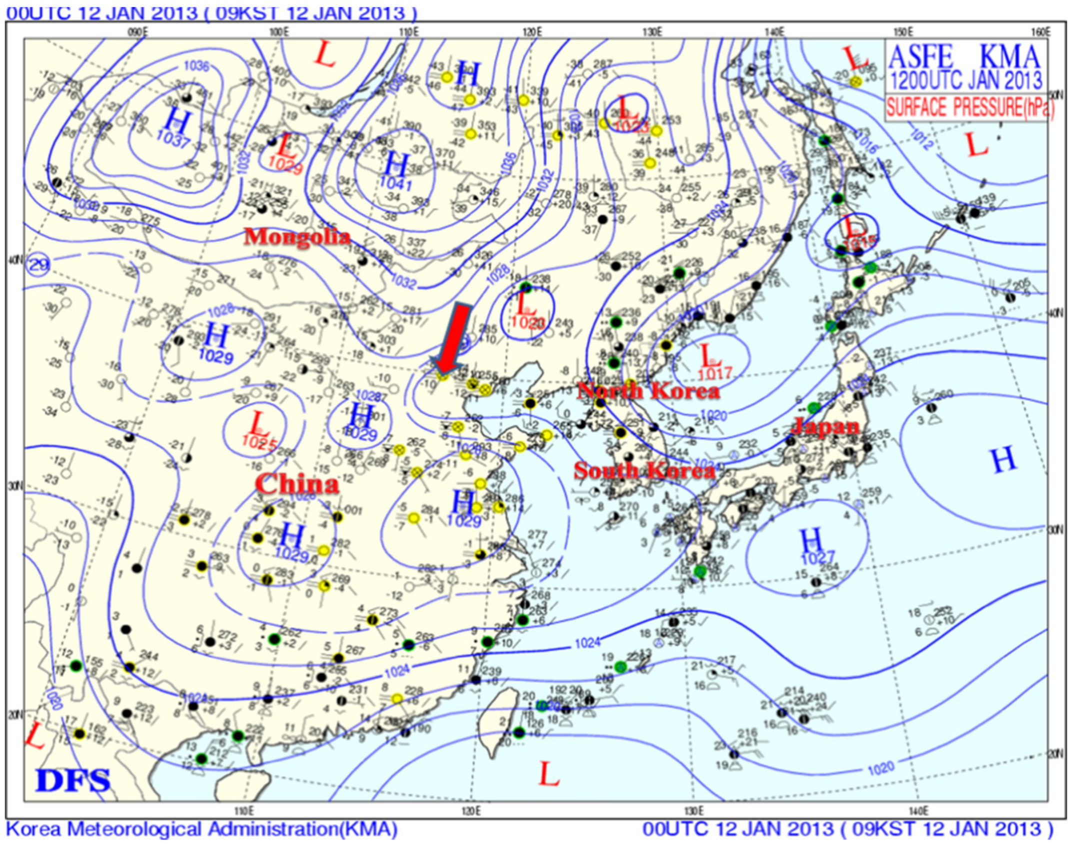

Fig. 4. Weather maps at 0000 UTC on January 12th, 2013.

(RH) and wind speed at Beijing and Miyun stations. The geographic locations of these two stations are marked in Fig. 1. The meteorological measurements were collected from the surface stations of the Chinese National Meteorological Center (http://cdc.cma.gov.cn/home.do). Local-scale meteorology is an important driver of regional air pollution (Pearce et al., 2011), so the accuracy of simulated meteorological variables is critical to the performance of air quality modeling. Simulated temperature, relative humidity and wind speed capture the main features and variations as observed at both Beijing and Miyun stations, although small underestimation of temperature (~−3 °C) and overestimation of wind speed exist at Beijing station. Mean bias (MB), mean error (ME) and root mean square error (RMSE) are calculated for temperature, relative humidity and wind speed at the abovementioned two stations and these evaluation metrics are summarized in Table 3. Emery et al. (2001) proposed that good model performance would be classified as temperature bias smaller than 0.5°, wind speed RMSE smaller than 2 m/s, and wind speed bias smaller than 0.5 m/s. In general, the model performs better at Miyun station than at Beijing station.

The simulated hourly $PM_{2.5}$, CO and NO_2 were compared with air quality measurements at Beijing, Tianjin, Xianghe, and Xinglong stations in the CARE-China network. The geographic locations of these four stations are marked in Fig. 1. As shown in Fig. 3, the WRF-Chem model generally captured the hourly variation of $PM_{2.5}$, CO and NO_2 at these four stations. Underestimations of CO exist at Xianghe and Xinglong stations compared to measured concentrations. This might be caused by large uncertainties in emission inventories. On January 13, there is also an underestimation of $PM_{2.5}$. The underestimations of air pollutants may be caused by the overestimation of wind speeds (e.g. Fig. 2). Since this paper focuses on the health impacts assessment of $PM_{2.5}$ using numerical modeling results, the accuracy of modeling $PM_{2.5}$ plays a crucial role in the assessment. MB, ME, correlation coefficients R, the normalized mean bias (NMB), the normalized mean error (NME), the mean fractional bias (MFB), and the mean fractional error (MFE) were calculated versus 24-h observations for simulated and measured $PM_{2.5}$ at four stations. The MB values for $PM_{2.5}$ at Beijing, Tianjin, Xianghe and Xinglong are 2.4, −13.7, −13.9 and −12.8, respectively (Table 4). Boylan and Russell (2006) proposed that MFB should be within ±60% and MFE should be below 75% for a satisfactory model performance. For our results, MFB values are within 22% and MFE values are below 50%, indicating that our model results are good and reasonably represent the real pollution states. As shown in Table 4, $PM_{2.5}$ is slightly overestimated at Beijing station and underestimated at other three stations. Large uncertainties in aerosol emissions and lack of some secondary aerosol formation mechanisms in WRF-Chem are the two main reasons for the overestimation and underestimation. During haze events, secondary aerosols play an important role (Sun et al., 2014), which has not been well represented in WRF-Chem model.

4. Results and discussion

4.1. Causing factors of this event

4.1.1. Weather system

As shown in Fig. 3, $PM_{2.5}$ in Beijing reached peak value (677 μg/m^3) at 17:00 on January 12th. Fig. 4 shows the surface weather analysis on that day (0000 UTC). The figure is provided by the Korea Meteorological Administration (KMA), and Beijing city is denoted with a red arrow in the figure. There were two weak high pressure systems, one in the western direction and another one in the southern direction of Beijing. Pressure gradient force was very small, and wind speeds were weak around the Beijing area, which is unfavorable to the dispersion of air pollutants.

4.1.2. Boundary layer

Fig. 6 shows observed vertical temperature and relative humidity profiles at 0000 UTC and 1200 UTC on January 12th, 2013. These atmospheric sounding data are retrieved from the NCAR Earth observing

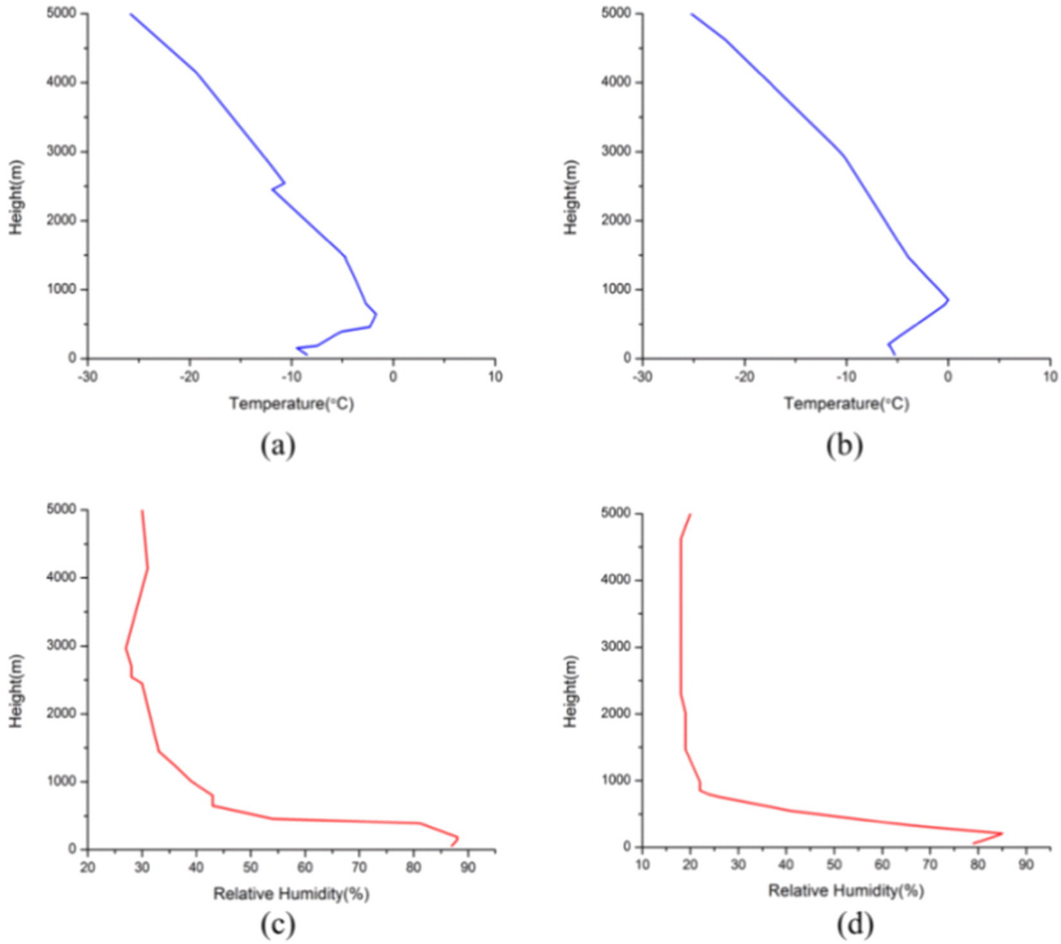

Fig. 5. Observed vertical profile of (a) temperature at 0800 (0000 UTC), (b) temperature at 2000 (1200 UTC), (c) relative humidity at 0800 (0000 UTC), and (d) relative humidity at 2000 (1200 UTC) on January 12th in Beijing.

laboratory (http://weather.uwyo.edu/upperair/sounding.html). Obvious strong temperature inversions are apparent up to heights of 500 and 1000 m (Fig. 5). The temperature inversions are 10 °C and 5 °C, indicating extreme stable vertical conditions near surface. Fig. 5(c) and (d) shows that relative humidity near surface on January 12th was above 80%. Sun et al (2006) pointed out that high RH can accelerate the formation of secondary species, such as sulfate and nitrate. The stable boundary layer and high RH near surface aggravate the pollution. In addition, Zheng et al. (2014) pointed out that regional transport played a significant role in this haze event.

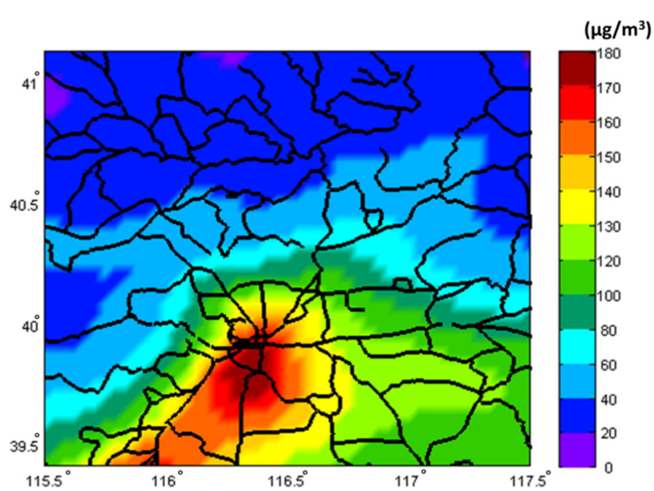

Fig. 6. Horizontal distribution of monthly averaged $PM_{2.5}$ (x axis means longitude and y axis means latitude).

4.2. Simulated $PM_{2.5}$ concentrations

The simulated monthly averaged $PM_{2.5}$ concentrations are displayed in Fig. 6 and daily averaged $PM_{2.5}$ concentrations are shown in Fig. 7. In the center of Beijing city, monthly averaged $PM_{2.5}$ is above 170 μg/m^3 and the pollution level for the north part of Beijing is lower than it in the south part. Rural areas are located in the north of Beijing and economically active areas are in the south of Beijing. As shown in Fig. 7, there are five small haze episodes in January, from 3 to 7, from 9 to 13, from 17 to 19, from 21 to 23 and from 25 to 31. High PM concentrations happen around urban area and extend to all southern regions.

4.3. Mortality and morbidity losses

Daily health impacts of $PM_{2.5}$ were calculated using Eq. (1) for the Beijing area specified in Fig. 1 by the black area. Total mortality during haze in Beijing in January was estimated to be 690, with the 95% confidence interval from 490 to 890. The total morbidities in January are summarized in Table 5. For comparison, Xie et al. (2014) estimated 201 deaths during period from 10 to 15 January, 2013. Li et al. (2013) estimated 725 premature deaths for the 2013 January haze in Beijing and Zhang et al. (2013) calculated 2725 premature deaths for 12 cities in NCP area from 10 to 31 January, 2013. These three studies used station measurements to represent pollution levels in the city. But pollution

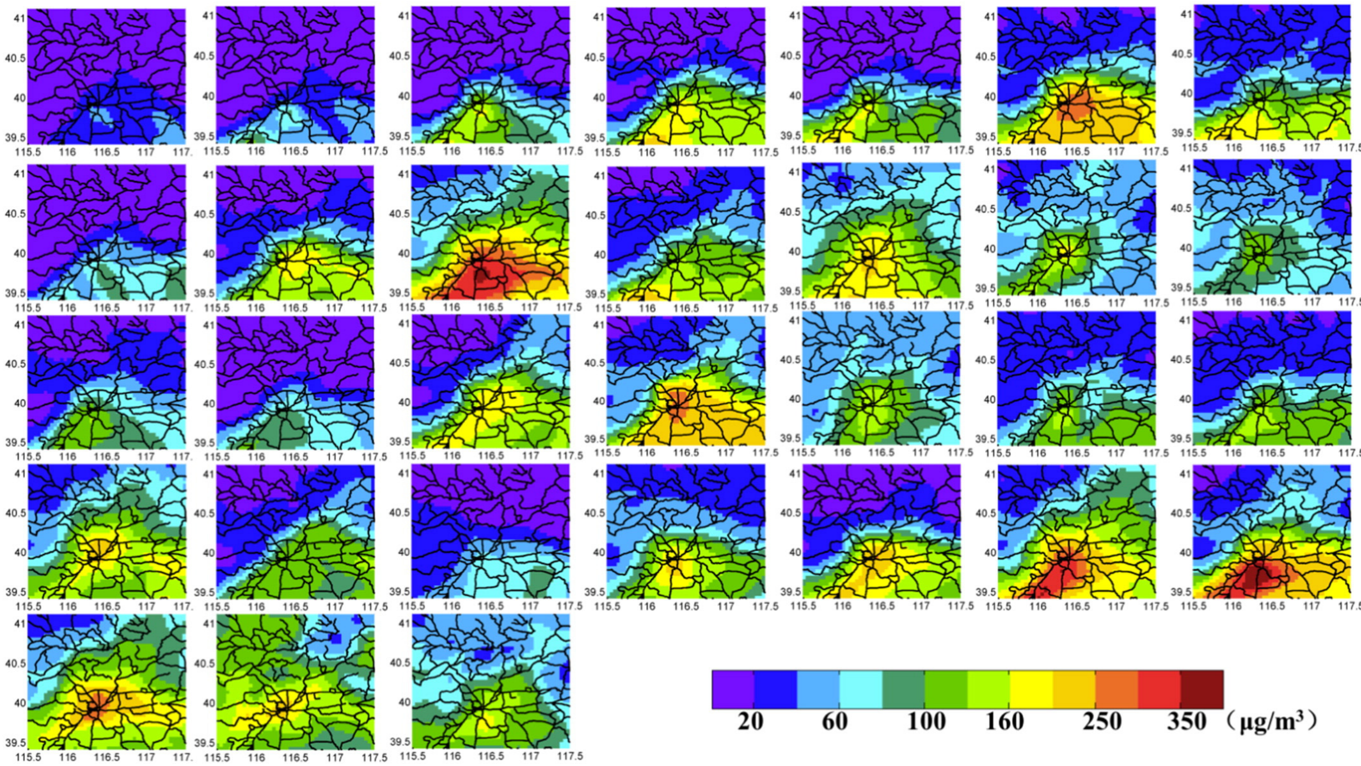

Fig. 7. Horizontal distribution of daily (from January 1 to January 31) averaged $PM_{2.5}$ (x axis means longitude and y axis means latitude).

has high variation from urban to rural areas. That people live in urban and rural areas are exposed to different pollution levels should be considered in air pollution exposure studies. For this study, monitoring data from four sites were used to validate the model results. As shown in Table 4, model performs well compared with observations but with lower mean at Tianjin, Xianghe and Xinglong stations. The simulated fields provide more detailed information regarding concentration gradients and thus our estimates may be more realistic.

4.4. Economic losses

Using the unit economic losses of various health endpoints, we calculated the economic losses for the haze in January. The total GDP of Beijing was 317,302.36 million US$ in 2013 (http://data.stats.gov.cn) and the total economic losses for this haze event in January is estimated at 253.8 (95% CI: (170.2, 331.2)) million US$, accounting for 0.08% (95% CI: (0.05%, 0.1%)) of the total annual GDP of Beijing. Detailed economic losses of each health endpoint are listed in Table 6. The calculations of confidence intervals in Table 6 only consider the uncertainties of β values. For all economic losses, economic losses due to deaths account for about 74.3% of the total losses. For economic losses of morbidity, the losses of hospitalization and acute bronchitis are dominant.

4.5. Uncertainty discussion

Evaluating health impacts and economic loss of air pollution is becoming critical since it can analyze the cost–benefits of air pollution control measures. Previous studies assume that the whole population was exposed to the average concentration levels recorded in monitoring stations (Kan and Chen, 2004). In this study, WRF-Chem was used to overcome this unreasonable assumption. However, using simulated $PM_{2.5}$ to approximate human exposure also has uncertainties. As mentioned in the model evaluation section, WRF-Chem overestimated $PM_{2.5}$ concentration in Beijing station slightly, and underestimated in other three stations. It's difficult to quantify how uncertainties of WRF-Chem affect the results in this study since data from only one monitoring station in Beijing area are available. For the available Beijing station, slightly overestimated $PM_{2.5}$ could lead overestimations of health impacts and economic losses. The sensitivity of health impacts and economic losses to $PM_{2.5}$ concentrations is calculated and summarized in Table 7. Estimated premature deaths increase 70 (about 10% increase) and estimated economic losses increase 25.7 million US$ (about 10% increase) when simulated $PM_{2.5}$ concentrations increase 10 μg/m^3. More studies are needed to reduce uncertainties in air quality modeling.

In the calculation of health impacts, it is crucial to choose the reference concentration. In this study, we used the WHO 24-hour mean $PM_{2.5}$ guideline 25 μg/m^3 as the reference concentration with the assumption that there are no health impacts under this threshold. At present, there is no scientific basis to set a threshold to evaluate health impacts, but the threshold is commonly used in health impacts evaluation studies. For example, Guttikunda and Goel (2013) set 10 μg/m^3 as the threshold value for the impact analysis. In addition, we explored the sensitivity of estimated premature deaths and total economic losses

Table 5
Health impacts estimates for January.

Mortality	Hospitalized for respiratory disease	Hospitalized for cardiovascular disease	Clinic visit (ages 0–14)	Clinic visit (age above 15)	Acute bronchitis	Asthma
690 (95% CI: (490, 890))	5470 (95% CI: (3800, 6990))	6080 (95% CI: (2720, 9190))	2610 (95% CI: (950, 4180))	88,160 (95% CI: (50,360, 124,660))	45,350 (95% CI: (21,640, 57,860))	23,720 (95% CI: (17,090, 29,710))

Table 6
Economic loss estimates.

Health Endpoints	Economic cost (million US$)
Mortality	188.7 (134, 243.4)
Hospitalization	31.9 (18,44.5)
Clinic visits	7.6 (4.3, 10.8)
Acute bronchitis	18.5 (8.8, 23.6)
Asthma	7.1 (5.1, 8.9)
Total	253.8 (170.2, 331.2)

Table 7
Sensitivity of health impacts and economic losses to 10 μg/m^3 increase of $PM_{2.5}$ concentrations (mean and 95% CI).

Deaths	Total economic losses (million US$)
70 (50, 90)	25.7 (17.5, 33.1)

Table 8
Sensitivity analysis of $PM_{2.5}$-related deaths and total economic losses using different thresholds.

Thresholds	Death number	Total economic losses (million US$)
0 μg/m^3	880 (620, 1130)	322.5 (215.6, 418. 7)
25 μg/m^3	690 (490, 890)	253.8 (170.2, 331.2)
75 μg/m^3	380 (270, 490)	141 (94.2, 184.8)

to different threshold assumptions. As shown in Table 8, the assumed threshold could significantly affect the estimations.

Concentration–response function is another important factor in health impacts assessment studies. In this study, most concentration–response functions used are from meta-analyses, which can avoid errors from individual study. Apart from this, 95% CI is included to represent the uncertainties caused by concentration–response functions. Burnett et al. (2014) considered nonlinear relation between $PM_{2.5}$ exposure and excess mortality relative risk, but it is ignored in this study, which could lead to overestimations of damages at high $PM_{2.5}$ levels. For health endpoint selection, only those endpoints that could be quantified are chosen, leading to an underestimation of the impacts. Furthermore, only acute effects are assessed in this study, while air pollution also has chronic effects on public health. Based on the abovementioned uncertainties, our assessments are very conservative.

5. Summary

During the 2013 January haze event in Beijing, $PM_{2.5}$ concentrations were extremely high, with maximum hourly concentration about 650 μg/m^3 and the high PM concentrations persisted for a long time. As mentioned above, ambient particulate pollution is one of the top 5 risk factors of deaths in East Asia (Lim et al., 2012) and there have been many examples in history that air pollution events caused extensive premature deaths and diseases. Summarizing methods of evaluating heath impacts and economic losses, we established an approach to assess health impacts and economic losses of the 2013 January severe haze event in Beijing, using simulated $PM_{2.5}$ concentrations, which were verified with both meteorology and air quality measurements, dose–response functions from epidemiology studies in Beijing area, and gridded population data. The results show that the $PM_{2.5}$ pollution levels in January might cause 690 (95% CI: (490, 890)) premature deaths and 253.8 (95% CI: (170.2, 331.2)) million US$ economic losses. We only evaluated the health impacts of $PM_{2.5}$ and the impacts of other air pollutants, like ozone, CO, are not included. Considering many aspects, including model uncertainty and reference concentration selection, our assessments are conservative. The results imply the severity of haze event from both health and economic loss perspectives.

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