NIH Public Access

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Sci Total Environ. Author manuscript; available in PMC 2014 April 15.

Published in final edited form as:

Sci Total Environ. 2013 April 15; 0: . doi:10.1016/j.scitotenv.2013.02.040.

Seasonal variation in the acute effect of particulate air pollution on mortality in the China Air Pollution and Health Effects Study (CAPES)

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Abstract

Epidemiological findings concerning the seasonal variation in the acute effect of particulate matter (PM) are inconsistent. We investigated the seasonality in the association between PM with an aerodynamic diameter of less than 10 μm (PM_{10}) and daily mortality in 17 Chinese cities. We fitted the "main" time-series model after adjustment for time-varying confounders using smooth functions with natural splines. We established a "seasonal" model to obtain the season-specific effect estimates of PM_{10}, and a "harmonic" model to show the seasonal pattern that allows PM_{10} effects to vary smoothly with the day in a year. At the national level, a 10 μg/m^3 increase in the two-day moving average concentrations (lag 01) of PM_{10} was associated with 0.45% [95% posterior interval (PI), 0.15% to 0.76%], 0.17% (95% PI, −0.09% to 0.43%), 0.55% (95% PI, 0.15% to 0.96%) and 0.25% (95%PI, −0.05% to 0.56%) increases in total mortality for winter, spring, summer and fall, respectively. For the smoothly-varying plots of seasonality, we identified a two-peak pattern in winter and summer. The observed seasonal pattern was generally insensitive to model specifications. Our analyses suggest that the acute effect of particulate air pollution could vary by seasons with the largest effect in winter and summer in China. To our knowledge, this is the first multicity study in developing countries to analyze the seasonal variations of PM-related health effects.

Keywords

Seasonality; Air pollution; Particulate matter; Mortality; Time series

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Conflict of interest statement

The authors declare that there are no conflict of interest.

1. Introduction

Among the various air pollutants, particulate matter (PM) has shown the strongest evidence for adverse health effects (Pope et al., 2006). Numerous studies have demonstrated that short-term exposure to PM was associated with increased risk of mortality and morbidity (Pope et al., 2006). Because the sources and components of the PM mixture are known to vary throughout the year, in addition to the different exposure patterns of the population in different seasons, it is plausible that the short-term associations between particulate air pollution and daily mortality may change from season to season (Peng et al., 2005). Therefore, season is an important modifying factor when investigating the acute health effects of air pollution. A few multicity studies and single-city studies have provided evidence of seasonality for the short-term health effects of PM (Peng et al., 2005; Pope et al., 2006; Qian et al., 2010; Zeka et al., 2006). However, the evidence of seasonal variation in these associations is inconsistent. Researchers have identified peak effects of PM in winter, summer or transition seasons (spring/fall) in different settings, among which most were conducted in developed countries (Peng et al., 2005; Pope et al., 2006; Qian et al., 2010; Zeka et al., 2006). There remains a need for studies in cities of developing countries, where characteristics of outdoor air pollution (e.g. air pollution level and mixture, transport of pollutants) and socio-demographic status of local residents may be different from North America and Europe (Kan et al., 2012).

As the largest developing country, China may have the worst PM pollution in the world. Evidence regarding the seasonality of the acute effect of air pollution is limited and inconsistent in China. For example, in Shanghai (Kan et al., 2008) and Hong Kong (Wong et al., 2001), PM_{10} was shown to have higher effects in the cool season (October to April). Similarly, another study in Wuhan (Qian et al., 2010) estimated the largest effects in winter. In contrast, a recent analysis in Shenyang (Ma et al., 2011) indicated a much stronger association of fine particles with daily mortality in warm seasons (May to October). Therefore, a multicity study with common methodology is needed to clarify the seasonality in air pollution health impact in China.

The objective of this paper is to examine the seasonal pattern in the short-term associations of PM_{10} with daily mortality in 17 cities in the China Air Pollution and Health Effects Study (CAPES).

2. Materials and methods

2.1. Data collection

The CAPES database of daily mortality, air pollution and weather condition covers 17 main Chinese cities: Anshan, Beijing, Fuzhou, Guangzhou, Hangzhou, Hong Kong, Lanzhou, Nanjing, Shanghai, Shen-yang, Suzhou, Taiyuan, Tangshan, Tianjin, Urumqi, Wuhan, and Xi'an (Fig. 1). The 17 CAPES cities were divided into 3 regions according to their latitudes: North (Anshan, Beijing, Lanzhou, Shenyang, Taiyuan, Tangshan, Tianjin, Urumqi, and Xi'an), Middle (Hangzhou, Nanjing, Shanghai, Suzhou and Wuhan), and South (Fuzhou, Guangzhou and Hong Kong). As illustrated in Fig. 1, these cities generally represent different geographical and climatic characteristics. Specifically, the northern cities have a long cold winter and a short warm summer; the cities in the middle have distinct seasons with cold winter and hot summer; and the climate of southern cities is characteristic of a long and hot summer and a short warm winter. Our study areas were restricted to the urban areas of these cities, because of inadequate air pollution monitoring stations in the suburban areas.

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The daily mortality data of urban residents were obtained from the Municipal Center for Disease Control and Prevention in each city. The total mortality of natural causes was coded according to the International Classification of Diseases with codes A00–R99 in the 10th revision and 001–799 in the 9th revision. The air pollution data were collected from the National Air Pollution Monitoring System that is under the China National Quality Control. In each city, there were from 2 to 13 monitoring stations (Table 1). The daily (24-hour) average concentrations of PM_{10} were measured by using the method of tapered element oscillating microbalance (China State Environmental Protection Agency, 2000). The location of monitoring stations is mandated not to be in the direct vicinity of traffic or of industrial sources and not to be influenced by local pollution sources, thus also avoiding buildings or housing and large emitters such as coal-, waste-, or oil-burning boilers; furnaces; and incinerators. In each city, the daily air pollutants' concentrations were averaged from the available monitoring results across various stations. Missing data were not included in the statistical analysis.

To allow adjustment for the effect of weather conditions on mortality, daily (24-hour) mean temperature and relative humidity were obtained from the Meteorological Bureau in each city.

2.2. Statistical analysis

We applied 2-stage Bayesian hierarchical statistical models to estimate the national and regional average associations of PM_{10} with daily mortality (Bell et al., 2008). In the first stage, we obtained the city-specific estimates after adjusting for time-varying confounders by fitting a "main" model, "seasonal" model and a "harmonic" model, respectively. The second stage combined the city-specific estimates to generate the national and regional average estimates, accounting for their statistical uncertainty.

The "main" model assumes that the acute effect of PM_{10} on daily mortality is constant throughout the year and therefore contains no adjustment for season. In the "main" model, we used generalized additive models with natural spline (ns) smoothers to control for long-term and seasonal trends of daily mortality and weather conditions. In addition, *quasi*-Poisson regression, instead of Poisson regression, was used to allow for possible overdispersion. The "main" model is summarized as

$$\text{Log}E(Y_t)=\beta Z_{t-l}+ns(t,df)+ns(temp_t,6)+ns(humi_t,3)+DOW_t \quad (1)$$

where Y_t is the daily count of total mortality; β is the regression coefficient linking PM_{10} to daily mortality and it assumes the time-invariant effect of PM_{10}; Z_{t-l} is PM_{10} on day t at a lag of l day; $ns(t, df)$ is the natural spline smoothing function of calendar time with 7 degrees of freedom per year; $ns(temp_t, 6)$ is the natural spline of temperature on day t with 6 degrees of freedom for the whole period; $ns(humi_t, 3)$ is the natural spline of relative humidity on day t with 3 degrees of freedom for the whole period; DOW_t is the indicator of day of the week on day t. We only controlled the concurrent day's temperature and humidity because there was great collinearity of weather conditions among neighboring days.

The "seasonal" model allows the season-specific effect estimates by replacing the β with a pollutant×season interaction term:

$$\beta=\beta_w I_w+\beta_{sp} I_{sp}+\beta_{su} I_{su}+\beta_f I_f \quad (2)$$

where I_w, I_{sp}, I_{su} and I_f are indicators of winter, spring, summer and fall, respectively; β_w, β_{sp}, β_{su}, β_f are the regression coefficients in relation to PM_{10} effect on daily mortality in

winter, spring, summer and fall, respectively. Seasons were defined as the following 3-month periods: winter as December to February, spring as March to May, summer as June to August and fall as September to November.

In addition, we also allowed the temporal trend to differ by season, replacing the time trend term in Eq. (1), *ns*(*t*, *df*), with

$$ns(t, df) = ns(t, df)I_w + ns(t, df)I_{sp} + ns(t, df)I_{su} + ns(t, df)I_f. \quad (3)$$

In sum, the "seasonal" model inherits from the "main" model except for the above two replacements. It estimates the season-specified average effect of PM_{10} on daily mortality, acknowledging that the effect may change over time.

The "harmonic" model allows the effect of PM_{10} vary smoothly throughout the year by constructing a yearly periodic function for estimating seasonal patterns. The "harmonic" model is analogous to Eq. (1), replacing a sine/cosine model of the following form for β to estimate the time-varying effect of PM_{10}. No season indicators are in-corporated in the "harmonic" model.

$$\beta = \beta_0 + \beta_1 sin(2\pi t/365)/c_1 + \beta_2 cos(2\pi t/365)/c_2 + \beta_3 sin(2\pi t*2/365)/c_3 + \beta_4 cos(2\pi t*2/365)/c_4 \quad (4)$$

where c1 to c4 are known orthogonalizing constants, and β_0 to β_4 are estimated for each city.

Because previous CAPES publications have reported the average of the current-day and previous-day (lag 01) concentration of PM_{10} could generate the strongest effect estimate among single-day lag of 0 to 7 and multi-day average lag of 0–1, 0–4 and 0–7, we introduced pollution term of lag 01 in main analyses (Chen et al., 2012).

In the second-stage analysis, we used a hierarchical Bayesian model to pool the city-specific effect estimates obtained from the first-stage models (Peng et al., 2009; Samet et al., 2000). For a particular model, we have a city-specific maximum likelihood estimate $\hat{\beta}$, which is a scalar for the "main" model, a vector of length 4 for the "seasonal" model and a vector of length 5 for the "harmonic" model. $\hat{\beta}$ is assumed to be normally distributed around the true city-specific β with covariance matrix *V*, estimated within each city. We applied this hierarchical model by using 2-level normal independent sampling estimation with uniform priors as in Eq. (5), which could be implemented in TLNise package of R software (Everson and Morris, 2000). This procedure provides a sample from the posterior distribution of Σ, from which we can calculate posterior means and variances of the overall and city-specific PM_{10} effects.

$$\begin{aligned} \hat{\beta} | \beta &\sim N(\beta, V) \\ \hat{\beta} | \mu, \textstyle\sum &\sim N(\mu, \textstyle\sum), \end{aligned} \quad (5)$$

where Σ is the covariance matrix describing the between-city variation of β, and the diagonal elements of Σ measure the heterogeneity across cities; μ is the overall mean for the cities.

To characterize regional differences in seasonal patterns, we fitted all 3 models (main, seasonal, and harmonic) separately within each geographic region and pooled the estimates in the second-stage models. All the results were presented as the posterior mean of percentage increase in daily mortality and its 95% posterior intervals (PIs) associated with a 10 μg/m^3 increment in PM_{10}.

Difference in the range of daily variations in PM_{10} concentrations across seasons may affect the comparison of season-specific effect estimates, so we calculated excess log-relative mortality rate per an interquartile range (IQR) increase of PM_{10}. We also explored the sensitivity of main findings to adjustment of co-pollutants and the smoothness of time trends. Specifically, current-day sulfur dioxide (SO_2) or nitrogen dioxide (NO_2) concentrations were included alternatively in the "main" and "seasonal" models. Smoothness of time is an important component in the time-series models, so we performed the "harmonic" model with 4, 7 and 10 *df* per year of data separately so as to see how the seasonal pattern varied by different magnitude of smoothness of time. At last, we controlled the temperature in the models using the moving average of current day and previous 3 days (lag 0–3), the moving average of current day and previous 7 days (lag 0–7) and the moving average of current day and previous 14 days (lag 0–14), because previous studies have shown that the acute effects of temperature on daily mortality might be delayed and lasted for several days (Guo et al., 2011).

All of the above models were fitted using *mgcv* package and *dlnm* package in the R statistical software (R Development Core Team, 2011). The statistical tests were two-sided, and $P<0.05$ was considered as significant.

3. Results

Table 1 summarizes the population, mortality, PM_{10}, and temperature data in the 17 Chinese cities. The size of the permanent population in each city varies from 1.2 to 12.3 million. The research period for each city varies from two to seven years after 2000 (apart from Hong Kong with times series data starting from 1996). The daily mean deaths of all nonaccidental causes vary according to the size of the city, ranging from 11 to 119. The averaged daily concentrations of PM_{10} in the Chinese cities range from 52 $\mu g/m^3$ to 156 $\mu g/m^3$, much higher than those reported in developed countries (Samoli et al., 2008). The missing data is very scarce (less than 0.5% our data). Daily PM_{10} levels and mortality rates are shown to vary considerably across seasons. Generally, northern cities have their highest mean levels of PM_{10} than middle and southern cities. PM_{10} concentrations in winter/spring months tend to be higher than in summer/fall months. IQRs of PM_{10} concentrations in summer days are much smaller than winter. Daily deaths generally present a peak in winter and trough in summer (data not shown). The weather data was complete for every city. The averaged temperature ranged from 7 °C to 24 °C, reflecting distinct climatic characteristics among these cities.

We used the "main" model to estimate the whole-year average PM_{10} effects and "seasonal" model to obtain the season-specific estimates (Table 2). Across all seasons, a 10 $\mu g/m^3$ increase of PM_{10} is associated with 0.35% (95%PI: 0.13% to 0.56%) increase of mortality on a national average basis. This estimate is similar to a prior publication about CAPES project (Chen et al., 2012). At the national level, summer and winter show the largest effects, and there are no significant effects in spring and fall. A 10 $\mu g/m^3$ increase in the two-day moving average concentrations (lag 01) of PM_{10} is associated with 0.45% (95%PI: 0.15% to 0.76%), 0.17% (95%PI: −0.09% to 0.43%), 0.55% (95%PI: 0.15% to 0.96%) and 0.25% (95%PI: −0.05% to 0.56%) increases in daily total mortality for winter, spring, summer and fall, respectively. PM_{10} presents the largest effects in southern cities, followed by middle cities. The PM_{10} effects are lowest and non-significant in northern cities.

We explored differences of seasonal pattern in regional average estimates by including a region indicator variable in the second stage of hierarchical models (see Table 2). Northern cities exhibit no significant effects of PM_{10} in all the seasons, with the largest estimates in winter and summer. For the middle cities, there are significant effects for winter, summer

and fall, but summer estimates a 2 times higher effect than winter and fall. In the south, winter and summer show similar and higher estimates than fall, with no significant effects present in spring.

Fig. 2 shows the seasonal plots that allow the effect estimates corresponding to a 10 $\mu g/m^3$ increase of PM_{10} vary smoothly in one year using the "harmonic" model. The seasonal pattern of PM_{10} effects is characterized by two peaks in winter and summer. The two-peak pattern is clear and generally consistent across the northern, middle and southern region of China. The general consistency of findings from the "seasonal" model and the "harmonic" model might indicate that the seasonal patterns identified by the "seasonal" model are not an artifact of the choice of seasonal division, because the "harmonic" model just allows smooth variation in effect estimates throughout the year.

In order to account for temporal variation range of PM_{10} levels in different seasons, we also present the effect estimates associated with an IQR increase of PM_{10} (Table 2). At the national average levels, there is still a two-peak pattern for the season-specified estimates, although the effect is higher in winter than summer. In the middle, summer still has the greatest effect, but in the north and south, the effect magnitude in winter is much higher than summer, demonstrating the big difference of IQRs in different seasons.

Table 3 presents the result of 2-pollutant analysis using the "main" model and "seasonal" model. The effect magnitude increases slightly when SO_2 is simultaneously controlled and attenuates appreciably when adjusting for NO_2. Nevertheless, the seasonal pattern of PM_{10} effects changed little after adjustment for gaseous pollutants.

Fig. 3 shows the sensitivity of national average estimate of the smooth seasonal effects of a 10 $\mu g/m^3$ increase of PM_{10} to the *df*/year assigned for the smooth function of time. The curve using 4 *df* per year diverges from those with more aggressive smoothness (7 and 10 *df* per year), possibly because of a lack of adjustment in the models. Increasing the number of *df* from 7/year used in our main analyses to 10/year does not lead to a significant change of the seasonal estimates. The seasonal pattern of PM10 effects still remained with peaks in winter and summer when we controlled the confounding effects of temperature on more lag days (please see S-Table 1 in the supplemental materials).

As a supplementary analysis, we further examined the seasonal pattern of the short-term effects of PM_{10} on cardiovascular and respiratory mortality. The season-specified estimates of SO_2 and NO_2 on total, cardiovascular and respiratory mortality were also provided in the online supplement (please see S-Tables 2–4). These results indicated that: 1) the seasonal pattern of PM_{10}'s health effect was consistent for both total mortality and cardiorespiratory mortality; and 2) the seasonal pattern for SO_2 and NO_2 was generally similar with PM_{10}.

4. Discussion

In 17 Chinese cities, we observed a two-peak pattern in the acute effects of particulate air pollution with the highest in winter and summer. The seasonal variations were generally consistent in northern, middle and southern Chinese cities. This seasonal pattern remained stable after adjustment of co-pollutants and using alternative model specifications. To our knowledge, this is the first multicity study in developing countries to analyze the seasonal variations of PM-related health effects.

This study suggests evidence of seasonality but differs from previous studies with a clear two-peak pattern occurring in winter and summer. Previous studies have reported very inconsistent patterns of seasonality, with a single peak in winter, summer or transitional seasons (spring and fall) (Peng et al., 2005; Qian et al., 2010; Zeka et al., 2006). An early

analysis of particulate air pollution and mortality carried out in Steubenville, Ohio, Philadelphia, Pennsylvania, and Cook County, Illinois, indicated that the effects are strongly modified by season with the highest estimates in summer (Moolgavkar and Luebeck, 1996). A large time-series analysis according to the database of the National Morbidity and Mortality Air Pollution Study found a strong seasonal variation especially in the Northeast US with a peak in summer (Peng et al., 2005). The Air Pollution and Health: A European Approach project indicated higher mortality effects in warm seasons (Katsouyanni et al., 1997). A multicity case-crossover study in 9 Italian cities identified higher mortality effects of PM_{10} in summer (Stafoggia et al., 2008). In contrast, a seasonal analysis of short-term effects of fine particles on hospital admissions in 202 US counties revealed the highest estimates in winter on the hospital admissions (Bell et al., 2008). Several Asian single-city studies in Hong Kong (Wong et al., 2001), Shanghai (Kan et al., 2008), Wuhan (Qian et al., 2010), and Taichung (Liang et al., 2009) also estimated the largest effects in winter or cool months, but analyses in Shenyang (Ma et al., 2011), Seoul (Yi et al., 2010) and Bangkok (Wong et al., 2008) indicated higher effects in summer or warm months. Furthermore, other few researchers have found higher effects in transitional seasons (spring and fall) (Levy et al., 2001; Zanobetti and Schwartz, 2009; Zeka et al., 2006). The above inconsistent findings may be related with a number of factors such as PM components and levels (including peak concentrations and degree of variation), gaseous pollutants, climate conditions, exposure pattern of local residents, socioeconomic characteristics and analytical methods used.

Researchers generally reported their estimates of acute health effects associated with a 10 $\mu g/m^3$ increment in pollutant concentrations, but the variation range of the daily pollutant measurements may generate additional sources of heterogeneity when comparing these estimates across seasons. For example, the observed higher PM_{10} effects per 10 $\mu g/m^3$ increase in summer might be simply due to a relative lack of temporal variation of concentrations in this season. This might be an important source of uncertainty when comparing the effect estimates where the IQRs were different. Nonetheless, our estimates were generally not sensitive to the replacement of a 10 $\mu g/m^3$ increase with an IQR scale. Other mechanisms might be the underlying causes that were responsible for the two-peak seasonal pattern.

First, the PM sources and constituents may vary by season, with the most toxic particles having a winter/summer maximum (Bell et al., 2008; Peng et al., 2005). The estimated strongest effects may be an indicator of most toxic component of PM. Coal has long been the major energy source of China, and coal combustion is especially widespread in winter for heating and summer for cooling from coal-fired boilers and power plants. This was supported by considerably higher production of electricity in winter and summer than that in transitional seasons (see S-Fig. 1 in the supplementary material). The contents of metals, water-soluble inorganic ions and polycyclic aromatic hydrocarbons are shown to be in the highest level and fluctuate greatly in winter in some Chinese cities (Du, 2007; Huang et al., 2012; Tan et al., 2006). It is acknowledged that combustion-related particles have been associated with increased risk of mortality (Huang et al., 2012). On the other hand, sand storms, mainly originating in the deserts and deteriorated grasslands in northwest of China, can be transported eastward and southward, constituting one of the major sources of PM in spring in these cities (Wang et al., 2004). The crustal elements of wind-blown dust have been shown to have weaker hazards to health than combustion particles (Laden et al., 2000). This renders some support to our findings of the lowest and non-significant mortality effects in spring. Also, PM in the north is more affected by sand storms than in the south, which might contribute to a weaker health effect of PM_{10} in the north. A detailed analysis of the regional and seasonal variation in PM constituents warrants further investigations.

Second, the pattern of exposure to ambient PM in populations may also change from season to season. It is expected that the exposure measurement error would be smaller in warm months because of more outdoor activities and higher penetrations of outdoor particles into indoor environment due to increased natural ventilation (Peng et al., 2005; Sarnat et al., 2000). Increased use of air conditioning in summer would not decrease substantially the indoor–outdoor air exchange rate because air conditioners were not widely available, and even if available but not frequently used for economic reasons in Chinese households during the study period of 1996–2008.

Third, both high temperature in summer and low temperature in winter were shown to enhance the health effects of airborne PM through complex mechanism (Cheng and Kan, 2012; Qian et al., 2008, 2010; Roberts, 2004). It is known that high summer temperature, combined with strong sunlight, can improve the photochemical reaction and generate secondary particles.

Fourth, the higher relative effects observed in summer might be due to lower background mortality in summer, thus resulting in a larger pool of susceptible people in this season (Nawrot et al., 2007).

Investigating the temporal variations of the short-term health effects of PM is important to generate or validate specific hypotheses about the toxicity of PM components (Peng et al., 2005). Furthermore, better knowledge of potential modifying effect of season will help in public health policy determinations, risk assessment of air pollution, and setting air quality standards. Exploration of the seasonality of the short-term associations with PM_{10} exposure and mortality is hampered by the inherentvariability of the resulting effect estimates, especially when the data is further stratified into seasons (Peng et al., 2005). The 17 cities in present analysis are densely populated with 1.2 to 12.3 million of permanent residents in each city, and are generally with much higher levels of particulate air pollution compared with those in North America and Western Europe. Therefore, our findings, possibly more precise and stable, contributed to the scientific literature on seasonal variations of air pollution health effects for high exposure settings typical in developing countries.

However, some limitations should be addressed. First, as in most previous time-series studies, we simply averaged the monitoring results across various stations as the proxy for population true exposure level to PM. The resulting measurement error could produce complicated impact on the estimated effects of PM, which is difficult to quantify its magnitude and direction especially in bi-pollutant models with SO_2 and NO_2 (Zeka and Schwartz, 2004). Second, despite the season-specified PM_{10} effects that were not sensitive to co-adjustment of gaseous pollutants, it is still not easy to eliminate their confounding roles when deriving our findings, because of the high correlations between particles and gases. Third, we did not have data on several confounders that might affect the associations between PM_{10} and daily mortality. Influenza may modify the acute health effects of PM (Wong et al., 2009); however, we were not able to collect the influenza data in these Chinese cities. Fourth, ozone has not been regularly monitored in most Chinese cities, though some previous studies have indicated that ozone has acute effect on mortality independent of PM (Yang et al., 2012; Zhang et al., 2006). However, previous studies have shown that the correlation between ozone and particulate mass or its components was weak (Anderson et al., 2012), and the PM-morality associations might not be confounded by ozone (Kan et al., 2008). Fifth, the measurements of PM_{10} in our study were not corrected for volatilization (China State Environmental Protection Agency, 2000), which may have impact on our season findings. Finally, the lengths of study period varied in different cities; therefore, the statistical power might not be comparable for each city. At last, the different research

periods for each city might also add uncertainty when pooling the season-specified effect estimates (Dominici et al., 2007).

In summary, our analyses found that the acute effects of particulate air pollution and mortality could vary by season with two peaks in winter and summer in 17 big Chinese cities. To our knowledge, this study is the first to identify a “two-peak” pattern with considerably stronger effects of PM_{10} in winter and summer than in the transitional seasons (spring and fall). These findings provide new information regarding the seasonal pattern of air pollution health effects in developing countries and may have implications for local environmental and public health policies.

Supplementary Material

Refer to Web version on PubMed Central for supplementary material.

Acknowledgments

The study was supported by the National Basic Research Program (973 program) of China (2011CB503802), National Natural Science Foundation of China (30800892), Gong-Yi Program of China Ministry of Environmental Protection (201209008), Shanghai Municipal Committee of Science and Technology (12dz1202602), the Short-term International Visit Program for Doctoral Students of Fudan University, and National Institute of Environmental Health Sciences (R01ES019560).

We thanked all members of the CAPES collaborative group. The content is solely the responsibility of the authors and does not necessarily represent the official views of all the entities that funded our work. The authors declare that they have no competing financial interests.

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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.scitotenv.2013.02.040.

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HIGHLIGHTS

- Evidence concerning the seasonality of PM health effects is inconsistent.
- We observed a two-peak (winter and summer) seasonal pattern in 17 Chinese cities.
- This is the first multicity study in developing countries to analyze the seasonality of PM health impact.

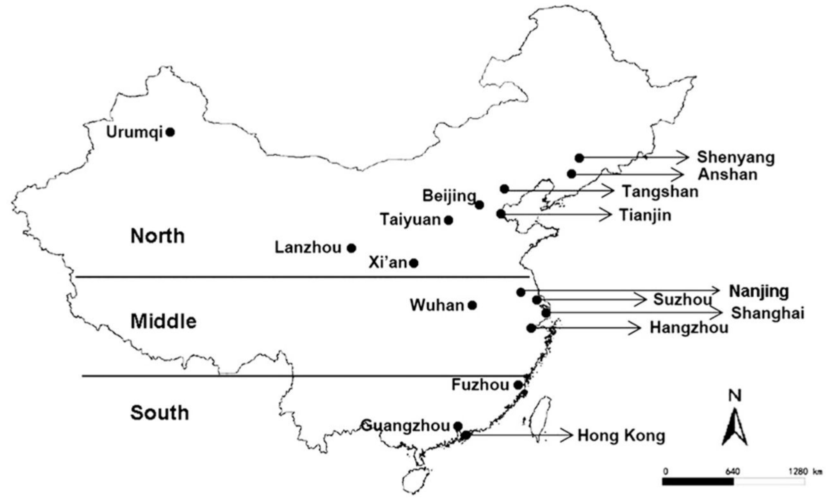

Fig. 1.
Location of the cities within the China Air Pollution and Health Effects Study (CAPES).

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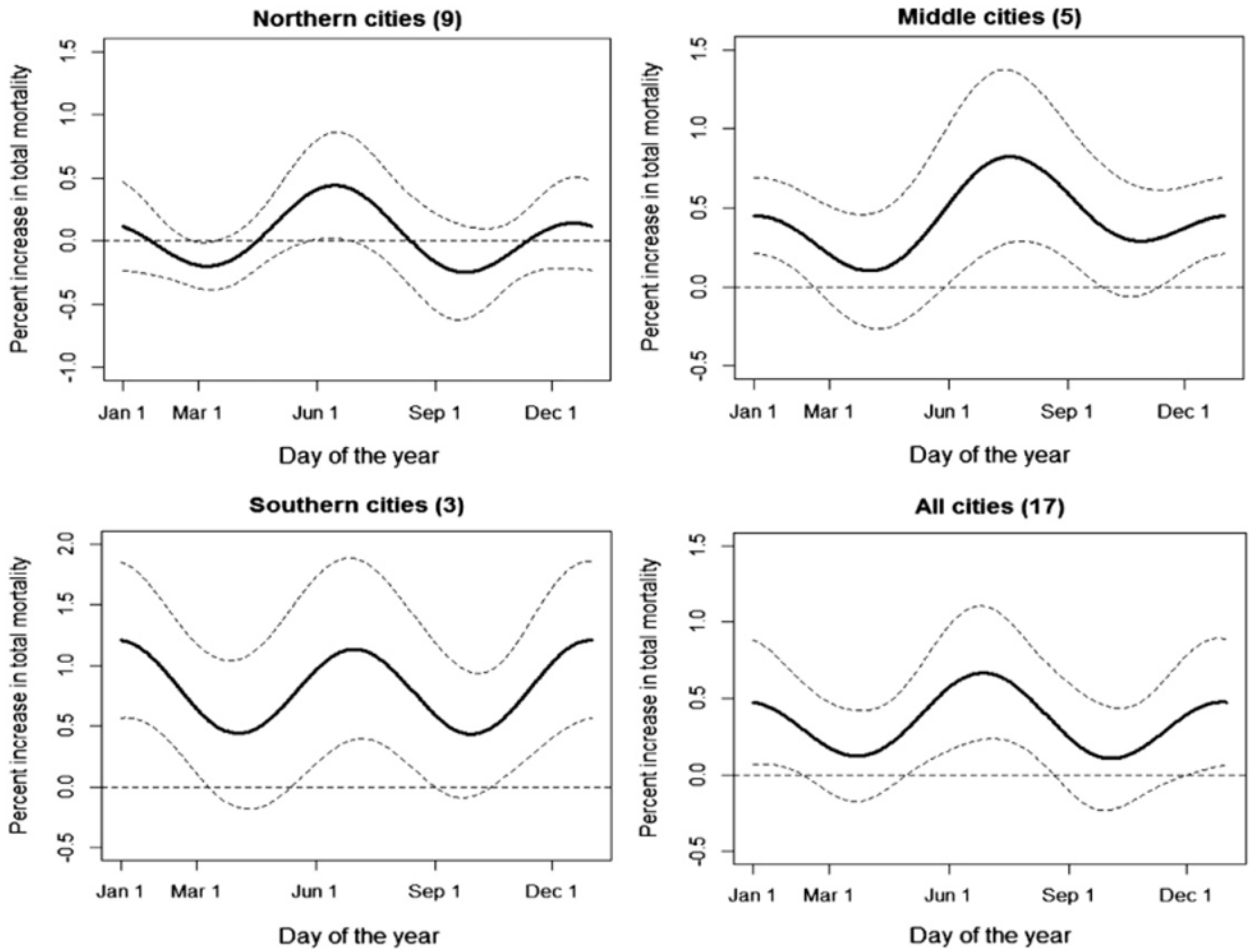

Fig. 2.
National and regional smooth seasonal effects of a 10 μg/m^3 increase of particulate matter less than 10 μm in aerodynamic diameter (PM_{10}) at lag 01 day in the in the China Air Pollution and Health Effects Study (CAPES). Dotted lines represent 95% posterior intervals of the effect estimates.

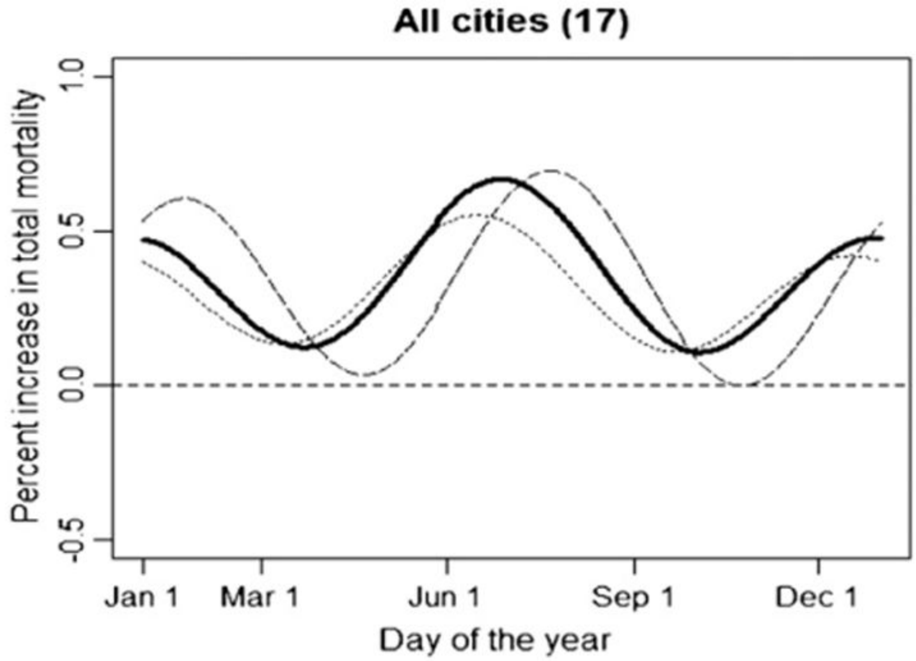

Fig. 3.
Sensitivity of national average estimate of the smooth seasonal effects of a 10 μg/m^3 increase of particulate matter less than 10 μm in aerodynamic diameter (PM_{10}) at a lag 01 day to the degrees of freedom assigned for the smooth function of time in the China Air Pollution and Health Effects Study (CAPES). The degrees of freedom selected were 4 *df* (dashed line), 7 *df* (solid line) and 10 *df* (dotted line).

Table 1

Descriptive data on the study period, exposure (PM_{10}, μg/m^3), outcome (daily total deaths) and temperature in the CAPES cities.

Region	City	Study period	Population (million)	Daily average deaths	PM_{10} (mean and IQR, μg/m^3)					No. of monitors	Temperature (°C)[a]
					All-year	Winter	Spring	Summer	Fall		
North	Anshan	2001–2004	2.4	28	111 (70)	112 (78)	112 (72)	93 (59)	112 (66)	2	11
	Beijing	2007–2008	12.3	118	139 (98)	134 (103)	148 (100)	125 (67)	111 (83)	12	15
	Lanzhou	2004–2008	1.9	19	156 (98)	206 (138)	144 (95)	104 (45)	125 (64)	5	7
	Shenyang	2005–2008	6.4	67	114 (51)	127 (58)	113 (47)	100 (41)	103 (49)	2	8
	Taiyuan	2004–2008	2.6	24	132 (69)	137 (90)	142 (73)	113 (50)	127 (65)	9	11
	Tangshan	2006–2008	1.9	19	98 (61)	116 (78)	99 (43)	80 (29)	91 (53)	6	13
	Tianjin	2005–2008	1.2	11	101 (61)	106 (82)	105 (57)	83 (41)	99 (66)	13	13
	Urumqi	2006–2007	2.3	17	144 (115)	269 (194)	109 (68)	56 (25)	155 (62)	3	9
	Xi'an	2004–2008	3.4	26	132 (56)	163 (72)	128 (43)	105 (44)	124 (60)	7	13
Middle	Suzhou	2005–2008	4.1	34	90 (57)	90 (60)	100 (58)	73 (42)	89 (58)	8	17
	Nanjing	2007–2010	5.4	40	101 (61)	107 (67)	108 (59)	85 (48)	105 (64)	9	17
	Shanghai	2001–2004	8.5	119	102 (72)	113 (96)	107 (67)	81 (53)	94 (72)	9	18
	Wuhan	2003–2005	4.5	58	130 (77)	151 (83)	129 (58)	96 (51)	138 (78)	10	18
	Hangzhou	2002–2004	2.5	20	121 (65)	136 (87)	120 (58)	98 (54)	124 (54)	10	18
South	Guangzhou	2007–2008	6.5	79	74 (47)	94 (76)	74 (47)	50 (26)	76 (38)	9	23
	Fuzhou	2004–2006	1.8	16	72 (46)	73 (45)	89 (58)	56 (33)	72 (44)	4	21
	Hong Kong	1996–2002	6.7	84	52 (35)	65 (72)	50 (25)	33 (12)	58 (31)	7	24

Abbreviations: PM_{10}, particulate matter less than 10 μm in aerodynamic diameter; CAPES, China Air Pollution and Health Effects Study; IQR, interquartile range.

[a] Annual average temperature.

Table 2

Region-and season-specified percentage change of daily mortality associated with a 10 μg/m^3 increase and an interquartile range increase of PM_{10} at lag 01 day in the CAPES cities.

	Main model		Seasonal model							
	Yearly		Winter		Spring		Summer		Fall	
	Mean	95% PI	Mean	95% PI	Mean	95% PI	Mean	95% PI	Mean	95% PI
National (17)										
Per 10 μg/m^3	0.35	0.13, 0.56	0.45	0.15, 0.76	0.17	−0.09, 0.43	0.55	0.15, 0.96	0.25	−0.05, 0.56
Per IQR	1.91	0.82, 3.00	3.60	1.32, 5.87	0.87	−0.55, 2.29	2.04	0.43, 3.64	1.35	−0.38, 3.08
North (9)										
Per 10 μg/m^3	0.13	−0.17, 0.43	0.18	−0.40, 0.76	−0.02	−0.41, 0.37	0.19	−0.25, 0.62	−0.09	−0.56, 0.37
Per IQR	0.80	−1.09, 2.68	2.21	−2.2, 7.14	0.18	−2.94, 3.29	0.34	−2.03, 2.72	−0.07	−4.00, 3.85
Middle (5)										
Per 10 μg/m^3	0.39	0.18, 0.60	0.39	0.14, 0.64	0.07	−0.26, 0.41	0.90	0.11, 1.69	0.40	0.13, 0.68
Per IQR	2.62	1.08, 4.16	3.16	1.42, 4.91	0.46	−1.52, 2.43	4.45	0.45, 8.45	2.67	0.80, 4.55
South (3)										
Per 10 μg/m^3	0.90	0.53, 1.27	1.17	0.76, 1.59	0.63	−0.03, 1.30	1.12	0.44, 1.80	0.69	0.04, 1.35
Per IQR	3.90	1.75, 6.05	9.02	7.25, 10.80	2.82	−1.74, 7.39	1.29	0.83, 1.75	1.52	0.70, 2.33

Abbreviations: PM_{10}, particulate matter less than 10 μm in aerodynamic diameter; CAPES, China Air Pollution and Health Effects Study; IQR, interquartile range; PI, posterior intervals.

Table 3

National season-specified percentage change of daily mortality associated with a 10 μg/m^3 increase and an interquartile range increase of PM_{10} at lag 01 day in bi-pollutant models in the China Air Pollution and Health Effects Study (CAPES).

	Main model		Seasonal model							
	Yearly		Winter		Spring		Summer		Fall	
	Mean	95% PI	Mean	95% PI	Mean	95% PI	Mean	95% PI	Mean	95% PI
PM_{10} only										
Per 10 μg/m^3	0.35	0.13, 0.56	0.45	0.15, 0.76	0.17	−0.09, 0.43	0.55	0.15, 0.96	0.25	−0.05, 0.56
Per IQR	1.91	0.82, 3.00	3.60	1.32, 5.87	0.87	−0.55, 2.29	2.04	0.43, 3.64	1.35	−0.38, 3.08
With SO_2										
Per 10 μg/m^3	0.37	0.12, 0.63	0.50	0.15, 0.84	0.22	−0.08, 0.52	0.56	0.13, 0.99	0.27	−0.04, 0.59
Per IQR	2.17	0.75, 3.59	3.94	1.35, 6.53	1.16	−0.53, 2.86	2.19	0.44, 3.94	1.49	−0.40, 3.40
With NO_2										
Per 10 μg/m^3	0.27	0.05, 0.49	0.36	0.07, 0.66	0.13	−0.13, 0.39	0.44	0.03, 0.84	0.15	−0.18, 0.49
Per IQR	1.56	0.25, 2.86	3.18	0.68, 5.67	0.76	−0.84, 2.35	1.90	0.15, 3.65	0.82	−1.19, 2.84

Abbreviations: PM_{10}, particulate matter less than 10 μm in aerodynamic diameter; CAPES, China Air Pollution and Health Effects Study; IQR, interquartile range; SO_2, sulfur dioxide; NO_2, nitrogen dioxide; PI, posterior intervals.