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Ambient Air Pollution and Health Effects in Shanghai

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AMBIENT AIR POLLUTION
AND HEALTH EFFECTS
IN SHANGHAI
Trend, Challenges, and Opportunities
Wei Tu, Zhijing Lin, Lili Du, Haidong Kan
and Weichun Ma

Introduction
Previous studies have provided convincing evidence of the association between exposures to air pollutants and human morbidity and mortality (WHO and Global Environment Monitoring System 1992; Folinsbee 1993; Wagner 1994; Schwela 2000; Samet and White 2004; Ren and Tong 2008; Gurjar et al. 2010; Kingham 2011; WHO 2013c; Landrigan 2017; Kinney 2018). Over the last few decades, great efforts have been made to control industrial air pollution in developing countries, but, according to the standards outlined in the World Health Organization Air Quality Guidelines (WHOAQG), many remain highly polluted, particularly in densely populated urban areas (WHO 2017). It was estimated that air pollution had caused 1.2 million premature deaths in China in 2010 alone (Lim et al. 2013) and that total economic loss had been between 1% and 7% of the nation’s total GDP (Zhang, P. et al. 2011).

Since the late 1970s, China’s largest megacity, Shanghai, has been rapidly transforming from a heavily polluted industrial city to a rapidly rising global city. During this period, Shanghai has significantly reduced industrial pollution and lowered its energy and emission intensity (Tu and Shi 2006). Shanghai has also implemented the nation’s most stringent policy to limit the ownership of private vehicles. As a result, both the total emissions and the concentration of air pollutants in Shanghai showed overall downward trends between the mid-1990s and mid-2000s (Chan and Yao 2008).

However, total energy consumption has been rapidly increasing, with fossil fuel remaining the major energy source. Motor vehicles, particularly privately owned vehicles, grew at a much faster pace than the population. As the primary source of air pollution shifted gradually from conventional coal combustion to a mixture of coal combustion and motor vehicle emissions (Kan et al. 2012; Yang, G.Y. et al. 2012), vehicular emissions have become the leading source of air pollution. With this context of increasing vehicular emissions, particulate matter (PM) has become the primary air pollutant in the region. A high level of ambient PM causes more frequent haze events of high pollution and low visibility (Wang, Y. et al. 2014; Leng et al. 2016). These changes have made outdoor air pollution the most challenging environmental problem in Shanghai.

Past studies have reported a significant association between short-term exposure to air pollution and mortality and morbidity in Shanghai (Kan 2009; Chen, R. et al. 2012), although the level
of impacts on health outcomes varied across different air pollutants as well as population cohorts (Lai et al. 2013). In addition, the increased health risks observed among the Chinese population were generally lower in magnitude, per amount of increased pollution, than those reported in the developed countries. However, the health risks to the Chinese population may be higher considering the levels of air pollution and the large size and rapidly aging population, particularly in big cities such as Shanghai (Kan et al. 2012). One study estimated that PM$_{2.5}$ pollution in Shanghai, without control measures, would result in approximately 2.26% and 3.14%, GDP values and welfare losses, respectively, in 2030 (Wu, R. et al. 2017).

Despite the growing interest, there are several gaps in the current literature on air pollution and health effects in Shanghai. First, air pollution and health effects are usually studied separately. Rarely are these two problems examined together. Second, there is an absence of research on current trends and conditions in Shanghai’s ambient air pollution. Third, few studies have provided a comprehensive summary from the findings from individual epidemiological studies. To bridge these gaps, this chapter has the following objectives: 1) to present the trend and current state of the total emissions and concentrations of criteria ambient air pollutants, including PM, NO$_2$, SO$_2$, CO, and O$_3$; 2) to summarize the major epidemiological evidence of the short-term human health effects of exposure to ambient air pollution; 3) to analyze the main challenges in air pollution and health effects studies; and 4) to suggest areas of future research. Thus the chapter next provides a brief outline of Shanghai; it then summarizes the trend as well as the level of ambient air pollution; it goes on to review the major health effects of short-term exposure to air pollution; and it finishes by discussing the current challenges and suggesting directions for future research.

**Shanghai**

Shanghai is located at the center of China’s coastline facing the Pacific Ocean and at the mouth of the Yangtze River. Jiangsu and Zhejiang are the two neighboring provinces west of the city. The Shanghai Municipal Government (SMG) currently administers 18 districts and one county, which cover a land area of 6,341 km$^2$ (2,448 mi$^2$) (see Figure 32.1).

The subtropical monsoon brings Shanghai sufficient sunshine, abundant rainfall, and distinct seasons. Shanghai receives on average 1,255 mm (over 49 inches) of precipitation from its 112 rainy days annually. The frequent rain events help clean the pollution from the air. However, a combination of high temperature and more than 75% monthly average relative humidity (RH) (Xu et al. 1997; Yao et al. 2002; Zhang, Y.X. et al. 2006) creates favorable conditions for the generation of secondary air pollutants such as SO$_4^{2-}$ and NO$_3^-$ (He et al. 2001; Song, Y. et al. 2002). In addition, while the dominating northwest to northeast wind in winter helps transport air pollutants from inland regions to the city, the prevailing southeastern to southwestern wind in summer brings clean air from above the East China Sea (Feng, J. et al. 2006; Wang, Y. et al. 2006).

Shanghai has been undergoing unprecedented growth and change since the early 1990s (Wu, F. 2000; Marton and Wu 2006; Wei et al. 2006). Between 1990 and 2015, the total GDP rose from US$16.34 billion to US$403.36 billion, and the per capita GDP increased from US$1,445 to US$16,665. Between these years, the total population increased from 13.34 million to 24.15 million. Although the permanent residents grew by only 12%, the migrant (floating) residents exploded from 0.51 million to 9.72 million (SMSB 1990–2015) (Figure 32.2). Moreover, the urbanized area has expanded into the city’s traditional agricultural hinterland. One study estimated that the urbanized area had grown from about 304 km$^2$ (117 mi$^2$) in 1984 to 1,302 km$^2$ (503 mi$^2$) in 2014, or roughly 50 km$^2$ (19 mi$^2$) per year (Zhao, S. et al. 2016). Moderate haze of air pollution can be seen clearly from several photos taken in spring 2018 (Figures 32.3 to 32.5).
Figure 32.1 Location map of Shanghai and the location of air quality monitoring stations.
Source: Chinese Geographical Information Monitoring Cloud Platform.
Figure 32.2  Population (consists of registered residents and floating residents) and GDP in Shanghai, 1990–2015.

Source: Data from SMSB (1990–2015).

The automobile fleet has also grown rapidly. Between 2001 and 2015, the total number of motor vehicles grew from 1.19 million to 3.32 million and the number of private motor vehicles increased from 0.67 million to 2.51 million. Paved roads and highways have also been substantially extended. Between 1990 and 2015, the paved road network increased from 1,663 km (1,033 mi) to 18,184 km (11,299 mi), while highways were expanded from 36 km (22 mi) to 825 km (513 mi) (SMSB 1990–2015). This expanding investment in automobiles, trucks, and complementary infrastructure has made vehicle emissions a primary source of air pollution in Shanghai (Wang, H. et al. 2010; Ling 2014).

Since the late 1970s, Shanghai has implemented a series of policy measures to reduce industrial pollution and improve energy efficiency. The important initiatives include economic restructuring, increasing clean energy reliance, relocating and phasing out heavily polluted industries, promoting green technologies, and limiting private vehicle ownership. Between 1990 and 2015, the proportion of the tertiary sectors (services) increased from 30.9% to 67.8% in Shanghai’s economy, while the proportion of the secondary sectors (manufacturing) decreased from 64.7% to 31.8% (SMSB 1990–2015). Heavily polluted factories were either shut down or relocated to suburban industrial development zones, away from major residential areas. There were also initiatives introduced to improve pollution treatment (Tu and Shi 2006).
Energy efficiency in Shanghai has also improved dramatically. Between 1990 and 2015, total GDP grew more than 23 times, while total energy consumption increased only 2.56 times, from 31.91 million tons of coal equivalent (TCE) to 113.87 million TCE. During the same period of time, the share of coal in total consumption decreased from 43.1% to 32.3%, and the share of natural gas increased from 5.2% to 8.6% (SMSB 1990–2015). In addition, new environmental technologies such as desulfurization, de-nitration, dust collecting, selective catalytic/non-catalytic reduction (SCR/SNCR) systems, and high-energy-efficiency boilers and kilns were widely installed to reduce emissions and improve energy efficiency. Furthermore, since the mid-2000s, the SMG has gradually adopted more stringent emission standards (Jin et al. 2016).

Private vehicle ownership has also been a focus of emission reduction policy. In 1994, the SMG introduced a quota policy to limit the ownership of private motor vehicles. Prospective car owners have to obtain vehicle license plates through public auctions (Hao et al. 2011; Chen, R. et al. 2013). In December of 2016, the lowest winning bid was nearly ¥88,300 (US$12,600), a significant increase from bids of around ¥15,000 (US$1,800) from the early 2000s when the auction system first started (SMG 2017). In addition, vehicle emission standards have been tightened over the years to more closely match standards in high-income countries (TransportPolicy.net 2017). Shanghai has also developed an extensive public transit system—with multiple transportation mode options, including buses, taxis, metro (underground rapid transit), and bike sharing—in the attempt to reduce private automobile use. Shanghai Metro alone serves more than 10 million passengers on an average work day (Shen et al. 2016).
Level and Trend of Ambient Air Pollution

Transportation, industrial and household coal combustion, and PM from open construction sites are currently the three major sources of air pollution in Shanghai (Liu, H. et al. 2007; Wang, H. et al. 2008; Wang, L. et al. 2009; Ling 2014). This section presents the temporal trend of total emissions as well as ambient concentrations of criteria air pollutants from 1995 to 2015. The total emission data are based on inventory of total pollutant emission from various sources (common measuring unit: metric ton). They were collected from the Shanghai Statistical Yearbook (SSY) (SMSB 1990–2015). The ambient concentrations of criteria air pollutants data (measuring unit: μg/m³) were obtained from Shanghai Environment Bulletin (SEB) (SEPB 1995–2015). Gaseous pollutants are described first in the following subsection, including SO₂, NO₂, CO, and O₃, while the subsequent subsection discusses particulate matter emission and concentration. Hereafter, concentration refers to annual average concentration for all the above pollutants except for O₃, for which concentration refers to the daily maximum eight-hour average.
Figure 32.5  Typical streetscape and traffic in Shanghai, with a heavy haze seen in the background.
Source: Yukun Yang.

Figure 32.6  Annual industrial waste gas emission in Shanghai, 1995–2015.
Source: Data from SMSB (1990–2015).
Gaseous Pollutants

Total Gaseous Emissions

Gaseous emissions data are available only for industrial waste gas (major pollutants: SO\textsubscript{2} and NOx) and SO\textsubscript{2}. The annual total waste gas emission increased from 462.5 billion m\textsuperscript{3} in 1995 to 1,369.2 billion m\textsuperscript{3} in 2011 and then decreased gradually to 1,280.2 billion m\textsuperscript{3} in 2015 (Figure 32.6). Note that Baoshan district, the home of Shanghai’s heavy industry manufacturing zone, alone contributed more than half of the total emission in 2015.

Industrial and non-industrial sources (e.g., domestic heating) accounted for 61.4% and 38.6% of the SO\textsubscript{2} emission in 2015, respectively. The SO\textsubscript{2} emission fluctuated between 1995 and 2006, with the annual total emission ranging from 403,000 tons to 534,000 tons. The emission decreased rapidly from 498,000 tons in 2007 to 171,000 tons in 2015 (Figure 32.7). While the trend of the industrial SO\textsubscript{2} emission matched closely with the total emission, the non-industrial SO\textsubscript{2} emission showed more fluctuation; the annual emission was between 72,000 and 173,000 tons from 1996 to 2010 and then remained at a relatively low level, between 30,000 and 66,000 tons, from 2011 to 2015. The overall decreasing trend of SO\textsubscript{2} emission indicates that the effect of the major growth factor of SO\textsubscript{2} emission, coal consumption, was suppressed by the two SO\textsubscript{2} reduction factors, reducing high-sulfur coal consumption and more efficient desulfurization in emission control (Yang, X. et al. 2016).

Figure 32.7 Annual SO\textsubscript{2} emission in Shanghai, 1995–2015.

Source: Data from SMSB (1990–2015).
The SO\textsubscript{2} concentration decreased from 32 μg/m\textsuperscript{3} in 1995 to 20 μg/m\textsuperscript{3} in 1999 and gradually increased to 55 μg/m\textsuperscript{3} in 2004. It then fluctuated between 41 μg/m\textsuperscript{3} and 55 μg/m\textsuperscript{3} between 2004 and 2008. The concentration has halved from 35 μg/m\textsuperscript{3} in 2009 to 17 μg/m\textsuperscript{3} in 2015 (Figure 32.8). The concentration was below the NAAQS Grade II standard value (60 μg/m\textsuperscript{3}) throughout the entire observation period and was lower than the Grade I standard value (20 μg/m\textsuperscript{3}) in 2014 and 2015.

The NO\textsubscript{2} concentration ranged from 42 μg/m\textsuperscript{3} to 62 μg/m\textsuperscript{3} between 1995 and 2015 (Figure 32.8). The concentration was worse than the NAAQS Grade II standard value (40 μg/m\textsuperscript{3}) during the whole period of observation. Previous studies also showed that areas with an intensive road network, high population density, and industries had higher levels of NO\textsubscript{2} (Meng et al. 2015). Recent decreases of NO\textsubscript{2} concentration are attributed to stricter emission standards, shorter vehicle turnover, and policies limiting private vehicle ownership (Liu, J. et al. 2016).

The four-year average (2012–2015) CO concentration was 840 μg/m\textsuperscript{3}, but the data are not of a long enough duration to evaluate the trend. In addition, the CO concentration (annual average) could not be compared with NAAQS standard values because the latter were daily average values. In addition, the three-year average (2013–2015) O\textsubscript{3} concentration was 157.7 μg/m\textsuperscript{3}, lower than the NAAQS Grade II standard value (160 μg/m\textsuperscript{3}) (Figure 32.9).

However, some studies found that O\textsubscript{3} concentration has been increasing. For example, Gao et al. (2017) observed that the O\textsubscript{3} concentration had increased from 15.3 ppbv in 2006 to 25.9 ppbv in 2015 at Xujiahui. Xujiahui is an upscale commercial district in southwest Shanghai, and it is also where the Shanghai Environmental Monitoring Center (SEMC) is located. Previous studies...
Table 32.1 Concentration limits for selected air pollutants in National Ambient Air Quality Standards (NAAQS, GB 3095-2012) and the World Health Organization Air Quality Guidelines (WHOAQG) (μg/m³)

<table>
<thead>
<tr>
<th>Pollutants</th>
<th>Averaging time</th>
<th>NAAQS</th>
<th>WHOAQG</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Grade I</td>
<td>Grade II</td>
<td>Grade III</td>
</tr>
</tbody>
</table>
| SO₂        | Daily          | 50     | 150     | 250     | –
|            | Annual         | 20     | 60      | 100     | 20
| NO₂        | Daily          | 80     | 80      | 120     | –
|            | Annual         | 40     | 40      | 80      | 40
| CO         | Daily          | 4000   | 4000    | 6000    | –
| O₃         | Hourly         | 120    | 160     | 200     | 100
| PM₁₀       | Daily          | 50     | 150     | 250     | 50
|            | Annual         | 40     | 70      | 150     | 20
| PM₂₅       | Daily          | 35     | 75      | –       | 25
|            | Annual         | 15     | 35      | –       | 10

Note: a Not available. b Daily maximum eight-hour average.

Figure 32.9 Average ambient concentration of CO and O₃ in Shanghai, 2012–2015.

have reported negative relationships between O₃ concentration and NOx concentration. For instance, Gao et al. (2017) found that the O₃ concentration had been significantly inhibited by high NOx concentration, based on the observation at Xujiahui between 2006 and 2015. The strong suppressing effect of NOx on O₃ production was also observed in another study conducted in 2010 (Ran et al. 2012).
The total PM emission (PM$_{10}$ or PM$_{2.5}$) data are not available, and the total soot emission, which is mainly composed of PM$_{10}$ and PM$_{2.5}$, is the closest surrogate data that could be used to estimate the trend of the total PM emission. The total soot emission decreased from 208,000 tons in 1995 to 121,000 tons in 2015, but note the rebound between 2013 and 2014 (Figure 32.10).

Annual averages of PM concentration data are available for the examination of the trend of ambient PM pollution. The concentration of total suspended particulate (TSP) had been monitored and used as the indicator for PM pollution before it was replaced by PM$_{10}$ in 2001; later the ambient concentration of PM$_{2.5}$ also was added to the list of the criteria air pollutants, and the annual average concentration data are available from 2013. Overall, the three PM indicators all showed a downward trend: the TSP concentration from 237 μg/m$^3$ in 1995 to 154 μg/m$^3$ in 2010; the PM$_{10}$ concentration from 100 μg/m$^3$ in 2001 to 69 μg/m$^3$ in 2015; and the PM$_{2.5}$ concentration from 62 μg/m$^3$ in 2013 to 53 μg/m$^3$ in 2015 (Figure 32.11).

The 2015 PM$_{10}$ concentration was lower than the NAAQS Grade II standard (70 μg/m$^3$) but it was almost 3.5 times the WHOAQG standard value (20 μg/m$^3$). The PM$_{2.5}$ concentrations were all higher than the NAAQS Grade II standard value (35 μg/m$^3$) and certainly much higher than the WHOAQG standard value (20 μg/m$^3$). In addition, the downward trend of PM$_{2.5}$ can be confirmed by the estimates from several previous studies in Shanghai, from 101.7 μg/m$^3$ in 2003 (Cao et al. 2007), to 94.6 μg/m$^3$ in 2003–2005 (Feng, Y. et al. 2009), and 50 μg/m$^3$ in 2014 (Zhang, H. et al. 2016).

Studies also found that PM emission had not all been generated locally. For instance, Timmermans et al. (2017) estimated that only 44.6% of the background PM$_{2.5}$ and 38.7% of the PM$_{10}$ had been

Figure 32.10 Annual soot emission in Shanghai, 1995–2015.

generated locally in Shanghai in 2013. The same study also found that industry, residential combustion, and vehicle emissions together had contributed approximately 75% and 67% of the PM$_{2.5}$ and PM$_{10}$ emissions, respectively. Ding et al. (2017) found in a 2015 study that the Pearl River Delta region, the Yangtze River Delta region, the Beijing–Tianjin region, and ship emissions from the East China Sea were all significant contributors of PM to Shanghai (particularly PM$_{1.8}$).

**Health Impacts of Ambient Air Pollution**

Over the past 20 years, there has been growing research in understanding the relationship between the ambient level of air pollution and mortality and morbidity in Shanghai (Chen, B. et al. 2004; Wong, C. et al. 2008; Cao et al. 2009; Qiao et al. 2014; Xie et al. 2014). In addition, the effects of ambient air pollution on respiratory symptoms, lung function, birth outcomes, and sub-clinical parameters have been studied (Kan et al. 2012; Cai et al. 2015; Zhao, Z. et al. 2016). Further, most of the existing studies adopted standard epidemiological research designs including time-series, case crossover, panel, and cohort studies. This section summarizes the major findings in the literature. “Ambient air pollution” is shortened to “air pollution” throughout the section, unless specified otherwise.

**Daily Mortality**

All-cause (total) and cause-specific mortality are treated commonly as the most important health endpoint associated with ambient air pollution (Kan et al. 2012). Several time-series and case crossover
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studies assessed the association between short-term exposure to air pollution and daily mortality. This subsection reviews studies of total mortality and three major types of cause-specific mortality.

All-Cause Mortality

Kan and Chen (2003), based on the analysis of mortality data from June 2000 to December 2001, reported that a 10 μg/m³ increase in the moving average over 48 hours of PM₁₀, SO₂, and NO₂ concentration had corresponded to an increase of relative risk (RR) of non-accident total mortality of 1.003 (95% CI: 1.001, 1.005), 1.016 (95% CI: 1.011, 1.021), and 1.020 (95% CI: 1.012, 1.027), respectively. In a later study, the association between air pollution and daily mortality was found to be stronger in cool seasons, among females (over males), among elderly (over younger) populations, and among the less educated (over the more educated) (Kan et al. 2008). In a study based on the data from residents in nine urban districts between 2001 and 2004, G. Chen et al. (2008) found that a 10 μg/m³ increase of PM₁₀, SO₂, and NO₂ had corresponded to an increase of mortality of 0.25% (95% CI: 0.14%, 0.37%), 0.95% (95% CI: 0.62%, 1.27%) and 0.96% (95% CI: 0.66%, 1.26%), respectively.

Dai et al. (2004) assessed daily mortality data in Zhabei, a northern Shanghai district (merged into Jing’an district in 2015), between 2002 and 2003 and showed that a 10 μg/m³ increase of PM₁₀ and PM₂.₅ had been associated with a 0.53% (95% CI: 0.22%, 0.85%) and a 0.85% (95% CI: 0.32%, 1.39%) increase in mortality, respectively. Li and Pan (2013) examined the effect of PM₂.₅ exposure on the RR of mortality using the mortality data between 2004 and 2009 and found that per 1 μg/m³ increase of PM₂.₅ had corresponded to mortality by RR of 1.0056 (95% CI: 1.0022, 1.0091) and that exposure to PM₂.₅ had caused 2,980 excessive acute deaths.

Far fewer studies have investigated the effects of O₃ exposure owing to the lack of monitoring data. Kan et al. (2010) analyzed a data set between 2001 and 2004 and reported that an increase of 10 μg/m³ in the two-day average O₃ concentration had corresponded to a 0.31% (95% CI: 0.04%, 0.58%) increase in mortality. In another study, based on daily mortality data between January 2001 and December 2004, significant association between O₃ concentration and mortality was found in the cool season (1.38%; 95% CI: 0.68%, 2.07%). In addition, the effect of O₃ was found to be confounded by NO₂ but by neither PM₁₀ nor SO₂ in the multipollutant models (Zhang, Y.H. et al. 2006).

Cardiovascular Mortality

Cardiovascular disease (CVD) is one of the world’s leading causes of mortality and morbidity (Chen, Z. 2008; Yang, G. et al. 2008; WHO 2013a). Kan et al. (2003) studied the relationship between air pollution and CVD mortality using daily mortality data between June 2000 and December 2001. It was estimated that an increase in 10 μg/m³ of PM₁₀, SO₂, and NO₂ had corresponded to CVD mortality by RR of 1.004 (95% CI: 1.001, 1.007), 1.017 (95% CI: 1.009, 1.026), and 1.024 (95% CI: 1.011, 1.036), respectively. In addition, Zhang, Y.H. et al. (2006) reported that a 10 μg/m³ increase of O₃ concentration had been associated with a 0.53% increase in CVD mortality using mortality data between January 2001 and December 2004.

Respiratory Mortality

Chronic respiratory disease related to severe outdoor air pollution has become an increasingly serious public health issue in China (WHO 2013b). Kan et al. (2003) analyzed mortality data from 2000 to 2001 and estimated that the RR of daily chronic obstructive pulmonary disease (COPD) mortality on a 10 μg/m³ increase over a 48-hour moving average of PM₁₀, SO₂, and NO₂ had been
Growing evidence has also suggested an association between the exposure to air pollution and an elevated risk of cerebrovascular disease (Ljungman and Mittleman 2014; Stafoggia et al. 2014). Using mortality data collected in Zhabei district in 2001, Kan et al. (2004) reported that an increase of 10 μg/m³ of PM₁₀, SO₂, and NO₂ concentration had corresponded to an increase of stroke mortality by RR of 1.008 (95% CI: 1.000, 1.016), 1.017 (95% CI: 0.998, 1.036), and 1.029 (95% CI: 1.001, 1.057), respectively. Based on daily stroke mortality of adults aged over 65 between 2003 and 2008, Qian et al. (2013) found that the association between total stroke and ischemic stroke mortalities and the three air pollutants had all been significant, the association between hemorrhagic stroke and SO₂ and NO₂ had been significant, and the association between ischemic stroke mortality and NO₂ was much stronger among cardiac disease patients than patients without the disease.

**Morbidity**

**Air Pollution and Cardiovascular Diseases**

Epidemiological studies have consistently shown increased risk of cardiovascular diseases with the exposure to air pollution, particularly to PM (Brook et al. 2010). Using daily hospital admission data (2005–2007), R. Chen et al. (2010) reported that a 10 μg/m³ increase of PM₁₀, SO₂, and NO₂ concentration had been associated with a 0.18% (95% CI: −0.15%, 0.52%), 0.63% (95% CI: 0.03%, 1.23%), and 0.99% (95% CI: 0.10%, 1.88%) increase in total hospital admission. The study also found that a 10 μg/m³ increase of PM₁₀, SO₂, and NO₂ concentration had been associated with a 0.23% (95% CI: −0.03%, 0.48%), 0.65% (95% CI: 0.19%, 1.12%), and 0.80% (95% CI: 0.10%, 1.49%) increase in cardiovascular hospital admission, respectively.

A. Zhao et al. (2014) investigated the association between the exposure to PM₁₀, SO₂, and NO₂ and outpatient visits for arrhythmia using data collected from Shanghai Yangpu District Central Hospital between 2010 and 2011. A 10 μg/m³ increase in the present day concentrations of PM₁₀, SO₂, and NO₂ was found to be associated with an increase of 0.56% (95% CI: 0.42%, 0.70%), 2.07% (95% CI: 1.49%, 2.64%), and 2.90% (95% CI: 2.53%, 3.27%) in outpatient arrhythmia visits, respectively. In addition, stronger associations were found among female patients and patients aged 65 years or over.

Ye et al. (2016) investigated the association between PM exposure (the same day and the day before, lag 01) and the incidence of coronary heart disease (CHD) using daily CHD morbidity data in people aged above 40 years between 2005 and 2012. A 10 μg/m³ increase in PM₁₀ and PM₂.₅ concentration (lag 01) was found to be associated with an increase of 0.23% (95% CI: 0.12%, 0.34%) and 0.74% (95% CI: 0.44%, 1.04%) in CHD morbidity, respectively.

J. Wang et al. (2016) analyzed the relationship between the exposure to PM₂.₅, PM₁₀, NO₂, SO₂, and CO and acute myocardial infarction (AMI) using data between 2013 and 2014. The estimated odds ratio (OR) (95% CI) of AMI was 1.16% (95% CI: 1.03%, 1.29%), 1.05% (95% CI: 1.01%, 1.16%), 0.82% (95% CI: 0.75%, 1.02%), 0.87% (95% CI: 0.63%, 1.95%), and 1.08% (95% CI: 1.02%, 1.21%) per 10 μg/m³ increase of PM₂.₅, PM₁₀, NO₂, SO₂, and CO, respectively. In addition, more AMI occurrences were found to be associated with worse overall air quality as measured by an air quality index (AQI).
Air Pollution and Respiratory Diseases

Cai et al. (2014) assessed the acute effects of PM$_{10}$, SO$_2$, NO$_2$, and black carbon (BC) on asthmatic hospitalization using daily adult asthmatic hospitalization data from nine urban districts between 2005 and 2011. It was estimated that an interquartile range (IQR) increase in the moving average of the present day and the previous day concentrations of PM$_{10}$, SO$_2$, NO$_2$, and BC had corresponded to a 1.82% (95% CI: −1.57%, 5.20%), 6.41% (95% CI: 2.32%, 10.49%), 8.26% (95% CI: 4.48%, 12.05%) and 6.62% (95% CI: 1.74%, 11.50%) increase in asthmatic hospitalization, respectively. Moreover, SO$_2$ and NO$_2$ had stronger effects on asthma hospitalization than PM$_{10}$, and the concentration–response curves showed a linear relationship between air pollution and the risk of asthmatic hospitalization.

H. Zhang et al. (2018) analyzed data on the daily visits to the emergency and outpatient department for five main respiratory diseases and their medical expenditures between 2013 and 2015. Their models showed significant increments in emergency visits (8.81–17.26%) and corresponding expenditures (0.33–25.81%) for pediatric respiratory diseases, upper respiratory infection (URI), and COPD for an IQR increase of air pollutant concentrations over four lag days. In addition there were significant but smaller increments in outpatient visits (1.36–4.52%) and expenditures (1.38–3.18%) for pediatric respiratory diseases and URI.

The health effect of CO is more complicated. It is commonly known that high levels of CO are harmful to humans. However, studies have also shown that low-level CO has been beneficial to human health under certain conditions (Durante et al. 2006; Otterbein 2009; Tian et al. 2014). Using hospital admission COPD data between 2006 and 2008, Cai et al. (2015) found a negative association between CO and daily COPD hospitalization. More specifically, an IQR (0.6 mg/m$^3$) increase in CO concentration at a three-day lag was found to correspond to a 2.97% (95% CI: 1.31%, 4.63%) decrease in COPD hospitalization. Moreover, the negative association became more significant adjusting for co-pollutants (PM$_{10}$, SO$_2$, and NO$_2$), and the protective effect of CO was also much stronger in the cool season.

Air Pollution and Sub-Clinical Parameters

Only one study was found to have examined the association between air pollution and sub-clinical parameters. Z. Zhao et al. (2016) conducted a longitudinal panel study on the acute effect of CO on fractional exhaled nitric oxide (FeNO, a well-established biomarker of airway inflammation) among 75 healthy young adults between April and June in 2013. Exposure to CO preceding health tests was found to be significantly associated with decreased FeNO levels. In addition, an IQR increase (0.3 mg/m$^3$) of a two-hour CO exposure corresponded to a 10.6% decrease in FeNO in healthy young adults.

Air Pollution and the Total Outpatient and Emergency Room Visits

Cao et al. (2009) examined the association between exposure to air pollution and the total hospital outpatients and the total emergency room visits using data between 2005 and 2007. The results showed that a 10 μg/m$^3$ increase in concentration of PM$_{10}$, SO$_2$, and NO$_2$ had corresponded to a 0.11% (95% CI: −0.03%, 0.26%), 0.34% (95% CI: 0.06%, 0.61%), and 0.55% (95% CI: 0.14%, 0.97%) increase in total outpatient visits. In addition, a 10 μg/m$^3$ increase in concentration of PM$_{10}$, SO$_2$, and NO$_2$ had corresponded to a 0.01% (95% CI: −0.09%, 0.10%), 0.17% (95% CI: 0.00%, 0.35%), and 0.08% (95% CI: −0.18%, 0.33%) increase in total emergency room visits, respectively.
Researchers have established several mechanisms to link adverse birth outcomes to air pollution exposure. However, the reported associations have not always been consistent (Shah and Balkhair 2011). L.L. Jiang et al. (2007) studied the association between preterm birth (PTB) and the average concentrations of PM$\text{_{10}}$, SO$_2$, NO$_2$, and O$_3$ using daily PTB numbers in 2004. A significant association was found between PTB and an eight-week average concentration of air pollutants prior to birth. It was found that an increase of 10 μg/m$^3$ of eight-week average PM$\text{_{10}}$, SO$_2$, NO$_2$, and O$_3$ corresponded to a 4.42% (95% CI: 1.60%, 7.25%), 11.89% (95% CI: 6.69%, 17.09%), 5.43% (95% CI: 1.78%, 9.08%), and 4.63% (95% CI: 0.35%, 8.91%) increase in the PTB risk, respectively.

A. Liu et al. (2017) calculated concentration–response relationships between PM$\text{_{2.5}}$ exposure during pregnancy and the PTB and low birth weight (LBW) rates using a 10km×10km population grid in 2013. Based on the NAAQS Grade I standard value of PM$\text{_{2.5}}$ (15 μg/m$^3$), the weighted RRs of PTB and LBW associated with PM$\text{_{2.5}}$ were 1.49 (95% CI: 1.16, 1.80) and 1.31 (95% CI: 1.04, 1.67), respectively. It was also estimated that 32.61% (95% CI: 13.93%, 44.42%) or 4,160 (95% CI: 1778, 5667) PTB cases and 23.36% (95% CI: 3.86%, 40.02%) or 1,882 (95% CI: 311, 3224) LBW cases might be attributable to PM$\text{_{2.5}}$ exposure.

The Size and Chemical Constituents of PM and Health Outcomes

The health effects of PM exposure vary significantly with the size of PM (Chen, R., Zhao et al. 2015; Thomson et al. 2015). Using data collected from the Tianping Community, a neighborhood in southwest Shanghai, between April and June in 2013, A. Zhao et al. (2015) found that an IQR increase in a 24-hour average number concentration of PM$_{0.25-0.40}$ had corresponded to an increase of 3.61 mmHg in systolic blood pressure (SBP) and 2.96 mmHg in pulse pressure (PP), respectively. R. Chen, Zhao et al. (2015) reported a positive association between exposure to PM and ten biomarkers of inflammation, coagulation, and vasoconstriction in a panel study involving 34 healthy young adults from April to June 2013. Small-sized PM was found to have a stronger association, and the strongest association was found at size fractions between 0.25 and 0.40 μm measured as particulate number concentrations (PNC) and <1 μm measured as particulate mass concentrations (PMC). C. Wang et al. (2015) demonstrated that particle size (PM$_{0.25-0.40}$) and time windows (even within two hours) of exposure had been responsible for increases in biomarkers among diabetes patients using data collected from 35 diabetes patients between April and June in 2013.

Furthermore, the main chemical constituents of PM are organic carbon (OC), elemental carbon (EC), and ions. The varying chemical constituents of PM may impact health effects (Brook et al. 2010). Qiao et al. (2014) examined the effects of PM$_{2.5}$ constituents including OC, EC, and eight water-soluble ions on emergency room visits using data between 2011 and 2012. For a one-day lag, an IQR increase (36.47 μg/m$^3$) in PM$_{2.5}$ was found to correspond with a 0.57% (95% CI: 0.13%, 1.01%) increase in emergency room visits. Significant positive associations between emergency room visits and OC and EC were also found after controlling for the total PM$_{2.5}$ mass. In addition, the association between most of the ions and emergency room visits was significant before adjusting for the total PM$_{2.5}$ mass.

Based on a panel study conducted from May to July 2014 on 30 retired COPD patients from a central urban community, R. Chen, Qiao et al. (2015) investigated the effects of PM$_{2.5}$ constituents (Cl$^-$, NO$_3^-$, SO$_4^{2-}$, NH$_4^+$, Na$^+$, K$^+$, Mg$^{2+}$, Ca$^{2+}$, OC, and EC) on FeNO and the DNA methylation of its encoding gene (NOS2A) among COPD patients. It was found that OC, EC, NO$_3^-$, and NH$_4^+$ had been associated with decreased NOS2A DNA methylation and elevated FeNO among COPD patients. Shi et al. (2016) performed a longitudinal panel study on 32 healthy young adults from January to February 2015, with the results showing that EC had been responsible for an increase in airway inflammation among these adults.
Discussion

Air Pollution and Health Effects

Over the past 25 years, Shanghai has made significant progress in controlling industrial pollution, with air pollution showing an overall downward trend based on official emission and concentration data. However, vehicle emissions have become a major pollution source, and PM-related air pollution is still severe. Shanghai has established an extensive air pollution monitoring network, with 56 stations across the city (Figure 32.1), but only a small portion of the aggregated data have been made available to the public. In addition, few official emission inventories are easily accessible despite the growing scholarly work (Liu, H. et al. 2007; Wang, W. et al. 2007; Wang, H. et al. 2008; Fu et al. 2010). Thus this chapter has at best painted a crude picture of the current level and the temporal trend of ambient air pollution in Shanghai.

Efforts have been made in understanding the spatial distribution of air pollution despite the problem in data availability. Pollution surfaces were interpolated using various methods, including geographically weighted gradient boosting machine (GW-GBM), empirical Bayesian kriging (EBK), and land use regression (LUR) (Meng et al. 2015; Rohde and Muller 2015; Zhang, H. et al. 2016; Zhan et al. 2017). The results from these studies have provided the basis to further explore important questions, including exposure equality and population-weighted-average (PWA) concentrations of air pollution (Ivy et al. 2008; Song, C. et al. 2017).

Our literature review suggests that there has been a significant association between the short-term exposure to ambient air pollution and mortality and morbidity in Shanghai. However, such associations need to be interpreted with caution because of several well-known methodological challenges in the standard epidemiological research designs adopted by most of these studies (Ren and Tong 2008). In addition, the availability of and accessibility to health data, very much as with pollution data, are relatively restricted. Access to data remains a critical barrier to research efforts, particularly to long-term exposure analyses.

Suggestions and Directions for Future Research

Greater transparency and accessibility of air pollution and health data are the foundation for the future research that is critical to improving our understanding of the interactions between emissions, atmospheric chemistry, air quality, local weather, global climate changes, and health outcomes. Better coordination is needed between air pollution monitoring and environmental health assessment and prediction. Active collaboration across disciplines—including public health and atmospheric and environmental science—will benefit complex monitoring and modeling tasks, especially for exposure to complex air pollution mixtures. The use of supersites to perform simultaneous studies using the same monitoring and health evaluation approaches across Shanghai is strongly recommended. In addition, to make the routine monitoring data more available, dynamic emission sources, including vehicular and ship emissions, should be routinely updated.

Researchers may take advantage of the free air pollution data provided by the growing number of non-government sources. For instance, the Chinese Air Quality Online Monitoring and Analysis Platform currently offers real-time air pollution data from ten monitoring stations in Shanghai (Wang 2017). An air quality data portal, World Air Quality Index, publishes data from more than 9,000 monitoring stations in 600 major cities of more than 70 countries, including China (World Air Quality Index 2017). Multi-resolution Emission Inventory for China (MEIC) provides emission data of ten pollutants and greenhouse gases based on the estimation from more than 700 anthropogenic emission sources (MEIC 2017). Furthermore, new technologies and approaches in data collection and dissemination, including mobile monitoring devices and social media, have shown great
potential in improving the spatial and temporal resolution of data and providing near real-time data and timely health risk warnings (Jiang, W. et al. 2015; Kay et al. 2015; Gozzi et al. 2016; Sun et al. 2016; Castell et al. 2017; Luo et al. 2017).

While this chapter focused on outdoor air pollution, humans spend most of their time indoors (Sexton et al. 2004). Many technological and socioeconomic changes—the use of new synthetic materials, reliance on childcare and elderly care facilities, and the increase in office jobs—have also intensified Shanghai’s population exposure to indoor air pollution (IAP). Thus, future research should investigate the impacts of both indoor and outdoor pollutants (Jones 1999; Tham 2016).

Future air pollution health research may consider prospective cohort studies, examining long-term and cumulative exposure, covering the most susceptible time periods and population groups (e.g., floating and aged populations), and including gene–environment interactions, and pathophysiological links between air pollution and cardiopulmonary diseases. The relative strengths of interpolation methods, shape of the exposure response curve, threshold of air pollution, interactive effects of air pollution, and weather conditions should also be explored. The potential health risks of climate change also need to be seriously investigated (USCDC 2018).

From a health prevention and intervention perspective, efforts should be made to translate science into compelling and simple messages so that the general public, particularly susceptible individuals, can be informed about the real-time health risk (Kelly and Fussell 2015). The Air Quality Health Index (AQHI) has been proven to be an effective communication tool to help achieve such a goal (Chen, R. et al. 2012; Wong, T.W. et al. 2013).

The Shanghai Master Plan 2040 states clearly that Shanghai will be “a city of prosperity and innovation with global competitiveness, a green and livable city with sustainable development capacity, and a more humanistic city with more cultural attractiveness and happiness” (China City Planning Review 2016). Some evidence suggests that the population health in Shanghai has been improving significantly since the 2000s (Gusmano et al. 2015), but air pollution from sources both within and outside the city remains a major health threat to all the residents in Shanghai. Thus, it is necessary for us to continue research on the complex relationships between urbanization, demographic and behavioral changes, socioeconomic development, air pollution, climate change, and population health.

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References


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## Abbreviations

<table>
<thead>
<tr>
<th>Abbreviation</th>
<th>Description</th>
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<tbody>
<tr>
<td>AMI</td>
<td>Acute myocardial infarction</td>
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<tr>
<td>AQHI</td>
<td>Air Quality Health Index</td>
</tr>
<tr>
<td>AQI</td>
<td>Air quality index</td>
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<tr>
<td>BC</td>
<td>Black carbon</td>
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<tr>
<td>CHD</td>
<td>Coronary heart disease</td>
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<tr>
<td>CO</td>
<td>Carbon oxide</td>
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<tr>
<td>COPD</td>
<td>Chronic obstructive pulmonary disease</td>
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<tr>
<td>CVD</td>
<td>Cardiovascular disease</td>
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<td>EBK</td>
<td>Empirical Bayesian kriging</td>
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<td>EC</td>
<td>Elemental carbon</td>
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<td>FeNO</td>
<td>Fractional exhaled nitric oxide</td>
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<tr>
<td>GDP</td>
<td>Gross domestic product</td>
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<td>GW-GBM</td>
<td>Geographically weighted gradient boosting machine</td>
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<td>IAP</td>
<td>Indoor air pollution</td>
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<td>IQR</td>
<td>Interquartile range</td>
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<tr>
<td>LBW</td>
<td>Low birth weight</td>
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<td>LUR</td>
<td>Land use regression</td>
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<td>MEIC</td>
<td>Multi-resolution Emission Inventory for China</td>
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<td>NAAQS</td>
<td>National Ambient Air Quality Standards</td>
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<td>NOx</td>
<td>Nitrogen oxides</td>
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<td>NO$_2$</td>
<td>Nitrogen dioxide</td>
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<td>O$_3$</td>
<td>Ozone</td>
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<tr>
<td>OC</td>
<td>Organic carbon</td>
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<tr>
<td>PM</td>
<td>Particulate matter</td>
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<tr>
<td>PM$_{2.5}$</td>
<td>Particulate matter with aerodynamic diameters ≤2.5 μm</td>
</tr>
<tr>
<td>PM$_{10}$</td>
<td>Particulate matter with aerodynamic diameters ≤10 μm</td>
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<tr>
<td>PMC</td>
<td>Particulate mass concentration</td>
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<tr>
<td>PNC</td>
<td>Particulate number concentration</td>
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<td>PP</td>
<td>Pulse pressure</td>
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<tr>
<td>PTB</td>
<td>Preterm birth</td>
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<tr>
<td>PWA</td>
<td>Population-weighted-average</td>
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<td>RH</td>
<td>Relative humidity</td>
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<tr>
<td>RR</td>
<td>Relative risk</td>
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<tr>
<td>SBP</td>
<td>Systolic blood pressure</td>
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<tr>
<td>SCR/SNCR</td>
<td>Selective catalytic/non-catalytic reduction</td>
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<tr>
<td>SEB</td>
<td>Shanghai Environment Bulletin</td>
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<tr>
<td>SMG</td>
<td>Shanghai Municipal Government</td>
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<tr>
<td>SO$_2$</td>
<td>Sulfur dioxide</td>
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<tr>
<td>SSY</td>
<td>Shanghai Statistical Yearbook</td>
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<tr>
<td>TCE</td>
<td>Tons of coal equivalent</td>
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<td>TSP</td>
<td>Total suspended particulate</td>
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<tr>
<td>URI</td>
<td>Upper respiratory infection</td>
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<tr>
<td>WHOAQG</td>
<td>World Health Organization Air Quality Guidelines</td>
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