Current scenario of ozone pollution and its influence on population health in China

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Abstract: Ozone (O$_3$) is a secondary pollutant formed by photochemical reactions in the atmosphere, significantly contributing to air pollution, particularly in global cities and economically developed regions. The escalating O$_3$ concentration has emerged as a critical air pollution concern in China. When near-surface O$_3$ surpasses natural levels, it adversely impacts human health. Our understanding of O$_3$ pollution remains limited, partly due to the delayed implementation of atmospheric O$_3$ and its precursors' monitoring. Accordingly, utilizing observed data, this paper assesses the O$_3$ levels in China over the past decade. We found that surface O$_3$ concentrations had consistently risen since 2013, with the only decline noted in 2018. O$_3$ pollution is particularly severe in economically developed areas such as the Beijing-Tianjin-Hebei region, the Yangtze River Delta, the Pearl River Delta, and the Weihe Plain. Chronic exposure to O$_3$ can negatively impact respiratory and cardiovascular systems. By introducing research findings related to O$_3$ exposure and human health, we offer suggestions for future research on human health implications of surface O$_3$ exposure. These findings underscore the importance of O$_3$ as a focal point in China's future air quality management.
quality policy and highlight the urgent necessity for stricter control of precursor emissions.

**Keywords:** Ozone; China; pollution trends; ozone exposure; human health

**Introduction**

Air pollution is an important global public health concern. Ozone (O₃) is a common component of air pollutants and is produced by the photochemical reactions of volatile organic compounds (VOCs) and nitrogen oxides (NOₓ) emitted by natural and anthropogenic sources under high-temperature ultraviolet irradiation (Sillman, 1999). A number of air pollution incidents have harmed human health in the past, such as the smog incidents in Los Angeles in the United States and in London in the United Kingdom (Guan et al., 2016). As the largest developing country and the second-largest economy in the world, China has made great economic achievements since its reform and opening up. With the rapid development of China's economy, however, there has been an increase in the consumption of chemical fuels and O₃ pollution is becoming increasingly serious (Liu et al., 2015). Previous studies indicated that the distribution of O₃ had strong seasonality and generally showed an inverted V-shaped trend (Lei et al., 2019; Li et al., 2019; Wang et al., 2020). In recent years, particulate matter pollution has significantly decreased, but the pollution of O₃ has worsened. According to the China Environment Report 2019 (available at https://www.mee.gov.cn, in Chinese), 30% of the 338 cities in China have O₃ concentrations exceeding the secondary limit of Environmental Air Quality Standards (GB3095-2012) (160 µg/m³ [101.325 kPa, 20 °C]), especially in the Beijing-Tianjin-Hebei and Yangtze River Delta regions (Wei et al., 2017; Zhang et al., 2015).

With rapid industrialization and urbanization, O₃ has become a major air pollutant in China (Al-Jassim et al, 2018; Lu et al., 2018). Due to its low water solubility, it easily enters the respiratory tract; therefore, it has a strong stimulating effect and undergoes strong oxidation
(Nuvolone et al., 2017; Zhang et al., 2019b). The respiratory system is the first part of the human body to receive O$_3$ following inhalation and, therefore, suffers the most obvious effects of O$_3$ exposure. Epidemiological studies have shown a significant positive correlation between acute O$_3$ exposure and decreased lung function in both average and susceptible individuals, especially in seasons with high O$_3$ concentrations (Adams & William, 2002, 2003, 2006). According to several epidemiological studies, Long - and short-term exposure to high concentrations of O$_3$ can lead to respiratory diseases such as infections and asthma, cardiovascular diseases such as stroke and arrhythmias, and neurological diseases such as autism and Alzheimer's disease in children (Bell et al., 2005; Dominici et al., 2006; Pope & Dockery, 2006). O$_3$ exposure is an important factor that can lead to premature death in humans. It can activate a large number of inflammatory mediators in the respiratory system, leading to the accumulation of toxic lipid oxidation products and eventually chronic inflammation (Canella et al., 2016). In addition, it can produce strongly oxidizing free radicals in the human body, disrupt cell metabolism, accelerate senescence, induce chromosomal lesions in lymphocytes, and damage the immune system (Rider & Carlsten, 2019). The exposure of certain groups to high concentrations of O$_3$, such as pregnant women, infants, and young children, may cause serious health threats and even increase the risk of mortality (Silva et al., 2013). A review that O$_3$ pollution trends and health impacts in China is therefore needed to aid in formulating a mitigation policy and to guide future research.

Specifically, this study aims to evaluate the current state of O$_3$ pollution in mainland Chinese cities on an annual basis from 2013 to 2021 and review O$_3$ pollution's impact on population health. The significance of this study is twofold. Firstly, it elucidates the prevailing situation and trends of O$_3$ pollution in mainland China over the past decade, with the aim of drawing attention to and prompting further research on this issue. Secondly, the study can aid in understanding the health
effects of O₃ and support the formulation of O₃ standards in China.

2. Current scenario of O₃ pollution in China

Since the 1990s, with the increase in the number of motor vehicles, there has been a parallel increase in coal, oil, and other forms of energy consumption, such that compound air pollution has replaced the typical soot-type pollution in China. With the gradual implementation of the government’s environmental optimization policy, particulate matter pollution has significantly reduced, although O₃ pollution is becoming increasingly serious. In recent years, O₃ concentrations exceeding the Chinese standard have become increasingly common.

According to the bulletin on the state of China’s environment issued by the Ministry of Ecology and Environment from 2013 to 2021, China has set up national air monitoring stations in 74 cities in the regions of Beijing, Tianjin, Hebei, Yangtze River Delta, Pearl River Delta, and in municipalities directly under the Central Government, as well as in provincial capitals and cities separately listed on the State plan, to carry out air monitoring in accordance with the new Environmental Air Quality Standards (GB3095-2012). The 2013–2021 average concentration of O₃-8h (Daily maximum 8h average) and the 90th percentile range in Chinese cities are shown in Table 1, and the changing trends are shown in Fig. 1.

Table 1 Mean value of O₃-8h concentration and 90th percentile range in some Chinese cities from 2013 to 2021

<table>
<thead>
<tr>
<th>Year</th>
<th>Number of cities (number)</th>
<th>Average O₃-8h concentration</th>
<th>Range of 90th percentile of O₃-8h concentration (μg/m³)</th>
<th>Over-standard rate (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
### (µg/m³)

<table>
<thead>
<tr>
<th>Year</th>
<th>City</th>
<th>O₃ Concentration</th>
<th>Range</th>
<th>Note</th>
</tr>
</thead>
<tbody>
<tr>
<td>2013</td>
<td>74</td>
<td>139</td>
<td>72–190</td>
<td>—</td>
</tr>
<tr>
<td>2014</td>
<td>161</td>
<td>140</td>
<td>69–210</td>
<td>6.1</td>
</tr>
<tr>
<td>2015</td>
<td>338</td>
<td>134</td>
<td>62–203</td>
<td>4.6</td>
</tr>
<tr>
<td>2016</td>
<td>338</td>
<td>138</td>
<td>73–200</td>
<td>5.2</td>
</tr>
<tr>
<td>2017</td>
<td>338</td>
<td>149</td>
<td>78–218</td>
<td>7.6</td>
</tr>
<tr>
<td>2018</td>
<td>338</td>
<td>151</td>
<td>76–217</td>
<td>8.4</td>
</tr>
<tr>
<td>2019</td>
<td>338</td>
<td>148</td>
<td>—</td>
<td>7.6</td>
</tr>
<tr>
<td>2020</td>
<td>338</td>
<td>138</td>
<td>—</td>
<td>4.9</td>
</tr>
<tr>
<td>2021</td>
<td>338</td>
<td>137</td>
<td>—</td>
<td>4.4</td>
</tr>
</tbody>
</table>


**Figure 1.** The changing trend of O₃ concentration in the key regions of China (Beijing-Tianjin-Hebei and Yangtze River Delta) and the entire country from 2013 to 2021.

Overall, the O₃ concentrations in the Beijing-Tianjin-Hebei region, Yangtze River Delta, Pearl...
River Delta, and Weihe Plain were higher than those in other areas, as shown in Table 2. The concentration of O₃ in the Beijing-Tianjin-Hebei region was the highest, with an average annual concentration of 176.67 μg/m³, and the proportion of over- Environmental Air Quality Standard (GB3095-2012) for O₃ in the Pearl River Delta was the largest, with was 57.33%.

Table 2 O₃ pollution in the key regions since 2013

<table>
<thead>
<tr>
<th>Year</th>
<th>Index</th>
<th>BTH</th>
<th>YRD</th>
<th>PRD</th>
<th>WP</th>
</tr>
</thead>
<tbody>
<tr>
<td>2013</td>
<td>O₃ mean concentration (μg/m³)</td>
<td>155</td>
<td>144</td>
<td>155</td>
<td>—</td>
</tr>
<tr>
<td></td>
<td>O₃ exceedance ratio (%)</td>
<td>7.6</td>
<td>13.9</td>
<td>31.9</td>
<td>—</td>
</tr>
<tr>
<td>2014</td>
<td>O₃ mean concentration (μg/m³)</td>
<td>162</td>
<td>154</td>
<td>156</td>
<td>—</td>
</tr>
<tr>
<td></td>
<td>O₃ exceedance ratio (%)</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td>2015</td>
<td>O₃ mean concentration (μg/m³)</td>
<td>162</td>
<td>163</td>
<td>145</td>
<td>—</td>
</tr>
<tr>
<td></td>
<td>O₃ exceedance ratio (%)</td>
<td>17.2</td>
<td>37.2</td>
<td>56.5</td>
<td>—</td>
</tr>
<tr>
<td>2016</td>
<td>O₃ mean concentration (μg/m³)</td>
<td>172</td>
<td>159</td>
<td>151</td>
<td>—</td>
</tr>
<tr>
<td></td>
<td>O₃ exceedance ratio (%)</td>
<td>26.3</td>
<td>39.8</td>
<td>70.3</td>
<td>—</td>
</tr>
<tr>
<td>2017</td>
<td>O₃ mean concentration (μg/m³)</td>
<td>193</td>
<td>170</td>
<td>165</td>
<td>—</td>
</tr>
<tr>
<td></td>
<td>O₃ exceedance ratio (%)</td>
<td>41.0</td>
<td>50.4</td>
<td>70.6</td>
<td>—</td>
</tr>
<tr>
<td>2018</td>
<td>O₃ mean concentration (μg/m³)</td>
<td>199</td>
<td>167</td>
<td>—</td>
<td>180</td>
</tr>
<tr>
<td></td>
<td>O₃ exceedance ratio (%)</td>
<td>46.0</td>
<td>49.3</td>
<td>—</td>
<td>36.4</td>
</tr>
<tr>
<td>2019</td>
<td>O₃ mean concentration (μg/m³)</td>
<td>196</td>
<td>164</td>
<td>—</td>
<td>171</td>
</tr>
<tr>
<td></td>
<td>O₃ exceedance ratio (%)</td>
<td>48.2</td>
<td>49.5</td>
<td>—</td>
<td>37.6</td>
</tr>
<tr>
<td>2020</td>
<td>O₃ mean concentration (μg/m³)</td>
<td>180</td>
<td>152</td>
<td>—</td>
<td>161</td>
</tr>
<tr>
<td></td>
<td>O₃ exceedance ratio (%)</td>
<td>46.6</td>
<td>50.7</td>
<td>—</td>
<td>36.1</td>
</tr>
</tbody>
</table>
O3 mean concentration (μg/m³) | 171 | 151 | — | 165
---|---|---|---|---
O3 exceedance ratio (%) | 41.8 | 55.4 | — | 39.3

Note: BTH, Beijing-Tianjin-Hebei; YRD, Yangtze River Delta; PRD, Pearl River Delta; WP, Weihe Plain.

O3 is primarily formed through the photochemical reactions of NOX and VOCs (Sillman, 1999), which mainly stem from biomass fuel combustion, urban construction, automobile exhaust, and natural sources (Xu et al., 2019; Zong et al., 2018). The concentration of O3 in the economically developed eastern part of China is higher than in other regions (Peng et al., 2017). Kamal et al. (2019) reported that O3 concentrations in China range from 74 to 201 μg/m³. Approximately 30% of the population experiences O3 levels exceeding 160 μg/m³, and approximately 67.2% of the population is exposed to an O3 environment greater than 100 μg/m³ (Kamal et al., 2019). The comprehensive control and monitoring of O3 in China began relatively late. Since 2012, the Ministry of Ecology and Environment of the People's Republic of China has been listing O3 as an environmental air pollutant. In 2013, large-scale air monitoring stations were built; in recent years, air monitoring stations have been gradually installed in the entire country. As shown in Table 1 and Fig. 2, before 2018, the over-standard rate of O3 increased annually; after 2018, the over-standard rate of O3 also improved significantly as the government issued a series of targeted control measures on atmospheric O3 pollution.
**Figure 2.** Ozone levels exceeding the standard rate in some Chinese cities, from 2013 to 2021 (Note in The data come from the Ministry of Ecology and Environment of the People's Republic of China, available at https://www.mee.gov.cn)

3. An epidemiological study of the effect of O$_3$ on human health

Since the smog events in London and Los Angeles in the last century, increasing attention has been paid to air pollution, and researchers from various countries have begun to study the effects of O$_3$ exposure on human health. These effects include an increase in mortality and morbidity, and a decrease in lung function (Huangfu & Atkinson, 2020; Li et al., 2020). A Chinese multi-city study found that when the concentration of O$_3$-8h increased by 10 μg/m$^3$, the total risks of mortality, cardiovascular disease, hypertension, coronary artery disease, and stroke mortality increased by 0.23%, 0.27%, 0.60%, 0.24%, and 0.29%, respectively (Peng et al., 2017). The main effects of O$_3$ exposure on the respiratory and cardiovascular systems and potential mechanisms of its impact on human health are detailed in the following paragraphs.
3.1. Effects of O$_3$ on the respiratory system

The respiratory tract is the primary route through which air pollutants enter the human body. Most respiratory diseases involve airway lesions, including airway inflammation, remodeling, changes in responsiveness, and decreased host resistance to infection (Thurston et al., 2016). The forced expiratory volume in 1 sec (FEV1) is an important index for studying the effect of O$_3$ exposure on the respiratory system. O$_3$ concentration is usually controlled at 40–600 ppb in a study population of healthy non-smoking adults (McDonnell et al., 2012). A study of 30 healthy young people reported that FEV1 decreased significantly with O$_3$ exposure at 0.08 ppm, compared with 0.06 ppm. In studies conducted under similar exposure conditions (0.06 ppm exposure for 6.6 h), Brown et al. (2008) discovered that O$_3$ exposure led to an average 2.85% decrease in FEV1 (Brown et al., 2008). Concurrently, Chong et al. (2011) reported a 1.71% decrease in FEV1 expression (Chong et al., 2011). Furthermore, in a separate study, FEV1 decreased by 2.72% with 0.060 ppm of O$_3$, 5.34% with 0.070 ppm, 7.02% with 0.080 ppm, and as much as 11.42% with 0.087 ppm (Schelegle et al., 2009).

In summary, there was a significant negative correlation between O$_3$ concentration and pulmonary function and a positive correlation between O$_3$ concentration and number of hospitalizations and visits to the emergency department, especially in the hot season (Darrow et al., 2011; Stieb et al., 2009; Strickland et al., 2010). Studies at home and abroad have found a positive correlation between O$_3$ exposure and hospital admission for respiratory diseases (Cakmak et al., 2006; Dales et al., 2006; Mercedes et al., 2006; Silverman & Ito, 2010; Wong et al., 2010; Yang et al., 2005).

Recent studies have focused on the relationship between O$_3$ exposure and respiratory mortality. Single- and multi-city studies have found that there is a significant positive correlation between O$_3$
exposure and respiratory mortality (Bell & Michelle, 2004; Lei et al., 2019; Li et al., 2020; Peng et al., 2017; Wu et al., 2019). These studies also found that the effect of O₃ on the respiratory system exhibited lag and seasonal effects. Shang et al. (2013) conducted a meta-analysis, and the results showed that when the O₃ concentration increased by 10 μg/m³, the mortality rate of respiratory diseases increased by 0.73% (95% CI: 0.49–0.97%) (Shang et al., 2013).

3.2. Effects of O₃ on the cardiovascular system

Cardiovascular disease poses one of the most important public health problems in our country. O₃ exposure can lead to changes in heart rate and a decrease in heart rate variability has been identified as a predictor of increased cardiovascular morbidity and mortality. Studies by Liao et al. (2004) and others demonstrated a relationship between O₃ exposure and cardiovascular diseases (Liao et al., 2004; Park et al., 2005; Ruidavets & J.-B., 2005). It has also been reported that short-term O₃ exposure has no effect on the vascular function, blood pressure, or heart rate (Barath et al., 2013; Hoffmann et al., 2012). In a study of veterans, it was found that for every 2.6 μg/m³ increase in O₃ concentration, heart rate variability decreased by 11.5% (95% CI: 0.4–21.3%), and this effect was more evident in men with hypertension and ischemic heart disease than in men with good heart health (Park et al., 2005).

Some studies have shown that O₃ exposure has a greater impact on cardiovascular mortality in aging populations as compared to young ones, and the effect of O₃ is stronger in summer (Samoli et al., 2009). Studies carried out in the United States and China found different impacts of O₃ on the health of populations.

A study of 48 cities in the United States found that each 10 ppb increase in the concentration of O₃ increased cardiovascular mortality by 0.5% (95% CI: 0.30–0.60%) (Zanobetti et al., 2008), while another study of 96 US cities showed an increase in cardiovascular mortality by 1.09% (95%
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CI: 0.30–2.25%) (Jerrett et al., 2009). Li et al.’s research in Guangzhou showed that every 10 μg/m³ increase in O₃ concentration increased the mortality rate of cardiovascular disease by 0.59% (95% CI: 0.30–0.88%) (Li et al., 2021), whereas Zhang et al.’s study in Nanjing demonstrated that the mortality rate of cardiovascular disease increased by 0.98% (95% CI: 0.59–1.38%) (Zhang et al., 2019b). Therefore, overall, O₃ exposure has a greater impact on the health of Chinese residents compared with that of United States residents.

3.3. Mechanisms underlying the effects of O₃ on human health

O₃ is ubiquitous and has low water solubility, allowing it to easily enter deep into the respiratory tract, where it has a strong stimulating effect and strong oxidation. More than 80% of the O₃ absorbed by humans is inhaled through respiratory tract, where it can cause injury. Studies have shown that O₃ in certain concentrations reacts with cells in the respiratory tract and leads to increase in airway inflammation (Wu et al., 2011). Related studies have shown that O₃ can cause epithelial cells to release reactive oxygen species (ROS) and prostaglandin E2 (PGE2), activate the NF-κB signaling pathway, initiate the transcription of pro-inflammatory factors such as interleukin-8 (IL-8), and cause neutrophils to gather in the bronchi, leading to bronchial inflammation and injury. The effect of O₃ is aggravated when combined with the synergistic action of particulate fine particulate matter (Damera et al., 2009; Li et al., 2013; Sunil et al., 2013; Wu et al., 2011). Researchers have found that O₃ can also activate tyrosine kinases, promote the phosphorylation of epithelial cells, and upregulate the expression of IL-8 in the bronchi (Weidong et al., 2015).

O₃ exposure can cause oxidative decomposition, peroxidative modification, and peroxidation of lipids in the body, resulting in potential lipid peroxidation. Peroxidation increases aging-related risks and cancer, while ROS stimulate the rapid division of tumor cells (Wang et al.). In vitro
studies have demonstrated that O₃ can stimulate alveolar cells to release ROS and induce cell membrane lipid peroxidation, leading to a mixture of ROS and lipid ozonation products (Kadiiska et al., 2013). This can result in damage to the lungs and other organs. Some researchers have found that O₃ exposure can also lead to the destruction of spirochete DNA structures, induce DNA mutations, and inhibit DNA replication. Additionally, O₃ can induce systemic effects by regulating the activation of the neurohormonal stress response pathway.

3.4. Comparison of the effects of O₃ exposure on human health among countries worldwide

Globally, O₃ exposure is believed to have contributed to 250,000 premature deaths in 2015 (Lin et al., 2018). A study that examined O₃ levels and their health consequences in 50 cities in the eastern United States determined that higher O₃ levels were associated with an increase in total daily mortality of approximately 0.11–0.27% (Bell et al., 2007). Moreover, studies have indicated that in the United States, elevated O₃ levels due to climate change may result in an additional 50 premature deaths per year nationwide (Stowell et al., 2017). Future O₃ levels in Fennoscandia and the northern United Kingdom are predicted to exceed the World Health Organization Global Air Quality Guidelines level of 50 ppb. Consequently, by 2050, it is anticipated that acute effects of O₃ on human health will diminish in the majority of Fennoscandia, but chronic effects are likely to persist or perhaps worsen (Karlsson et al., 2017). In Athens, Greece, long-term exposure to O₃ was identified as the main cause of reduced life expectancy, while short-term exposure to O₃ was not found to have the same impact; the relevant characterization factors for human health damage suggested that the impact of short-term exposure to O₃ ranged between \(1.58 \times 10^{-7}\) and \(4.71 \times 10^{-7}\) years of life lost (Kassomenos et al., 2013). Surface O₃ levels in Delhi, India have been measured at concentrations that are considerably higher than hazardous levels, exceed the exposure threshold for
human health for up to 45 days annually, and far exceed the European Union directive's maximum of 25 days annually (Ghude et al., 2009). There is strong evidence that there is a positive correlation between daily $O_3$ concentrations that exceed existing regulatory requirements and daily non-accidental death (Bell et al., 2007; Karlsson et al., 2017; Nuvolone et al., 2017; Stowell et al., 2017; Zhang et al., 2019).

4. Ozone exposure assessment model

Typical studies of the atmospheric $O_3$ take the concentration exposure levels from the monitoring station to be those of the population, which could lead to serious measurement errors. With the continued development of computer technology and geographic information, global positioning, and remote sensing systems, air pollutant exposure assessment models have diversified. At present, the exposure assessment models used by researchers mainly include proximity, spatial interpolation, land-use regression, atmospheric diffusion, and Bayesian spatiotemporal models. These statistical models provide a more precise assessment of air pollutants.

The parameters used by each exposure assessment model are different, with varying advantages and disadvantages, as shown in Table 4 below.
Table 4 Principles, advantages, and disadvantages of various exposure assessment models

<table>
<thead>
<tr>
<th>Exposure model</th>
<th>Principle</th>
<th>Advantages</th>
<th>Disadvantages</th>
</tr>
</thead>
<tbody>
<tr>
<td>Proximity model (Andersson et al., 2011)</td>
<td>Assign the parameters needed by the research object to the coordinates of the geographic information system (GIS), combined with the environmental variables in the study area, compare the distance between the research site and the air pollution source, and evaluate the impact of the pollution source on the health of the research object</td>
<td>Easy to operate and less expensive, and can also use geographic information systems</td>
<td>Only qualitative analyses can be carried out. The results are subject to the influence of environmental contaminants' sources and concentration distributions, and there is a certain degree of report deviation.</td>
</tr>
<tr>
<td>Spatial interpolation model (Jin &amp; Heap, 2011)</td>
<td>Some geographic information statistical techniques are used to infer the concentration of pollutants in the study area by obtaining pollution data from monitoring points around the area</td>
<td>This model uses the monitoring points to obtain the temporal and spatial changes of pollutants</td>
<td>Because of the large consumption of funds, professional and technical personnel are needed, and the monitoring data have errors and...</td>
</tr>
</tbody>
</table>
Land use regression model (Alexeeff et al., 2015; Morley & Gulliver, 2018) The concentration of pollutants measured by the air quality monitoring station is used as the dependent variable, and geographical variables such as land use, traffic roads, topography, and population distribution around the monitoring station are predictive variables to establish a regression model. Thus, the regression model is used to predict the concentration of pollutants at any spatial site in the study area.

Compared with the interpolation model, this model has a lower economic cost and more accurate prediction of atmospheric pollutant concentrations. It cannot effectively distinguish the influence of major pollutants, struggles to predict the subtle changes of air pollutants, and is easily affected by confounding factors. According to the Gaussian equation, the temporal and spatial exposure assessment of pollutant concentration is carried out by using pollutant emission, meteorological, and topographic data. It can combine the changes of air pollution in time and space without intensive monitoring. Hardware and software equipment is expensive and professional operators need to be trained; monitoring data need to be cross-checked with each other; temporary mismatch or...
Bayesian space-time mode (Szpiro et al., 2010) based on local geographical, meteorological, and station monitoring data, pollutant concentration in a certain area is divided into several time-space domains. When fully considering various possible uncertain factors, the indoor pollutant monitoring data of the outdoor concentration of each study object are obtained by simulation. Then, combined with the characteristics of the house and the permeability coefficient, the indoor concentration of the house is simulated, and according to the time ratio of indoor and outdoor activities, the exposure concentration of each object is measured by weight. Bayesian models have higher inference accuracy, acceptable spatiotemporal interaction, and over-discretization in small sample data, which overcomes the limitations of classical statistical methods, such as the needs of randomness, independence, uniform distribution, linearity, and isotropy. Excessive smooth processing, difficultly exploring complex spatiotemporal data information.
5. Summary and recommendations

This paper evaluates ground-level O$_3$ concentrations in major urban areas across China and reviews research findings on the health effects of O$_3$ exposure. This includes discussions on the pathogenic mechanisms of O$_3$ and the evaluation of O$_3$ exposure models. We propose the following conclusions and recommendations:

(1) Following the implementation of regulations aimed at curbing particulate matter and other forms of air pollution in 2013, China has seen a consistent increase in surface O$_3$ levels, which only began to decelerate in 2018. The available data unequivocally indicate that O$_3$ pollution is a serious issue in China's major cities, with concentrations in the Beijing-Tianjin-Hebei region, Yangtze River Delta, Pearl River Delta, and Weihe Plain surpassing those in other areas. China has initiated a monitoring system for O$_3$ and its precursors that can assist with O$_3$ pollution prevention and control measures. This system provides a comprehensive understanding of O$_3$ pollution across China. However, the quality control and assurance of O$_3$ and its precursors' observations need further enhancement to furnish robust scientific and technological support for O$_3$ pollution prevention and control efforts.

(2) O$_3$ pollution is a global health concern. As previously mentioned, O$_3$ concentrations are projected to rise in many parts of the world, potentially increasing ozone-related mortality and morbidity rates. Due to the current uncertainty surrounding the cardiovascular effects of O$_3$, the overall impact could be even more substantial when considering other potential effects of O$_3$ exposure. At present, there are many studies on
the relationship between O₃ and human health in the developed areas of China, but there are few studies focusing on the central and western regions, and the spatial representation is insufficient. Most of the O₃ exposure-response curves are complex linear, which makes the study of O₃ health effects very challenging. Moreover, studies often focus on populations, so it is difficult to accurately evaluate the level of individual O₃ exposure. There are variations in models and parameters used among studies, such that the results differ; therefore, choosing the correct exposure coefficient is difficult.

Future research should focus on the following aspects:

- Carrying out additional cohort studies in a large population
- Studying the effects of combined exposure of two or more pollutants on population health
- Carrying out research in different geographic areas to obtain sufficient spatial representation
- Determining the minimum human threshold for O₃ pollution, evaluating its effects on the human body, and further exploring its underlying mechanisms.

Declarations

Ethics approval and consent to participate Not applicable.

Consent for publication All authors consent for the publication of the manuscript and the materials incorporated.

Competing interests The author declares no competing interests.
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