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

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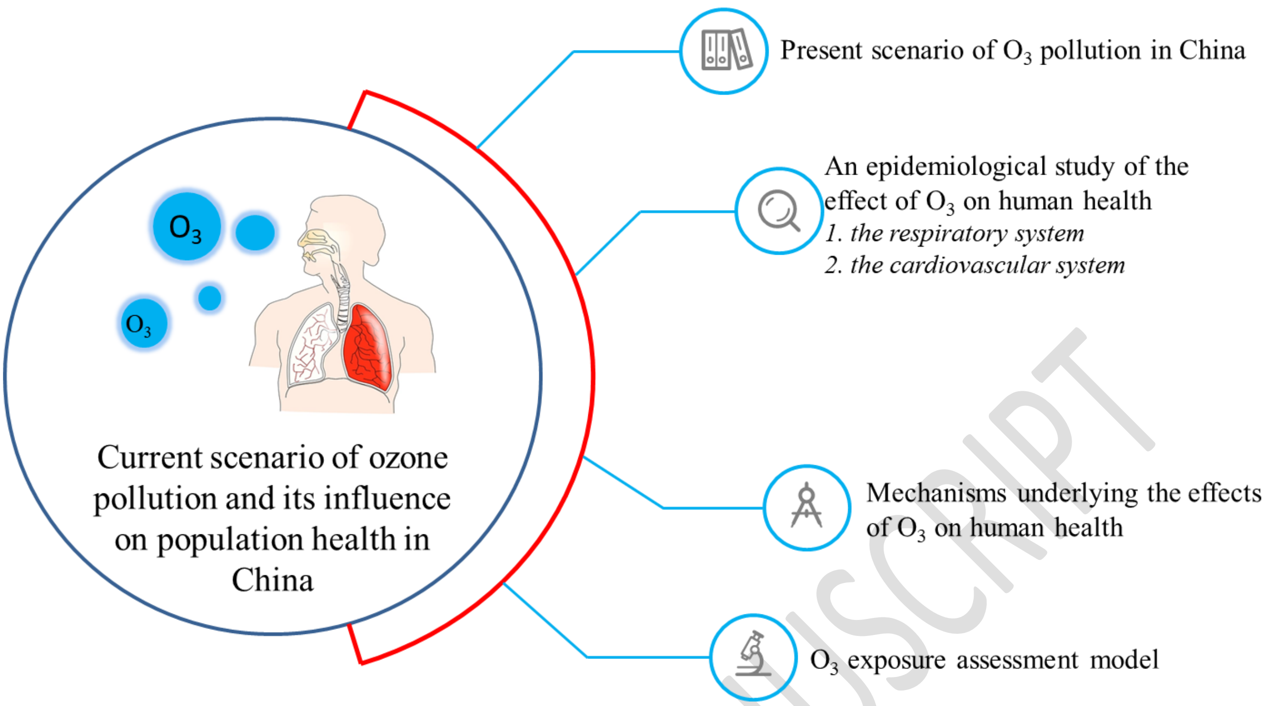
GRAPHICAL ABSTRACT



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 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 (O_3) is a common component of air pollutants and is produced by the photochemical reactions of volatile organic compounds (VOCs) and nitrogen oxides (NO_X) 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 O_3 pollution is becoming increasingly serious (Liu et al., 2015). Previous studies indicated that the distribution of O_3 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 O_3 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 O_3 concentrations exceeding the secondary limit of Environmental Air Quality Standards (GB3095-2012) (160 μg/m^3 [101.325 kPa, 20 ℃]), especially in the Beijing-Tianjin-Hebei and Yangtze River Delta regions (Wei et al., 2017; Zhang et al., 2015).

With rapid industrialization and urbanization, O_3 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 reviewO_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 O_3 and support the formulation of O_3 standards in China.

2. Current scenario of O_3 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 O_3 pollution is becoming increasingly serious. In recent years, O_3 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 O_3-8h (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 O_3-8h concentration and 90th percentile range in some Chinese cities from 2013 to 2021

Year	**Number of cities (number)**	**Average concentration of O_3-8h**	**Range of 90th percentile of O_3-8h concentration (μg/m^3)**	**Over-standard rate (%)**

		($\mu g/m^3$)		
2013	74	139	72–190	—
2014	161	140	69–210	6.1
2015	338	134	62–203	4.6
2016	338	138	73–200	5.2
2017	338	149	78–218	7.6
2018	338	151	76–217	8.4
2019	338	148	—	7.6
2020	338	138	—	4.9
2021	338	137	—	4.4

Note: —Statistical analysis of the 74 cities should refer to the year 2013. (The data come from the Ministry of Ecology and Environment of the People's Republic of China, available at https://www.mee.gov.cn).

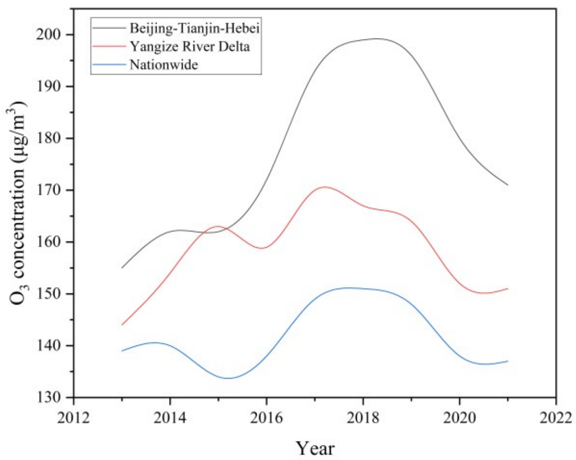


Figure 1. The changing trend of O_3 concentration in the key regions of China (Beijing-Tianjin-Hebei and Yangtze River Delta) and the entire country from 2013 to 2021

Overall, the O_3 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 O_3 in the Beijing-Tianjin-Hebei region was the highest, with an average annual concentration of 176.67μg/m^3, and the proportion of over- Environmental Air Quality Standard (GB3095-2012) for O_3 in the Pearl River Delta was the largest, with was 57.33%.

Table 2 O_3 pollution in the key regions since 2013

Year	Index	BTH	YRD	PRD	WP
2013	O_3 mean concentration (μg/m^3)	155	144	155	—
	O_3 exceedance ratio (%)	7.6	13.9	31.9	—
2014	O_3 mean concentration (μg/m^3)	162	154	156	—
	O_3 exceedance ratio (%)	—	—	—	—
2015	O_3 mean concentration (μg/m^3)	162	163	145	—
	O_3 exceedance ratio (%)	17.2	37.2	56.5	—
2016	O_3 mean concentration (μg/m^3)	172	159	151	—
	O_3 exceedance ratio (%)	26.3	39.8	70.3	—
2017	O_3 mean concentration (μg/m^3)	193	170	165	—
	O_3 exceedance ratio (%)	41.0	50.4	70.6	—
2018	O_3 mean concentration (μg/m^3)	199	167	—	180
	O_3 exceedance ratio (%)	46.0	49.3	—	36.4
2019	O_3 mean concentration (μg/m^3)	196	164	—	171
	O_3 exceedance ratio (%)	48.2	49.5	—	37.6
2020	O_3 mean concentration (μg/m^3)	180	152	—	161
	O_3 exceedance ratio (%)	46.6	50.7	—	36.1

2021	O_3 mean concentration (μg/m^3)	171	151	—	165
	O_3 exceedance ratio (%)	41.8	55.4	—	39.3

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

O_3 is primarily formed through the photochemical reactions of NO_X 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 O_3 in the economically developed eastern part of China is higher than in other regions (Peng et al., 2017). Kamal et al. (2019) reported that O_3 concentrations in China range from 74 to 201 μg/m^3. Approximately 30% of the population experiences O_3 levels exceeding 160 μg/m^3, and approximately 67.2% of the population is exposed to an O_3 environment greater than 100 μg/m^3 (Kamal et al., 2019).The comprehensive control and monitoring of O_3 in China began relatively late. Since 2012, the Ministry of Ecology and Environment of the People's Republic of China has been listing O_3 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 O_3 increased annually; after 2018, the over-standard rate of O_3 also improved significantly as the government issued a series of targeted control measures on atmospheric O_3 pollution.

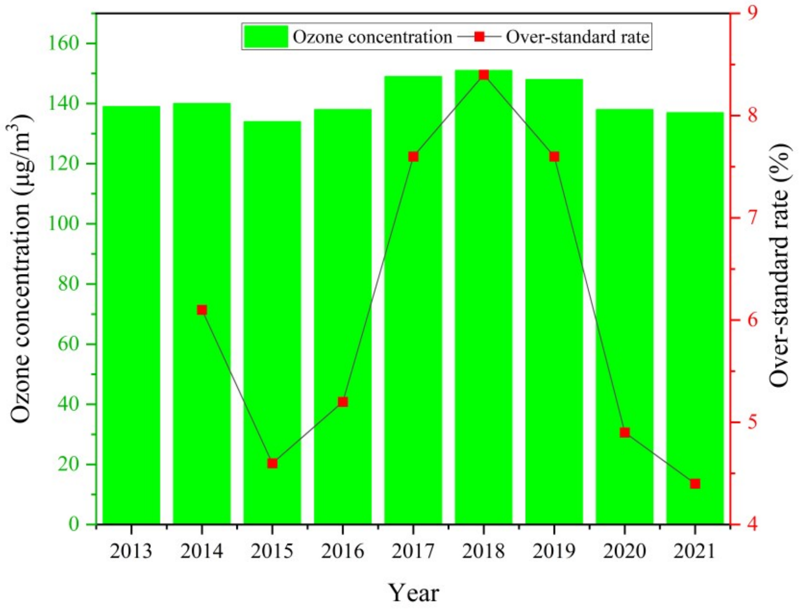


Figure 2. Ozone levels exceeding the standard rate in some Chinese cities, from 2013 to 2021 (Note in The data come from the Ministry of Ecology and Environment of the People's Republic of China, available at https://www.mee.gov.cn)

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 O_3 on the respiratory system exhibited lag and seasonal effects. Shang et al. (2013) conducted a meta-analysis, and the results showed that when the O_3 concentration increased by 10 μg/m^3, the mortality rate of respiratory diseases increased by 0.73% (95% CI: 0.49–0.97%) (Shang et al., 2013).

3.2. Effects of O_3 on the cardiovascular system

Cardiovascular disease poses one of the most important public health problems in our country. O_3 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 O_3 exposure and cardiovascular diseases (Liao et al., 2004; Park et al., 2005; Ruidavets & J.-B., 2005). It has also been reported that short-term O_3 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/m^3 increase in O_3 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 O_3 exposure has a greater impact on cardiovascular mortality in aging populations as compared to young ones, and the effect of O_3 is stronger in summer (Samoli et al., 2009). Studies carried out in the United States and China found different impacts of O_3 on the health of populations.

A study of 48 cities in the United States found that each 10 ppb increase in the concentration of O_3 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%

CI: 0.30–2.25%) (Jerrett et al., 2009). Li et al.'s research in Guangzhou showed that every 10 μg/m^3 increase in O_3 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, O_3 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 O_3 on human health

O_3 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 O_3 absorbed by humans is inhaled through respiratory tract, where it can cause injury. Studies have shown that O_3 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 O_3 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 O_3 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 O_3 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).

O_3 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 O_3 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 O_3 exposure can also lead to the destruction of spirochete DNA structures, induce DNA mutations, and inhibit DNA replication. Additionally, O_3 can induce systemic effects by regulating the activation of the neurohormonal stress response pathway.

3.4. Comparison of the effects of O_3 exposure on human health among countries worldwide

Globally, O_3 exposure is believed to have contributed to 250,000 premature deaths in 2015 (Lin et al., 2018). A study that examined O_3 levels and their health consequences in 50 cities in the eastern United States determined that higher O_3 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 O_3 levels due to climate change may result in an additional 50 premature deaths per year nationwide (Stowell et al., 2017). Future O_3 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 O_3 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 O_3 was identified as the main cause of reduced life expectancy, while short-term exposure to O_3 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 O_3 ranged between 1.58×10^{-7} and 4.71×10^{-7} years of life lost (Kassomenos et al., 2013). Surface O_3 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

Exposure model	**Principle**	**Advantages**	**Disadvantages**
Proximity model (Andersson et al., 2011)	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	Easy to operate and less expensive, and can also use geographic information systems	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.
Spatial interpolation model (Jin & Heap, 2011)	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	This model uses the monitoring points to obtain the temporal and spatial changes of pollutants	Because of the large consumption of funds, professional and technical personnel are needed, and the monitoring data have errors and

			great uncertainty.
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	Cannot effectively distinguish the influence of major pollutants, struggles to predict the subtle changes of air pollutants, and is easily affected by confounding factors
Atmospheric diffusion model (Sørensen et al., 2012)	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 networks, and has higher resolution	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

			misclassification of data may cause estimation offset
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 O_3 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 O_3 exposure-response curves are complex linear, which makes the study of O_3 health effects very challenging. Moreover, studies often focus on populations, so it is difficult to accurately evaluate the level of individual O_3 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 O_3 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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