Heliyon 9 (2023) e21803

Contents lists available at ScienceDirect

Heliyon

journal homepage: www.cell.com/heliyon

Impact of air pollution changes and meteorology on asthma outpatient visits in a megacity in North China Plain

Jing Ding [a,b], Suqin Han [a,b,*], Xiaojia Wang [a,b], Qing Yao [a,b]

[a] Tianjin Environmental Meteorological Center, Tianjin 300070, China
[b] CMA-NKU Cooperative Laboratory for Atmospheric Environment-Health Research, Tianjin 300070, China

ARTICLE INFO

Keywords:
Asthma
Outpatient visits
Air pollution
Meteorology
North China Plain

ABSTRACT

The effects of air pollution and meteorology on asthma is less studied in North China Plain. In the last decade, air quality in this region is markedly mitigated. This study compared the short-term effects of air pollutants on daily asthma outpatient visits (AOV) within different sex and age groups from 2014 to 2016 and 2017–2019 in Tianjin, with the application of distributed lag nonlinear model. Moreover, relative humidity (RH) and temperature as well as the synergistic impact with air pollutants were assessed. Air pollutants-associated risk with linear (different reference values were used) and non-linear assumptions were compared. In 2014–2016, PM_{10} and $PM_{2.5}$ exhibited a larger impact on AOV, with the corresponding cumulative excess risks (ER) for every 10 μg/m^3 increase at 1.04 % (95%CI:0.67–1.40 %, similarly hereafter) and 0.79 % (0.35–1.23 %), as well as increased to 43 % (26–63 %) and 20 % (10–31 %) at severe pollution. In 2017–2019, NO_2 and MDA8 O_3 exhibited a larger impact on AOV, with a cumulative ER for every 10 μg/m^3 increase at 1.0 (0.63–1.4 %) and 0.36 % (0.15–0.57 %), with corresponding values of 7.9 % (4.8–11 %) and 5.6 % (2.3–9.0 %), at severe pollution. SO_2 associated risk was only significant from 2014 to 2016. Cold effect, including extremely low temperature exposure and sharp temperature drop could generate a pronounced increase in AOV at 9.6 % (3.8–16 %) and 24 % (9.1–41 %), respectively. Moderate low temperature combined with air pollutants can enhance AOV during winter. Higher temperature in spring and autumn could trigger asthma by increasing pollen levels. Low RH resulted in AOV increase by 4.6 % (2.4–6.9), while higher RH generated AOV increase by 3.4 % (1.6–5.3). Females, children, and older adults tended to have a higher risk for air pollution, non-optimum temperature, and RH. As air pollution-associated risks on AOV tends to be weaker due to air quality improvement in recent years, the impact of extreme meteorological condition amidst climate change on asthma visits warrants further attention.

1. Introduction

Asthma was the second leading cause of death among chronic respiratory diseases in 2017 [1], seriously affecting the quality of life and imposing a great burden on families and society. Factors triggering asthma attack include infections, exercise, allergens, air pollution, meteorological factors, and emotional stress [2,3]. Of these, air pollution and meteorological factors can be predicted and monitored, allowing for the proactive formulation of asthma management strategies. Understanding the relationship between asthma

* Corresponding author. Tianjin Environmental Meteorological Center, Tianjin 300070, China.
E-mail address: sq_han@126.com (S. Han).

https://doi.org/10.1016/j.heliyon.2023.e21803
Received 18 July 2023; Received in revised form 8 September 2023; Accepted 28 October 2023
Available online 30 October 2023
2405-8440/

attack and air pollution and meteorological factors is the basis for development of asthma-protective health services.

Previous studies conducted in China and other countries reported an overall significant association between air pollutants, non-optimum meteorological factors (mainly temperature) and the asthma-related emergency, outpatient in inpatient visits [4–9]. In China, most related studies were conducted in southern cities, including Shanghai, Hong Kong, Chongqing, Hefei, Chongqing, Hefei, and Hangzhou [10–16] primarily because of the higher prevalence of asthma. However, inconsistent results are observed among these studies, and heterogeneity in estimated effects across these studies may be attributed to variations in air pollution profiles and the distinct asthma subgroups being studied. The North China Plain is suffering the most severe air pollution [17]. In the last decade, a series of control measures have been implemented to mitigate air pollution. As a result of strict emission reduction measures, the concentration of PM_{10}, $PM_{2.5}$, NO_2, SO_2, and CO exhibited a declining trend; conversely, a considerable increase has been observed in O_3 concentration [17–19]. The long-term impact of air pollution changes on asthma incidence remains poorly studied. Moreover, the increasing frequency of extreme meteorological conditions driven by climate change, has become a prominent concern in recent years. Specifically, various studies have investigated the influence of extreme cold and heat as well as diurnal temperature fluctuations, on the risk of asthma attack [20–25]. However, these studies showed inconsistent results, which could be attributed to variations in demographic and geographic regions. Meteorological conditions not only directly increase the risk of asthma attack but also indirectly affect the interaction of air pollutants. The synergistic effect of air pollution and non-optimum meteorological factors has been insufficiently explored [26,27].

The impact of air pollution and meteorology on asthma in North China Plain is poorly investigated, particularly in Tianjin [28], which is the largest industrial and the second megacity in North China Plain. In this study, the short-term effects of air pollutants on daily asthma outpatient visits (AOV) across different sex and age groups between two distinct periods: 2014–2016 (characterized by severe particulate pollution) and 2017–2019 (characterized by critical O_3 pollution), were compared to evaluate the impact of air pollution changes on AOV. Key meteorological factors affecting AOV were identified, and their synergistic impact with air pollution on AOV were also assessed. Results of our study are expected to provide scientific evidence for meteorological and environmental risk forecast or warning of asthma.

2. Data and methods

The daily AOV including emergency visits and general outpatients from seven hospitals ("AAA" grade) in Tianjin from 2014 to 2022 were collected. The data governance was conducted by the Tianjin Health Medical Big Data Co., LTD. Patients diagnosed with asthma using the International Classification of Disease, Revision 10 (ICD-10 codes J45-J46), and those with residential addresses in Tianjin, were included. Records with incomplete information were excluded. AOV was classified according to sex and three age groups: ≤14 years, 15–64 years and ≥65 years. Daily averaged PM_{10}, $PM_{2.5}$, NO_2, SO_2, CO concentrations and daily maximum 8-h average O_3 (MDA8 O_3) concentrations were obtained from the China National Environmental Monitoring Centre (https://air.cnemc.cn:18007). Meteorological factors including daily averaged temperature (T_{ave}), minimum temperature (T_{min}), maximum temperature (T_{max}), relative humidity (RH), wind speed (WD), air pressure (P) and precipitation were collected from 13 national meteorological station, and the average value of all stations was used. The AOV characteristics from 2020 to 2022 during the coronavirus disease 2019 (COVID-19) pandemic was different from the normal years due to the control policy, and thus were excluded in the AOV analysis.

Distributed Lag Nonlinear Model (DLNM) combined with Poisson Generalized Additive Model (GAM) [29] was applied to evaluate the associations between AOV and air pollutants and meteorological factors by adjusting for time-varying confounders in the data. The basic function Eq. (1) is as follows.

$$log[EY_i] = \alpha + cb.() + s(Z_i, df) + s(time, df) + Dow + Holiday \tag{1}$$

where, E (Y_i) is the daily AOV; α is the constant term; and *cb.* is cross-basis function for building the basis for predictor and lags. In previous studies, the choice of maximum lag days as the basis for the lag space varied; some studies used 7–10 d [30–32] while others extended it to 14 d [22] or even 21 d [33]. Han et al. reported that the lag effect of extreme low temperature on asthma could last 3–30 d [34]; therefore, we specify the lagged effect of air pollutants and meteorological factors up to 14 d (i.e., middle value of the reported values, minimum lag equal to 0 by default), with a 3rd degree polynomial function. Single pollutant models were used to appraise the air pollutants-associated risk on AOV. The different reference value was considered, including the default value zero and class 2 limit values of the National Ambient Air Quality Standard (GB 3095-2012), as well as the Air Quality Guidelines (AQG 2021) published by World Health Organization (WHO); when the air pollutant exceeded the reference value, a linear relationship was considered in their effect on AOV. According to previous studies [31,35,36], non-linear assumption with a natural cubic spline with 3 degrees of freedom was also applied for air pollutant, and the results were compared. While modelling the relationship with meteorological factors through a natural cubic spline with 3 degrees of freedom, s (Z_i, df) represented the controlled confounding variable that has a nonlinear relationship with the AOV, and was set as natural cubic spline with 3 degrees of freedom. Additionally, the s ($time$, df) was utilized to control long-term trend and seasonality with 7 degrees of freedom per year. *DOW* refers to the day of week (1, 2, 3, …,7). *Holiday* is the classification variable to adjust for the effect of public holiday (1 for holiday and 0 for non-holiday).

On the basis of Eq. (1), tensor product smoothing function, ***ti***, was introduced to establish the interaction term between temperature and air pollutants; the basic function is as Eq. (2) follows. The synergistic effect of low temperature and $PM_{2.5}$, as well as high temperature and MDA8 O_3 was observed on AOV. Given the prolonged time of air pollution, the moving averaged concentration and $PM_{2.5}$, MDA8 O_3 as well as the temperature was incorporated into the function. According to our previous statistics of prolonged time of haze pollution and O_3 pollution, 3 d averaged $PM_{2.5}$ concentration and 5 d averaged MDA8 O_3 concentration were applied.

$$log[EY_i] = \alpha + t_i(T, X) + s(Z_i, df) + s(time, df) + Dow + Holiday \quad (2)$$

The DLNM package in R 4.2.3 software was used for fitting, and the relative risk ratio (RR) and excess risk (ER) were used to evaluate the impact of air pollution and extreme meteorological condition on AOV. Excess risk for every 10 μg/m^3 increase and severe polluted concentration (approximately 99 % of the concentration) for air pollutants (1 mg/m^3 increase for CO), and extreme meteorological condition (99 % value of each meteorological factor) in "best" lagging days on AOV was calculated. The 95 % confidence interval (95%CI) was used to determine whether the difference was statistically significant.

3. Results

3.1. Characteristics of air pollution, meteorology, and AOV

Table 1 summarizes the basic statistics in this study. There were 279,879 AOVs recorded in 2014–2019, and the annual total AOV showed an upward trend. The male to female ratio was 0.94:1. The proportion of children (≤14 years), adolescents and younger adults (15–64 years), and older adults (≥65 years) was 5 %, 73 % and 22 %, respectively (Fig. 1(a)). The PM_{10}, $PM_{2.5}$, SO_2 and CO concentrations in 2017–2019 significantly decreased compared to that in 2014–2016; meanwhile, NO_2 showed a slight decline. The MDA8 O_3 concentration markedly increased in 2017–2019. The annual variation for each air pollutant in Tianjin is shown in Fig. S1. The averaged T_{ave}, T_{max}, and T_{min} in 2017–2019 was close to that in 2014–2016; however, the extreme T_{max} was 2 °C higher while extreme T_{min} was 1 °C lower in 2017–2019 than in 2014–2016. The overall RH in 2017–2019 was lower than that in 2014–2016.

High AOV in Tianjin occurred in April and September to December (Fig. 1(b)). Notably, the elevated AOV in April and September were associated with pollen allergy. According to the pollen observation data in Tianjin from 2009 to 2018 (Fig. S2), high pollen concentration in spring occurred in late March to late April, with two peak period: from 7 to 16 April and from 22 to 26 April. In autumn, high pollen concentration generally occurred in late August towards the end of September, and the peak periods were from 24 August to 7 September and from 14 to 20 September. Strong positive correlation coefficient of 0.42 ($P < 0.001$), 0.59 ($P < 0.001$) and 0.55 ($P < 0.001$) was observed during the pollen season in 2014, 2016, and 2018, respectively.

3.2. Comparison of AOV risk induced by air pollutants in 2014–2016 and 2017–2019

The estimation of ER and lag effect for every 10 μg/m^3 increase in $PM_{2.5}$, PM_{10}, SO_2, NO_2, and MDA8 O_3 as well as every 1 mg/m^3 increase in CO on daily AOV with different assumptions in 2014–2016 and 2017–2019 are shown in Figs. S3–S8 and Table 2. Overall, the largest lag days were similar with different linear assumption. In 2014–2016, the strongest cumulative effects of CO, MDA8 O_3, NO_2, SO_2, PM_{10}, $PM_{2.5}$ on daily AOV occurred at lag1-2, lag1-2, lag1, lag1-4, lag0-10, and lag0-8, respectively. For the "best" lag time, the ER on daily AOV for each air pollutant was different depending on the reference value. The ER on daily AOV for SO_2, CO, and MDA8 O_3 was higher than other air pollutants if GB 3095-2012 and AQG 2021 limits were set as the reference; meanwhile, the ER for each air pollutant was close when the reference was zero. For severe pollution condition, PM_{10} and $PM_{2.5}$ exhibited the most impact on AOV, regardless of the reference value. In particular, the ER was the highest with non-linear assumption, followed by the GB 3095–2012 limits, while the ER with AQG 2021 limits and zero was lower and considerably close. The ER of SO_2 was the highest with AQG 2021 limit, while that of NO_2 and MDA8 O_3 was the highest with zero as the reference. CO associated risk on AOV was determined

Table 1
Summary statistics of daily AOV, air pollutants concentrations and key meteorological factors in Tianjin in 2014–2016 and 2017–2019.

	Mean	SD	P (1)	P (25)	P (50)	P (75)	P (99)
Daily AOV							
Age							
≤14	1 (7)	1 (6)	0 (0)	0 (3)	1 (6)	2 (10)	6 (25)
15-64	84 (108)	30 (34)	25 (31)	64 (85)	81 (107)	103 (130)	165 (196)
≥65	20 (32)	9 (14)	4 (7)	14 (22)	20 (32)	26 (41)	41 (67)
Sex							
Female	55 (75)	21 (26)	13 (21)	40 (57)	53 (74)	67 (91)	112 (146)
Male	52 (72)	18 (23)	13 (20)	39 (58)	51 (73)	64 (86)	96 (128)
Air pollutants concentration							
PM_{10} (μg/m^3)	123 (91)	74 (57)	28 (25)	71 (56)	104 (77)	154 (112)	487 (405)
$PM_{2.5}$ (μg/m^3)	76 (55)	54 (41)	15 (11)	38 (28)	61 (45)	96 (68)	290 (228)
SO_2 (μg/m^3)	34 (13)	32 (9)	6 (4)	13 (8)	23 (11)	42 (16)	165 (43)
NO_2 (μg/m^3)	48 (46)	23 (20)	14 (17)	31 (30)	44 (41)	60 (58)	120 (99)
MDA8 O_3 (μg/m^3)	81 (105)	49 (62)	8 (12)	42 (56)	67 (90)	114 (148)	208 (251)
CO (mg/m^3)	1.5 (1.1)	0.8 (0.6)	0.5 (0.4)	1.0 (0.8)	1.3 (1.0)	1.8 (1.3)	4.4 (3.0)
Key meteorological factors							
T_{ave} (°C)	14 (14)	11 (11)	−6 (−7)	3 (3)	16 (15)	24 (25)	30 (32)
T_{max} (°C)	19 (20)	11 (12)	−2 (−3)	8 (9)	22 (22)	29 (30)	36 (38)
T_{min} (°C)	9 (9)	11 (11)	−10 (−11)	−1 (−1)	11 (10)	19 (20)	26 (28)
RH (%)	58 (55)	18 (20)	19 (19)	45 (40)	59 (55)	72 (71)	93 (96)

Data in the bracket was in 2017–2019.

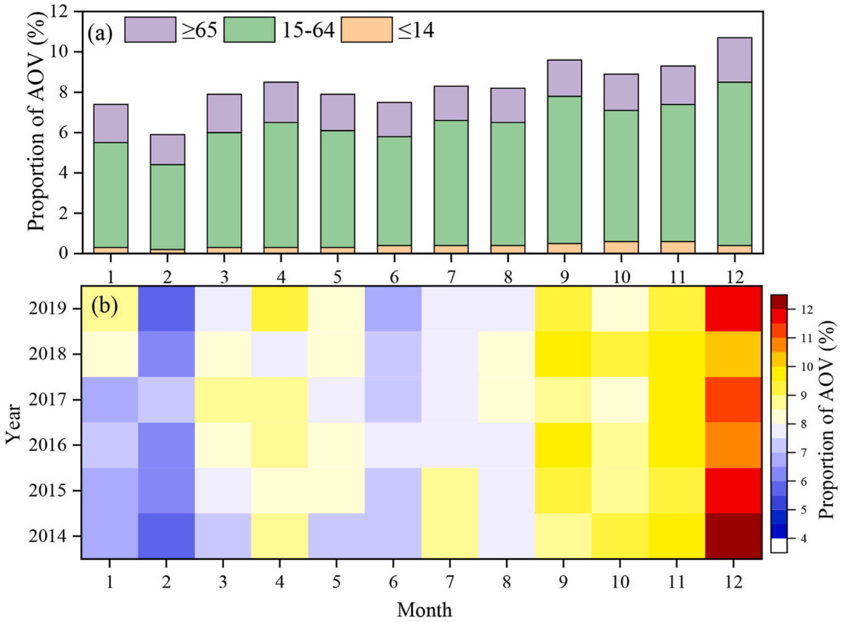

Fig. 1. (a) Monthly mean changes in the proportion of AOV for children, adolescents and younger adults as well as older adults, (b) annual monthly changes of total asthma outpatient visits in Tianjin.

Table 2
Contrastive effect of air pollutants on daily AOV with different assumptions in 2014–2016 and 2017–2019.

Year	Variable	Linear assumption			Non-linear assumption
		GB 3095-2012	AQG 2021	0	
2014–2016	PM_{10} (10 μg/m^3 increase)	1.0 (0.67, 1.40)	0.37 (0.14, 0.59)	0.34 (0.12, 0.57)	–
	$PM_{2.5}$ (10 μg/m^3 increase)	0.79 (0.35, 1.2)	0.50 (0.26, 0.74)	0.50 (0.26, 0.74)	–
	NO_2 (10 μg/m^3 increase)	1.0 (0.40, 1.6)	0.41 (−0.04, 0.85)	0.49 (0.063, 0.92)	–
	SO_2 (10 μg/m^3 increase)	6.4 (2.3, 11)	1.3 (0.58, 2.0)	0.42 (−0.081, 0.92)	–
	MDA8 O_3 (10 μg/m^3 increase)	2.6 (1.1, 4.2)	0.73 (0.31, 1.2)	0.40 (0.10, 0.71)	–
	CO (1 mg/m^3 increase)	2.6 (−0.92, 6.2)	2.6 (-0.92, 6.2)	0.84 (−0.21, 1.9)	
	PM_{10} (500 μg/m^3)	43 (26, 63)	18 (6.6, 31)	19 (6.2, 33)	82 (49, 122)
	$PM_{2.5}$ (300 μg/m^3)	20 (10, 31)	15 (7.7, 24)	16 (8.1, 25)	52 (37, 69)
	NO_2 (120 μg/m^3)	4.1 (1.6, 6.7)	3.9 (−0.32, 8.4)	6.1 (0.72, 12)	11 (6.7, 16)
	SO_2 (165 μg/m^3)	9.8 (3.5, 16)	17.4 (7.5, 28)	7.1 (−1.4, 16)	#
	MDA8 O_3 (200 μg/m^3)	6.8 (2.9, 11)	7.6 (3.2, 12)	8.4 (2.1, 15)	6.1 (2.2, 10)
	CO(5 mg/m^3)	2.6 (−0.92, 6.2)	2.6 (−0.92, 6.2)	4.3 (−1.1, 9.9)	4.9 (2.6, 7.3)
2017–2019	PM_{10} (10 μg/m^3 increase)	0.21 (0.032, 0.39)	0.14 (0.021, 0.25)	0.17 (0.022, 0.25)	–
	$PM_{2.5}$ (10 μg/m^3 increase)	0.38 (0.14, 0.63)	0.28 (0.10, 0.46)	0.28 (0.10, 0.46)	–
	NO_2 (10 μg/m^3 increase)	1.6 (0.11, 3.1)	1.0 (0.63, 1.4)	1.1 (0.25, 1.5)	–
	MDA8 O_3 (10 μg/m^3 increase)	0.45 (−0.16, 1.1)	0.36 (0.15, 0.57)	0.29 (0.080, 0.50)	–
	CO (1 mg/m^3 increase)	/	/	3.1 (1.5, 4.8)	–
	PM_{10} (400 μg/m^3)	5.4 (0.83, 10)	4.9 (0.73, 9.3)	7.1 (2.3, 12)	13 (4.1, 23)
	$PM_{2.5}$ (200 μg/m^3)	4.9 (1.7, 8.1)	5.3 (1.8, 8.9)	5.8 (2.1, 9.7)	2.6 (0.98, 4.3)
	NO_2 (100 μg/m^3)	3.2 (0.23, 6.3)	7.9 (4.8, 11)	12 (2.6, 17)	14 (9.3, 18)
	MDA8 O_3 (250 μg/m^3)	2.3 (−0.81, 5.4)	5.6 (2.3, 9.0)	7.4 (2.0, 13)	2.1 (−1.3, 5.6)
	CO (3 mg/m^3)	/	/	9.7 (4.5, 15)	8.4 (6.5, 10)

Excess risk was utilized in table and two significant digits were kept. CO concentration in 2017–2019 was lower than the GB 3095-2012 and QG 2021 limit 4 mg/m^3, the associated risk was not calculated, and marked as "/". Non-linear assumption for air pollutants can not be calculated for 10 μg/m^3 increase associated excess risk, marked as "—". "#" represents excess risk was lower than 0.

to be highly uncertain except for the non-linear assumption.

Compared to 2014–2016, under the same assumption, the PM_{10} and $PM_{2.5}$ associated risks in AOV dropped significantly in 2017–2019 (Table 2), and no discernible lag impact was observed (Figs. S6–S8). Conversely, the NO_2 associated risk markedly increased, and the SO_2 associated ER was lower than zero. Contrary to expectations, the MDA8 O_3 associated risks were reduced. According to the ER, every 10 μg/m^3 increase in NO_2 and MDA8 O_3 elicited a larger impact on AOV than $PM_{2.5}$ and PM_{10} in all assumptions. However, for severe pollution, key air pollutants affecting asthma attack changed depending on the reference values. The ER of NO_2 and MDA8 O_3 using AQG 2021 limit and zero as references had a greater impact on AOV than $PM_{2.5}$ and PM_{10}, whereas PM_{10}

and $PM_{2.5}$ remained the crucial pollutants affecting AOV using the GB 3095-2012 limit as reference.

In the stratified analysis (Tables 3 and 4), females were more sensitive to PM_{10}, $PM_{2.5}$, and NO_2, while males were more sensitive to SO_2 and O_3. Among the three age groups, children were more susceptible to PM_{10}, $PM_{2.5}$, and NO_2, while older adults were more vulnerable to SO_2. In 2014–2016, O_3 associated risk on adolescents and younger adults were relatively greater whereas children were most sensitive to O_3 in 2017–2019.

3.3. Impact of meteorology on AOV

The linear correlation between AOV and meteorological factors was analyzed by year. Among the meteorological factors with statistical differences, AOV were positively correlated with air pressure, while negatively correlated with air temperature in all years. Specifically, AOV has a highest correlation coefficient with 2 d averaged T_{min}. Moreover, the negative temperature change has an important effect on AOV. AOV were positively correlated with RH, and the effects of RH change on AOV in some years were significant. In addition, AOV were negatively correlated with precipitation, which may be attributed to the reduced necessary outing and air pollution or other allergen. The result in 2019 is shown in Table S1. Based on the meteorological factors that significantly affected AOV over the years and the meteorological forecasting factors for the public, temperature and RH were identified as the key meteorological factors affecting AOV.

The exposure-lag-response and cumulative effect between AOV and temperature and RH are shown in Figs. 2 and 3. The data in 2014–2019 were used since the magnitude of meteorological factors did not change dramatically (Table 1). A statistically significant relationships between AOV and temperature were observed at lag 1–4 d. Specifically, cold effect was remarkably significant. After the extremely low temperature exposure ($T_{min} = -10$ °C), the AOV increased by 9.6 % (3.8–16 %) at lag1-2 d (Table 5). Hot effect on AOV was complex; although the T_{ave} exceeded 30 °C in summer, the ER remained lower than 1. Inter-day temperature variability also exhibits a strong impact with AOV increase. As shown in Fig. 2b, the negative 24 h T_{ave} change above 7 °C at lag 1–4 d, significantly amplifying AOV. A 10 °C drop at lag 1–4 d cumulatively generated 24 % (9.1–41 %) increase in AOV. Specifically, temperature drops (above 5 °C) following heat days in summer played a more important role (Fig. S9(a)) than high temperature. Moreover, temperature rise slightly increased AOV, with a 5 °C raise at lag 0–2 d cumulatively evoking a 5.1 % (0.86–9.6 %) increase in AOV. In spring, the ER was markedly higher than 1 when 72 h T_{max} change was larger than 10 °C, and T_{max} was higher than 20 °C; temperature rise could facilitate bloom and the subsequent pollen spread. Both high and low RH can facilitate increase in risk, low RH generated the effect at lag0–2d, while high RH exerted the effect at lag 2–4d. Lower RH (RH = 20 %) posed a greater risk in AOV than higher RH (RH = 90 %) for adolescents and younger adults, while higher RH played a more important role among older adults with 12 % (6.8–18 %) increase in AOV at “best” lag time. The lag of wet effect may cover the impact of transition from high to low RH (Fig. S3b), and the ER was approximately 3–4% when RH decreased to 40 % from 80 to 90 %. In the stratified analysis, females were more sensitive to cold and wet effects, while males were more sensitive to dry effect. Children were susceptible to cold effect, while older adults were more vulnerable to both dry and wet effects.

3.4. Synergistic impact of meteorology and air pollution on AOV

In North China Plain, haze pollution was most severe during the winter. $PM_{2.5}$ combined a moderately low temperature can increase AOC, showing a positive synergistic impact (Fig. 4a). Other pollutants except for O_3, presented a co-variation with $PM_{2.5}$ under the same meteorological condition. Therefore the RR distribution with combination of low temperatures and PM_{10}, NO_2, SO_2 as well as CO were similar to that of $PM_{2.5}$ (Fig. S10). On average, $PM_{2.5}$ strengthened ER by 0.4 % in AOV with $T_{min} = -5$ °C at lag1-2 d (Table 4), indicating that low temperature dominated the synergistic effect. Pronounced high ER occurred in condition that T_{min}lag0-2 was within −8 to 0 °C, and 3 d averaged $PM_{2.5}$ concentration reached above 120 μg/m^3 (Fig. 4). In particular, when 3 d averaged $PM_{2.5}$ concentration exceeded 300 μg/m^3, the highest ER was 27 %. During a severe haze pollution that occurred in December 2015, there were five days with a daily averaged $PM_{2.5}$ concentration exceeding 200 μg/m^3, and the mean $PM_{2.5}$ concentration was 277 μg/m^3. Consequently, a 20 % increase in AOV was observed compared to the week before this event occurred. Sensitivity analyses were

Table 3
Effect of air pollutants on daily AOV in Tianjin in 2014–2016.

2014–2016	Gender		Age		
Variable	Male	Female	≤14	15–64	≥65
PM_{10} (10 μg/m^3 increase)	0.90 (0.38, 1.4)	1.2 (0.72, 1.7)	2.4 (−0.64, 5.4)	1.0 (0.63, 1.5)	1.4 (0.69, 2.2)
$PM_{2.5}$ (10 μg/m^3 increase)	0.78 (0.29, 1.3)	0.95 (0.41, 1.5)	2.4 (0.42, 4.5)	0.78 (0.46, 1.1)	1.6 (0.84, 2.4)
$NO_2^§$(10 μg/m^3 increase)	0.27 (−0.47, 1.0)	0.60 (−0.021, 1.2)	13 (4.1, 22)	0.36 (−0.22, 0.94)	1.2 (0.33, 2.1)
$SO_2^§$ (10 μg/m^3 increase)	1.8 (0.82, 2.9)	0.93 (0.062, 1.8)	1.0 (−3.9, 6.2)	1.5 (0.63, 2.3)	1.6 (−0.17, 3.5)
MDA8 $O_3^§$(10 μg/m^3 increase)	1.5 (0.39, 2.7)	0.96 (0.38, 1.6)	0.89 (−2.3, 4.2)	1.0 (0.56, 1.5)	0.49 (−0.37, 1.3)
PM_{10} (500 μg/m^3)	37 (14, 64)	54 (29, 83)	125 (−21, 534)	14 (25, 66)	65 (27, 115)
$PM_{2.5}$ (300 μg/m^3)	19 (6.7, 33)	24 (9.6, 39)	72 (9.9, 168)	19 (11, 28)	43 (21, 70)
NO_2 (120 μg/m^3)§	3.7 (0.14, 7.4)	4.7 (1.2, 8.3)	79 (0.34, 220)	3.8 (1.0, 6.7)	18 (0.57, 38)
SO_2 (165 μg/m^3)§	26 (11, 43)	12 (0.74, 25)	14 (−39, 112)	20 (8.2, 33)	23 (−2.1, 54)
MDA8 O_3 (200 μg/m^3)§	16 (3.9, 30)	10 (3.8, 17)	9.3 (−21, 51)	11 (5.7, 16)	4.9 (−3.7, 14)
CO (5 mg/m^3)	1.5 (−3.4, 6.8)	3.6 (−1.2, 8.7)	13 (−17, 54)	0.63 (−3.2, 4.6)	10 (−2.1, 19)

Table 4
Effect of air pollutants on daily AOV in Tianjin in 2017–2019.

2017–2019	Gender		Age		
Variable	Male	Female	≤14	15–64	≥65
PM_{10} (10 μg/m^3 increase)	0.10 (−0.15, 0.35)	0.29 (0.061, 0.53)	0.23 (−0.81, 1.27)	0.20 (0.00, 0.40)	0.14 (−0.23, 0.50)
$PM_{2.5}$ (10 μg/m^3 increase)	0.13 (−0.22, 0.49)	0.62 (0.27, 0.95)	0.72 (−0.45, 1.91)	0.28 (0.00, 0.56)	0.58 (0.070, 1.1)
$NO_2^§$(10 μg/m^3 increase)	0.82 (0.27, 1.4)	1.2 (0.70, 1.8)	1.6 (−0.13, 3.4)	0.88 (0.43, 1.3)	1.7 (0.83, 2.5)
MDA8 $O_3^§$(10 μg/m^3 increase)	0.37 (0.073, 0.67)	0.34 (0.052, 0.64)	1.0 (−0.13, 3.4)	0.35 (0.43, 1.3)	0.16 (−0.30, 0.62)
PM_{10} (400 μg/m^3)	2.6 (−3.5, 9.2)	7.6 (1.5, 14)	5.8 (−18, 37)	5.1 (0.061, 10)	3.5 (−5.4, 13)
$PM_{2.5}$ (200 μg/m^3)	1.7 (2.7, 6.3)	8.1 (3.4, 13)	9.4 (−5.4, 27)	3.5 (−0.12, 7.3)	7.6 (0.86, 15)
NO_2 (100 μg/m^3)§	6.3 (2.0, 11)	9.7 (5.4, 14)	13 (−0.94, 29)	6.8 (3.3, 10)	13 (6.4, 20)
MDA8 O_3 (250 μg/m^3)§	5.6 (1.0, 10)	5.3 (0.70, 10)	17 (1.2, 35)	5.4 (1.6, 9.4)	2.4 (−4.3, 9.7)

Excess risk for NO_2, SO_2 and MDA8 O_3 in AOV was the result with the reference of AQG 2021 limits, marked as "§", its validity was described in Discussion section. Two significant digits were kept in table.

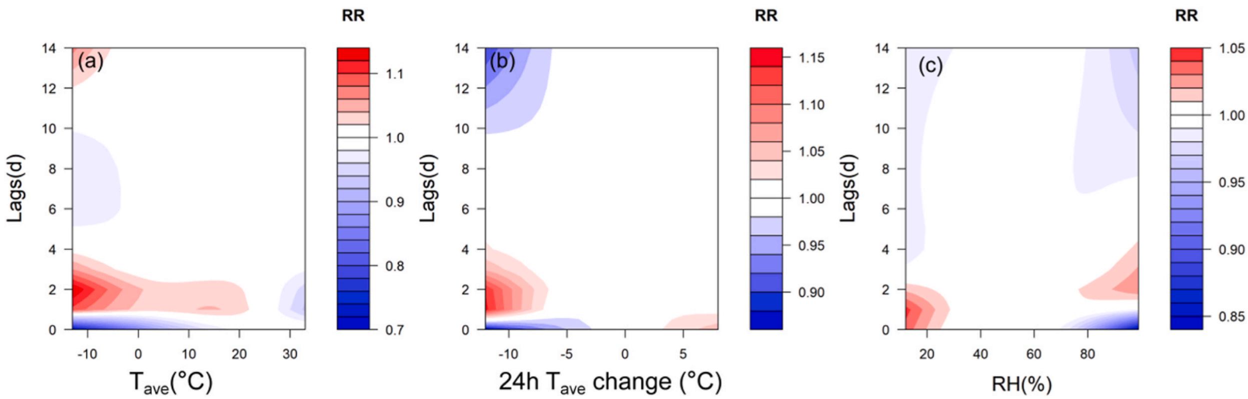

Fig. 2. The exposure-lag-response of AOV to (a) daily averaged T_{ave}, (b) 24 h T_{ave} change and (c) daily averaged RH in 2014–2019 in Tianjin.

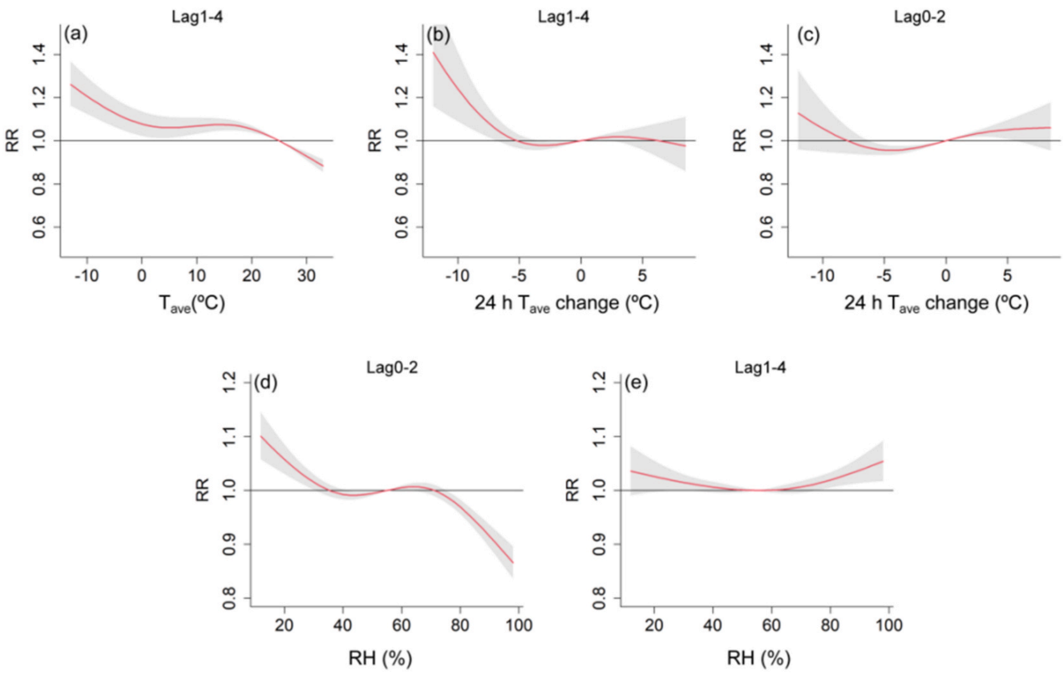

Fig. 3. The cumulative impact of (a) daily averaged temperature T_{ave}, (b) (c) 24 h T_{ave} change and (d) (e) daily averaged relative humidity on AOV in 2014–2019 in Tianjin.

Table 5
Effect of key meteorological variables on daily AOV in Tianjin in 2014–2019.

		Cold effect (T_{min} = −10 °C)	Cold effect (T_{min} = −5 °C)	Cold & $PM_{2.5}$ (T_{min} = −5 °C)	Dry effect (RH = 20 %)	Wet effect (RH = 90 %)
All		**9.6 (3.8, 16)**	**5.9 (1.0, 11)**	**6.3 (1.3, 12)**	**4.6 (2.4, 6.9)**	**3.4 (1.6, 5.3)**
Gender	Male	7.4 (−0.6, 16)	4.2 (−2.6, 12)	5.3 (−1.8, 13)	**6.7 (3.5, 10)**	2.5 (−0.10, 5.2)
	Female	**12 (3.7, 21)**	**7.7 (0.7, 15)**	**7.4 (0.42, 15)**	2.7 (−0.38, 5.8)	**4.4 (1.8, 7.0)**
Age	≤14	22 (−8.6, 64)	7.1 (−16, 36)	8.9 (−15, 39)	5.3 (−6.9, 19)	6.0 (−4.5, 18)
	15–64	**9.0 (2.3, 16)**	**6.0 (0.31, 12)**	**6.2 (0.44, 12)**	**4.0 (1.4, 6.6)**	**2.4 (0.00, 4.9)**
	≥65	10 (−2.1, 24)	6.3 (−4.2, 18)	7.3 (−3.5, 19)	**7.1 (2.3, 12)**	**12 (6.8, 18)**

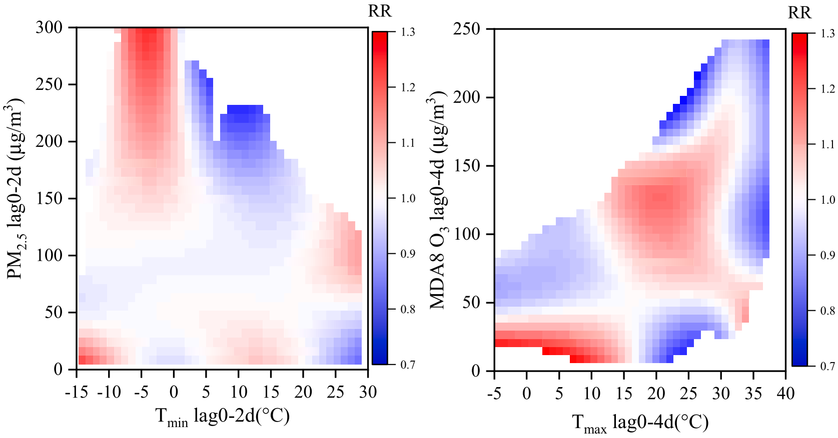

Fig. 4. Synergistic impact of (a) T_{min} and $PM_{2.5}$ concentration at lag0-2 d and (b) T_{max} and MDA8 O_3 concentration at lag0-4 d on AOV in 2014–2019 in Tianjin.

conducted to examine the robustness of associations for low temperatures by adjusting for the impacts of other air pollutants. In the sensitivity analyses, the cumulative ER for moderately low temperature (-5 °C) were slightly attenuated but did not change substantially after adjusting for SO_2, CO, and MDA8 O_3, while moderately increased after adjusting for PM_{10}; a significantly decrease was observed after adjusting for NO_2 (Table S3). In summer, synergistic impact on AOV induced by O_3 and high temperature was not observed (Fig. 4b). In the sensitivity analyses, the cumulative ER for high temperature (35 °C) was not statistically different after adjusting for other air pollutants (Table S4).

4. Discussion

Air pollution engendered an apparent impact on AOV. However, our study demonstrated that the assumption for the effect of air pollutants on AOV (linear or non-linear) could lead to a significant difference on ER. Most previous study used linear assumption to investigate the effect of every 10 μg/m³ increase in each air pollutant on disease incidence [5,35,37], while non-linear assumption was also adapted in few studies [36]. With linear assumption, different reference value affect the evaluation of ER. In most previous studies, the reference values were not mentioned, and the default value of zero may be used. Our results demonstrated that air pollution level and the distance to the reference values are two important factors affecting ER evaluation. For both 2014–2016 and 2017–2019, the PM_{10} and $PM_{2.5}$ concentrations were markedly higher than the AQG 2021 limit. During winter in Northern China, a seesaw effect exists between the cold and haze pollution, and lower particulate concentrations were typically connected with cold front, whereas the occurrence of haze was generally linked to temperature rise and calm wind [38]. Therefore, considering the greater cold effect on AOV when particulate concentration was markedly low, the reference value of pollutants does not necessarily follow the logic that lower is better; opting for AQG 2021 limit or 0 as reference may underestimate the ER. It is recommended to use the class 2 limits in GB 3095–2012 as the reference value for PM_{10} and $PM_{2.5}$. In contrast, AQG 2021 limits of SO_2, NO_2, and MDA8 O_3 were close to their corresponding 75th, 25th, 50th concentrations and were more suitable as reference values than the GB 3095–2012.

This study showed statistically significant associations between short-term exposure across all six air pollutants and AOC. In terms of the ER for every 10 μg/m³ increase in air pollutants, gaseous air pollutants have larger effects on AOV compared to the particulate, and similar results were observed in other epidemiological studies [5,11,12,31]. SO_2 can cause irritation and mainly comes from coal combustion. Previous studies have shown that ambient SO_2 was statistically associated with outpatient visits and mortality due to respiratory diseases [39,40]. In China, power plants were initially mandated to perform SO_2 ultra-low emission since 2015, with steel enterprises following suit. Presently, the annual SO_2 concentration in Tianjin is < 10 μg/m³, and the associated risk on AOV was only significant in 2014–2016. Outdoor NO_2 mainly comes from traffic sources and indoor NO_2 occurs in conjunction with the use of cooking gas, which can induce airways inflammation [41]. Notably, NO_2 may be a tracer of other traffic related pollutants, which may play a more important role in the respiratory diseases [11,42]. As a product of photochemical reaction, O_3 is associated with asthma incidence and aggravation [43]; however, we found that although O_3 pollution was aggravated from 2017 to 2019, the associated risk was lower than that during 2014–2016. The decreased risk may be attributed to less exposure induced by improved protection. The outpatient data indicated that during the peak season (June to August) of O_3 pollution, the proportion of AOV in 2017–2019 was lower than that in 2014–2016. Although the ER for every 10 μg/m^3 increase in PM_{10}, and $PM_{2.5}$ was lower for severe polluted condition, the effect of sandstorm and haze on asthma was horrendous, especially during 2014–2016. Particulate matter is an ideal carrier for toxic and harmful substances in the atmosphere. Previous studies found that increase in BC, NO_3^-, and K^+ were associated with the increased risk of AOV [11,44]. Due to the implementation of vigorous emission reduction measures since 2013, the concentration of air pollutants (except O_3) displayed a significant drop and air pollution sources shifted from coal combustion driving to mixed sources. Air pollutants associated risks on AOV tends to be weaker over the years, especially for SO_2, PM_{10} and $PM_{2.5}$. Currently, the effect of NO_2 and O_3 on AOV was relatively prominent. The further reduction in NO_2 remains challenging due to the increasing vehicle number. As NO_2 is a crucial precursor of O_3, it will also impede efforts to lower O_3 concentration.

We estimated that cold, sharp temperature drop, dry and wet effects had a significant impact on AOV in Tianjin. AOV increased by 9.6 % (3.8–16 %) at lag1-2 d after the extremely low temperature exposure ($T_{min} = -10$ °C), and negative inter-day temperature change pose a markedly higher risk on AOV raise than positive temperature variability. A daily averaged 10 °C drop could cumulatively generate 24 % (9.1–41 %) increase in AOV. In summer, temperature drops following heat days can lead to a more important impact than extreme hot effect itself, which in essence, represents a cooling effect. Zhu et al. found that greater diurnal peak expiratory flow variation was associated with sudden temperature drop, which may exacerbate asthma risk [45]. Previous studies have shown that cold effects on asthma attack were relatively consistent, and were considerably more significant than hot effect [34]. The cold effect could last for almost 3 days, and higher decrease in pulmonary function was associated with low than high temperature [46]. The associations between extreme hot temperature and respiratory hospital visits were also not significant in the neighbouring cities, Beijing [22] and Shijiazhuang [47]. Xiong et al. highlighted that influenza-related asthma exacerbation was common during warm and hot seasons [48]. Nevertheless, we found that temperature rise in spring could increase the levels of air allergens (e.g., pollen), and thus triggering asthma. Currently, climate variations are predicted to cause longer pollen seasons as well as influence start, duration, and intensity of the pollen season. Humidity is another important factor influencing AOV. In this study, lower (20 %) and higher (90 %) RH increased AOV by 4.6 % (2.4–6.9 %) at lag0-2 d and 3.4 % (1.6–5.3 %) at lag 2-4 d, respectively. Lower humidity, which is associated with drying airway, could result in dehydration of the nasal cavity, increasing the risk of infection with viruses or bacteria; conversely, humid air can foster the proliferation of bacteria and mildew, potentially inducing asthma attack [49]. Wet effect on AOV were distinguished in China; in certain northern cities, the air is dry and no significant wet effect on AOV was observed [32], whereas in southern cities, dampness, or mildew at home exhibit strongest associations with asthma [15]. As a coastal city, the frequency of RH $\leq$ 30 % and RH $\geq$ 80 % accounts for 9.1 % and 11.9 % in Tianjin, respectively. Thus, both dry and wet effects on AOV warrants attention.

Air pollutants combined with moderately low temperature ($-8<T_{min}<0$ °C) can amplify AOC during winter. Similarly, a stronger correlation exists between PM_{10} and respiratory diseases on moderately cold days in Beijing [50]. However, the sensitivity analyses in estimating temperature RR after adjusting for air pollutants showed that cold effect was the major driver for increased AOV, while air pollutants only act as additional stressors. For $T_{min} = -5$ °C at lag1–2 d, $PM_{2.5}$, PM_{10}, SO_2, CO, and MDA8 O_3 could only increase AOV by 0.4 %, 1.5 %, 0.7 %, 0.1 % and 0.8 % on average, respectively. Additionally, a previous study highlighted that meteorological factors showed stronger associations with asthma than air pollutants [51]. In Eastern China, 14.1 % and 35.9 % of childhood asthma were attributed to air pollution and weather changes, respectively [52]. Although the average impact of air pollutants on AOV was minimal, the effect of severe air pollution on AOV not be overlooked or underestimated. Our results demonstrated that severe haze pollution during winter could result in >20 % increase in AOV. During summer, no obvious synergistic impact was observed on AOV induced by air pollutants and high temperature, although O_3 was the primary pollutant. A recent study reviewed the combined effect of hot weather and outdoor air pollution on respiratory health, and found that the interaction between high temperature and air pollution continues to be vague [53], while some studies showed non-statistically significant coefficients for the interaction between air pollution, high temperature, and respiratory outcomes [50,54,55]. In some western countries, abnormally high levels of pollutants caused by wildfire during hot weather significantly affect respiratory health [56–58]. According to the data from China's National Bureau of Statistics, the number of air conditioners per 100 households has seen a notable increase, with 70.4 air conditioners in 2013, 96.1 in 2017, 115.6 in 2019 and 133.9 in 2022. Tianjin ranks as the sixth-highest city in terms of air conditioner ownership across the country [59]. Although high temperatures are accompanied by high O_3 concentrations during summer, effective protective measures such as air conditioner can significantly reduce their adverse impact. Considering extreme weather occurred more intensively amidst climate change, further studies assessing the synergistic effect of high temperature and O_3, especially in North China, as well as the impact of extreme cold weather on AOV, are warranted.

The effects of air pollution and meteorology on AOV varied across different subgroups. In our study, females were more sensitive to air pollution, and cold effect and wet effects; conversely, males were more sensitive to dry effect. Susceptibility among females were also reported in other works [31,50]; females have narrower respiratory tract luminal and may have weaker lung immunity, and thus are more vulnerable to adverse environmental conditions [31]. Notably, sex differences in asthma were inconsistent and may vary across individual vulnerabilities, geographic regions, season, environmental conditions, and socioeconomic status [60,61]. Age-stratified analyses in this study suggest that children and older adults were more sensitive to air pollution and meteorological factors, which corroborates with previous studies [5,31,34]. In our study, children were susceptible to air pollution and cold effect, while older adults were more vulnerable to air pollution, as well as dry effect and wet effects. The central heating policy during winter in Northern China may play a potentially protective role in alleviating the adverse effects of low temperature on asthma attack for older adults [46], while children are more exposed to cold air than older adults due to outdoor activities. Moreover, children have higher breathing rate, narrower airways, and less developed lungs and immune systems [62]. Older adults with asthma may have pre-existing diseases, and their asthma can be exacerbated when exposed to air pollution and non-optimum humidity. The effect of air pollutants on AOV for children did not exhibit statistical differences, which may be attributed to the low prevalence of childhood asthma in our dataset. The proportion of children with asthma in Tianjin was less than half of that in Shanghai [63].

This study has some limitations. First, we only included seven major hospitals. Second, the age range was only classified into three groups due to the limitation of the original data, and thus a more refined analysis could not be performed. Finally, the inter annual increase in AOV cannot be explained by our analyses, and additional factors including genetics, lifestyle and diet should be considered. Nonetheless, the results of this study serve as a valuable scientific evidence for the forecasting and warning of asthma risk in Tianjin.

5. Conclusions

As air quality in North China Plain has largely improved due to the vigorous emission reduction, the PM_{10}, $PM_{2.5}$, and SO_2 associated risks on AOV tends to be weaker. In the methodology, the assumption that air pollutants affect AOV by linear or non-linear may lead to a significant difference on ER, and the selection of reference value should consider the current pollution level for each air pollutant in the linear assumption. Cold, sharp temperature drop, as well as dry and wet effects significantly impacted AOV in Tianjin. Temperature rise in spring could trigger asthma through increased pollen levels. Whereas in summer, temperature drops following heat days could lead to a more important impact rather than extreme hot effect. During winter, low temperature was a major driver for increased AOV, while air pollutants act as an additional stressor; severe haze pollution could result to a marked increase in AOV. Females, children, and older adults were more vulnerable to air pollution, non-optimum temperature, and RH. Considering extreme meteorological conditions occurred more intensively amidst climate change, the impact of extreme weather on AOV warrants attention.

Funding statement

This study is supported by the Tianjin Meteorological Bureau Project (202318ybxm12) and Special Innovation and Development Project of China Meteorological Administration (CXFZ2023J076).

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.heliyon.2023.e21803.

References

[1] GBD 2019 Diseases and Injuries Collaborators, Global burden of 369 diseases and injuries in 204 countries and territories, a systematic analysis for the global burden of disease study 2019, Lancet 396 (2020) (1990-2019) 1204–1222, https://doi.org/10.1016/S0140-6736(20)30925-9.
[2] C. Gautier, D. Charpin, Environmental triggers and avoidance in the management of asthma, J. Asthma Allergy 10 (2017) 47–56, https://doi.org/10.2147/JAA.S121276.
[3] C. Zhang, Y. Kong, K. Shen, The age, sex, and geographical distribution of self-reported asthma triggers on children with asthma in China, Front. Pediatr. 9 (2021), 689024, https://doi.org/10.3389/fped.2021.689024.
[4] C. Lu, Y.P. Zhang, B.Z. Li, Z.H. Zhao, C. Huang, X. Zhang, et al., Interaction effect of prenatal and postnatal exposure to ambient air pollution and temperature on childhood asthma, Environ. Int. 167 (2022), 107456, https://doi.org/10.1016/j.envint.2022.107456.
[5] P. Lu, Y.M. Zhang, J.T. Lin, G.X. Xia, W.Y. Zhang, L.D. Knibbs, et al., Multi-city study on air pollution and hospital outpatient visits for asthma in China, Environ. Pollut. 257 (2020), 113638, https://doi.org/10.1016/j.envpol.2019.113638.
[6] X.Y. Zheng, H. Ding, L.N. Jiang, S.W. Chen, J.P. Zheng, M. Qiu, et al., Association between air pollutants and asthma emergency room visits and hospital admissions in time series studies: a systematic review and Meta-analysis, PLoS One 10 (2015), e0138146, https://doi.org/10.1371/journal.pone.0138146.
[7] S.C. Anenberg, D.K. Henze, V. Tinney, P.L. Kinney, W. Raich, N. Fannet, et al., Estimates of the global burden of ambient $PM_{2.5}$, Ozone, and NO_2 on asthma incidence and emergency room visits, Environ. Health Persp. 126 (2018), 107004, https://doi.org/10.1289/EHP3766.
[8] M.C. Mirabelli, A. Vaidyanathan, W.D. Flanders, X.T. Qin, P. Garbe, Outdoor $PM_{2.5}$, ambient air temperature, and asthma symptoms in the past 14 days among adults with active asthma, Environ. Health Perspect. 124 (2016) 1882–1890, https://doi.org/10.1289/EHP92.
[9] L. Makra, S. Tombacz, B. Balint, Z. Sümeghy, T. Sánta, T. Hirsch, Influences of meteorological parameters and biological and chemical air pollutants on the incidence of asthma and rhinitis, Clim. Res. 37 (2008) 99–119, https://doi.org/10.3354/cr00752.
[10] F.W.S. Ko, W. Tam, T.W. Wong, C.K.W. Lai, G.W.K. Wong, T.F. Leung, et al., Effects of air pollution on asthma hospitalization rates in different age groups in Hong Kong, Clin. Exp. Allergy 37 (2007) 1312–1319, https://doi.org/10.1111/j.1365-2222.2007.02791.x.
[11] J. Cai, A. Zhao, J.Z. Zhao, R.J. Chen, W.B. Wang, S.D. Ha, et al., Acute effects of air pollution on asthma hospitalization in Shanghai, China, Environ. Pollut. 191 (2014) 139–144, https://doi.org/10.1016/j.envpol.2014.04.028.
[12] L. Ding, D.J. Zhu, D.H. Peng, Y. Zhao, Air pollution and asthma attacks in children: a case-crossover analysis in the city of Chongqing, China, Environ. Pollut. 220 (2017) 348–353, https://doi.org/10.1016/j.envpol.2016.09.070.
[13] Y.W. Zhang, H. Ni, L.J. Bai, Q. Cheng, H. Zhang, S.S. Wang, et al., The short-term association between air pollution and childhood asthma hospital admissions in urban areas of Hefei City in China: a time-series study, Environ. Pollut. 169 (2019) 510–516, https://doi.org/10.1016/j.envres.2018.11.043.

[14] X. Liu, Y.Y. He, C. Tang, Q.N. Wei, Z.H. Xu, W.Z. Yi, et al., Association between cold spells and childhood asthma in Hefei, an analysis based on different definitions and characteristics, Environ. Res. 195 (2021), 110738, https://doi.org/10.1016/j.envres.2021.110738.
[15] J. Ren, J. Xu, P. Zhang, Y. Bao, Prevalence and risk factors of asthma in preschool children in Shanghai, China: a cross-sectional study, Front. Pediatr. 9 (2022), 793452, https://doi.org/10.3389/fped.2021.793452.
[16] Y.Q. Feng, Y.S. Wang, L. Wu, Q. Shu, H.M. Li, X. Yang, Causal relationship between outdoor atmospheric quality and pediatric asthma visits in hangzhou, Heliyon 9 (2023), e14271, https://doi.org/10.1016/j.heliyon.2023.e14271.
[17] Bulletin on Ecological and Environmental Conditions of China, 2014-2017. https://www.mee.gov.cn.
[18] Q. Zhang, Y. Zheng, D. Tong, J.M. Hao, et al., Drivers of improved $PM_{2.5}$ air quality in China from 2013 to 2017, Proc. Natl. Acad. Sci. U.S.A. 116 (2019) 24463–24469, https://doi.org/10.1073/pnas.1907956116.
[19] Y. Wang, W. Gao, S. Wang, T. Song, Z. Gong, D. Ji, et al., Contrasting trends of $PM_{2.5}$ and surface-ozone concentrations in China from 2013 to 2017, Natl. Sci. Rev. 7 (2020) 1331–1339, https://doi.org/10.1093/nsr/nwaa032.
[20] H.C.Y. Lam, A.L. Li, E.Y.Y. Chan, W.B. Goggins, The short-term association between asthma hospitalisations, ambient temperature, other meteorological factors and air pollutants in Hong Kong: a time-series study, Thorax 71 (2016) 1097–1109, https://doi.org/10.1136/thoraxjnl.2015-208054.
[21] X.W. Cong, X.J. Xu, Y.L. Zhang, Q.H. Wang, L. Xu, X. Huo, Temperature drop and the risk of asthma: a systematic review and meta-analysis, Environ. Sci. Pollut. Res. Int. 24 (2017) 22535–22546, https://doi.org/10.1007/s11356-017-9914-4.
[22] J.K. Fang, J. Song, R.S. Wu, Y.F. Xie, X. Xu, Y.P. Zeng, et al., Association between ambient temperature and childhood respiratory hospital visits in Beijing, China: a time-series study (2013-2017), Environ. Sci. Pollut. Res. Int. 28 (2021) 29445–29454, https://doi.org/10.1007/s11356-021.12817-w.
[23] F. Feng, Y. Ma, Y. Zhang, J. Shen, H. Wang, et al., Effects of extreme temperature on respiratory diseases in Lanzhou, a temperate climate city of China, Environ. Sci. Pollut. Res. Int. 28 (2021) 49278–49288, https://doi.org/10.1007/s11356-021-14169-x.
[24] K. Sangkharat, M.A. Mahmood, J.E. Thornes, P.A. Fisher, F.D. Pope, Impact of extreme temperatures on ambulance dispatches in London, UK, Environ. Res. 182 (2020), 109100, https://doi.org/10.1016/j.envres.2019.109100.
[25] Q.N. Wei, L.Q. Zhong, J.Q. Gao, W.Z. Yi, R.B. Pan, J.J. Gao, et al., Diurnal temperature range and childhood asthma in Hefei, China: does temperature modify the association? Sci. Total Environ. 724 (2020), 138206 https://doi.org/10.1016/j.scitotenv.2020.138206.
[26] Y. Hu, Z. Xu, F. Jiang, S. Li, S. Liu, M. Wu, et al., Relative impact of meteorological factors and air pollutants on childhood allergic diseases in Shanghai, China, Sci. Total Environ. 706 (2020), 135975, https://doi.org/10.1016/j.scitotenv.2019.135975.
[27] S. Yan, X. Wang, Z. Yao, J. Cheng, H. Ni, Z. Xu, et al., Seasonal characteristics of temperature variability impacts on childhood asthma hospitalization in Hefei, China: does $PM_{2.5}$ modify the association? Environ. Res. 29 (2022), 112078, https://doi.org/10.1016/j.envres.2021.112078.
[28] J.W. Zhang, L.H. Feng, C.C. Hou, Q. Gu, How the constituents of fine particulate matter and ozone affect the lung function of children in Tianjin, China, Environ. Geochem. Health 42 (2020) 3303–3316, https://doi.org/10.1007/s10653-020-00574-7.
[29] A. Gasparrini, B. Armstrong, M.G. Kenward, Distributed lag non-linear models, Stat. Med. 29 (2010) 2224–2234, https://doi.org/10.1002/sim.3940.
[30] R.B. Pan, J.J. Gao, X. Wang, L.J. Bai, Q.N. Wei, W.Z. Yi, et al., Impacts of exposure to humidex on the risk of childhood asthma hospitalizations in Hefei, China: effect modification by gender and age, Sci. Total Environ. 691 (2019) 296–305, https://doi.org/10.1016/j.scitotenv.2019.07.026.
[31] L. Jin, T. Zhou, S. Fang, X. Zhou, B. Han, Y. Bai, The short-term effects of air pollutants on pneumonia hospital admissions in Lanzhou, China, 2014-2019: evidence of ecological time-series study, Air Qual. Atmos. Health 15 (2022) 2199–2213, https://doi.org/10.1007/s11869-022-01244-6.
[32] S.S. Lin, H.H. He, R. Jia, J. Du, N.Y. Ma, Y. Li, Impact of air pollution and meteorological factors on different populations and acute and chronic respiratory diseases in Zhengzhou City. Occup and Health 39 (2023) 577–583, https://doi.org/10.13329/j.cnki.zyyjk.2023.0119 (in Chinese).
[33] R. Chen, P. Yin, L. Wang, C. Liu, Y. Niu, W. Wang, et al., Association between ambient temperature and mortality risk and burden: time series study in 272 main Chinese cities, BMJ 363 (2018), k4306, https://doi.org/10.1136/bmj.k4306, 2018.
[34] A. Han, S.Z. Deng, J.R. Yu, Y.L. Zhang, B. Jalaludin, C. Huang, Asthma triggered by extreme temperatures: from epidemiological evidence to biological plausibility, Environ. Res. 216 (2023), 114489, https://doi.org/10.1016/j.envres.2022.114489.
[35] Y.H. Tian, X. Xiang, J. Juan, K.X. Sun, J. Song, Y.Y. Cao, et al., Fine particulate air pollution and hospital visits for asthma in Beijing, China, Environ. Pollut. 230 (2017) 227–233, https://doi.org/10.1016/j.envpol.2017.06.029.
[36] Y.W. Zhang, H. Ni, L.J. Bai, Q. Cheng, H. Zhang, S.S. Wang, et al., The short-term association between air pollution and childhood asthma hospital admissions in urban areas of Hefei City in China: a time-series study, Environ. Pollut. 169 (2019) 510–516, https://doi.org/10.1016/j.envres.2018.11.043.
[37] X. Luo, H.Y. Hong, Y.T. Lu, S.M. Deng, N.G. Wu, Q.L. Zhou, et al., Impact of air pollution and meteorological factors on incidence of allergic rhinitis: a low-latitude multi-city study in China, Allergy 00 (2022) 1–4, https://doi.org/10.1111/all.15469.
[38] J.J. He, S.L. Gong, Y. Yu, L.J. Yu, L. Wu, H.J. Mao, et al., Air pollution characteristics and their relation to meteorological conditions during 2014-2015 in major Chinese cities, Environ. Pollut. 223 (2017) 484–496, https://doi.org/10.1016/j.envpol.2017.01.050.
[39] A.J. Cohen, H.R. Anderson, B. Ostro, K.D. Pandey, K. Michal, K. Nino, et al., The global burden of disease due to outdoor air pollution, J. Toxico Environ. Health 68 (2005) 1301–1307, https://doi.org/10.1080/15287390590936166.
[40] R.W. Atkinson, H.R. Anderson, D.P. Strachan, J.M. Bland, S.A. Bremner, A. Ponce de Leon, Short-term associations between outdoor air pollution and visits to accident and emergency departments in London for respiratory complaints, Eur. Respir. J. 13 (1999) 257–265, https://doi.org/10.1183/09031936.99.13225799.
[41] G. D'Amato, Outdoor air pollution in urban areas and allergic respiratory diseases, Monaldi Arch. Chest Dis. 54 (1999) 470–474.
[42] A. Lindgren, E. Stroh, U. Nihlen, P. Montnémery, A. Axmon, K. Jakobsson, Traffic exposure associated with allergic asthma and allergic rhinitis in adults. A cross-sectional study in southern Sweden, Int. J. Health Geogr. 8 (2009) 25, https://doi.org/10.1186/1476-072X-8-25.
[43] G. D'Amato, G. Liccardi, M. D'Amato, M. Cazzola, Outdoor air pollution, climatic changes and allergic bronchial asthma, Eur. Respir. J. 20 (2002) 763–776, https://doi.org/10.1183/09031936.02.00401402.
[44] C.R. Jung, L.H. Young, H.T. Hsu, M.Y. Lin, Y.C. Chen, B.F. Hwang, et al., $PM_{2.5}$ components and outpatient visits for asthma: a time-stratified case-crossover study in a suburban area, Environ. Pollut. 231 (2017) 1085–1092, https://doi.org/10.1016/j.envpol.2017.08.102.
[45] Y.X. Zhu, T. Yang, S.J. Huang, H.C. Li, J. Lei, X.W. Xue, et al., Cold temperature and sudden temperature drop as novel risk factors of asthma exacerbation: a longitudinal study in 18 Chinese cities, Sci. Total Environ. 814 (2022), 151959, https://doi.org/10.1016/j.scitotenv.2020.138206.
[46] J. Lei, L. Peng, T. Yang, S.J. Huang, Y.X. Zhu, Y. Gao, et al., Non-optimum ambient temperature may decrease pulmonary function: a longitudinal study with intensively repeated measurements among asthmatic adult patients in 25 Chinese cities, Environ. Int. 164 (2022), 107283, https://doi.org/10.1016/j.envint.2022.107283.
[47] H.Y. Liu, G.Q. Fu, J. Wang, The effect of meteorological elements on adult asthma hospitalization in Shijiazhuang, J. Meteor. Environ. 35 (2019) 137–143, https://doi.org/10.3969/j.issn.1673-503X.2019.05.018 (in Chinese).
[48] X. Xiong, Y.C. Wei, H.C.Y. Lam, C.K.H. Wong, S.Y.F. Lau, S. Zhao, et al., Association between cold weather, influenza infection, and asthma exacerbation in adults in Hong Kong, Sci. Total Environ. 857 (2023), 159362, https://doi.org/10.1016/j.scitotenv.2022.159362.
[49] A. Leitte, C. Petrescu, U. Franck, M. Richter, O. Suciu, R. Ionoviciet, et al., Respiratory health, effects of ambient air pollution and its modification by air humidity in Drobeta-Turnu Severin, Romania, Sci. Total Environ. 407 (2009) 4004–4011, https://doi.org/10.1016/j.scitotenv.2009.02.042.
[50] X. Song, L. Jiang, S. Wang, J. Tian, K. Yang, X. Wang, et al., The impact of main air pollutants on respiratory emergency department visits and the modification effects of temperature in Beijing, China, Environ. Sci. Pollut. Res. 28 (2021) 6990–7000, https://doi.org/10.1007/s11356-020-10949-z.
[51] H.H. Zhang, S. Liu, Z.J. Chen, B. Zu, Y.H. Zhao, Effects of variations in meteorological factors on daily hospital visits for asthma: a time-series study, Environ. Res. 182 (2020), 109115, https://doi.org/10.1016/j.envres.2020.109115.
[52] C. Zhang, Y. Kong, K. Shen, The age, sex, and geographical distribution of self-reported asthma triggers on children with asthma in China, Front. Pediatr. 9 (2021), 689024, https://doi.org/10.3389/fped.2021.689024.
[53] E. Grigorieva, A. Lukyanets, Combined effect of hot weather and outdoor air pollution on respiratory health: Literature review, Atmosphere 12 (2021) 790, https://doi.org/10.3390/atmos12060790.

[54] H. Achebak, D. Devolder, V. Ingole, J. Ballester, Reversal of the seasonality of temperature-attributable mortality from respiratory diseases in Spain, Nat. Commun. 11 (2020) 2457, https://doi.org/10.1038/s41467-020-16273-x.

[55] Y. Cheng, H. Kan, Effect of the interaction between outdoor air pollution and extreme temperature on daily mortality in Shanghai, China, J. Epidemiol. 22 (2012) 28–36, https://doi.org/10.2188/jea.JE20110049.

[56] V. Weilnhammer, J. Schmid, I. Mittermeier, F. Schreiber, L. Jiang, V. Pastuhovic, et al., Extreme weather events in europe and their health consequences–A systematic review, Int. J. Hyg Environ. Health 233 (2021), 113688, https://doi.org/10.1016/j.ijheh.2021.113688.

[57] M. Burke, A. Driscoll, S. Heft-Neal, J. Xue, J. Burney, M. Wara, The changing risk and burden of wildfire in the United States, Proc. Natl. Acad. Sci. USA 118 (2021), 2011048118, https://doi.org/10.1073/pnas.2011048118.

[58] C. Sorensen, J.A. House, K. O'Dell, S.J. Brey, B. Ford, J.R. Pierce, et al., Associations between wildfire-related $PM_{2.5}$ and intensive care unit admissions in the United States, 2006-2015. Geo. Health 5 (2021), e2021GH000385, https://doi.org/10.1029/2021GH000385.

[59] National Bureau of Statistics. https://data.stats.gov.cn/index.htm.

[60] M. Muenchhoff, P.J. Goulder, Ex differences in pediatric infectious diseases, J. Infect. Dis. 209 (2014) S120–S126, https://doi.org/10.1093/infdis/jiu232.

[61] M. Trivedi, E. Denton, Asthma in children and adults—what are the differences and what can they tell us about asthma? Front. Pediatr. 7 (2019) 256, https://doi.org/10.3389/fped.2019.00256.

[62] C. Lv, X. Wang, N. Pang, L. Wang, Y. Wang, T. Xu, et al., The impact of airborne particulate matter on pediatric hospital admissions for pneumonia among children in Jinan, China: a case-crossover study, J. Air Waste Manag. Assoc. 67 (2017) 669–676, https://doi.org/10.1080/10962247.2016.1265026.

[63] C.H. Liu, J.G. Hong, Y.X. Shang, J. Sun, M. Duolikun, M.N. Shan, et al., Comparison of asthma prevalence in children from 16 cities of China in 20 year. Chinese J. Pract. Pediatr 30 (2015) 596–600, https://doi.org/10.7504/ek2015080609 (in Chinese).