



Multipollutant air-quality improvement and ozone divergence across Chinese cities

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Abstract. Over the past decade, China has achieved substantial reductions in particulate matter and primary gaseous pollutants, while ground-level ozone (O₃) has shown a contrasting response. We analysed air-quality changes across 356 Chinese cities during 2015–2025, with trend and city-level divergence analyses for 331 cities. Annual mean PM_{2.5}, PM₁₀, SO₂, NO₂ and CO concentrations decreased by 40.8%, 35.3%, 70.3%, 36.2% and 45.0%, respectively, whereas O₃ increased by 10.3% nationally. Positive O₃ Sen slopes occurred in 76.1% of cities, although only 19.9% were statistically significant. Meteorological standardization across 318 cities reduced the mean early-to-late O₃ increase from 7.62 to 3.76–4.02 µg/m³, indicating a substantial influence of meteorological variability but leaving a persistent positive increase. Satellite observations further showed that warm-season HCHO changes between 2019–2021 and 2023–2025 were positively associated with concurrent O₃ changes (Spearman $\rho = 0.36$, $p < 0.001$), while the association with tropospheric NO₂ was weaker ($\rho = 0.21$, $p < 0.001$). The Fenwei Plain showed the largest regional O₃ increase (+14.3%), and seasonal relationships between O₃ and other pollutants reversed from positive in spring and summer to negative in winter. A city-level divergence framework further showed that similar reductions in conventional pollutants can be accompanied by markedly different O₃ responses. These findings demonstrate that reducing particulate and primary gaseous pollution alone does not guarantee lower photochemical pollution and highlight the need for coordinated, city-specific multipollutant control.

Keywords: air quality, ozone, multipollutant trends, city-level divergence, China



31 1. Introduction

32 Over the past decade, air quality in Chinese cities has undergone substantial changes.
33 Owing to large-scale national policies for atmospheric pollution control, concentrations
34 of particulate matter (PM_{2.5} and PM₁₀), sulfur dioxide (SO₂), nitrogen dioxide (NO₂)
35 and carbon monoxide (CO) have decreased markedly in most cities across the country
36 (Zheng et al., 2018; Zhang et al., 2019; Geng et al., 2024; Zheng et al., 2024; Wang et
37 al., 2025a). These changes represent one of the most prominent examples of rapid air-
38 quality improvement under conditions of large-scale urbanization and intensive
39 industrial activity. However, the reduction of most conventional pollutants does not
40 imply that the entire urban air-pollution system has changed in the same direction.

41 A key exception is ground-level ozone (O₃). Unlike primary pollutants and
42 particulate pollution, O₃ is a secondary photochemical pollutant whose formation is
43 controlled by nonlinear interactions among NO_x, VOCs, solar radiation, temperature,
44 aerosol loading and meteorological conditions (Sillman, 1999; Li et al., 2019a; Wang
45 et al., 2022a). VOC emissions have declined less consistently than several other
46 anthropogenic pollutants in China, and coordinated VOC–NO_x control has therefore
47 become increasingly important for mitigating O₃ pollution (Li et al., 2019b). Therefore,
48 reductions in primary pollutant and aerosol emissions do not necessarily lead to a
49 decrease in O₃. On the contrary, reduced aerosol loading may promote photolysis, alter
50 the oxidative capacity of the atmosphere and modify the conditions for O₃ formation
51 (Lou et al., 2014; Li et al., 2019b; Ivatt et al., 2022).

52 The significance of this problem extends beyond a simple assessment of pollutant
53 concentrations. PM_{2.5} and O₃ remain among the most important components of air-
54 pollution risk to public health (Cohen et al., 2017; Niu et al., 2022). Although PM_{2.5}
55 reductions have brought substantial benefits, the simultaneous increase in O₃ may partly
56 offset overall progress in reducing the health-related air-pollution burden (Achebak et
57 al., 2024; Silver et al., 2025). In addition, elevated O₃ can reduce agricultural
58 productivity, damage vegetation and forest ecosystems, and cause economic losses
59 through reduced crop yield and forest productivity (Tai et al., 2014; Mills et al., 2018;
60 Feng et al., 2019). Therefore, further improvement in air quality requires not only
61 continued PM control, but also a more precise understanding of the drivers of O₃
62 increases.



63 Despite a large body of research, important knowledge gaps remain. PM_{2.5} and O₃
64 are increasingly regarded as interconnected air-quality problems requiring coordinated
65 control, because the decline in PM_{2.5} in China has been accompanied by worsening O₃
66 pollution and changes in the health-related air-pollution burden (Zhao et al., 2021; Xiao
67 et al., 2022). However, many studies have focused either on national trends and health
68 burden or on the chemical mechanisms of O₃ increases, whereas the joint city-level
69 evolution of PM_{2.5}, PM₁₀, SO₂, NO₂, CO and O₃ within a unified multipollutant system
70 has been less systematically examined (Chu et al., 2020; Liu et al., 2023). In addition,
71 regional heterogeneity has often been analyzed at the level of individual pollutants or
72 individual urban agglomerations, which makes it difficult to compare regions with
73 different emission sources, climate, topography and photochemical conditions (Qiao et
74 al., 2019; Dai et al., 2021; Wang et al., 2021; Wang et al., 2022b; Zhang et al., 2022;
75 Qian et al., 2024; Xu et al., 2024). Finally, it remains insufficiently clear why cities
76 with comparable reductions in conventional pollutants can exhibit different trajectories
77 of O₃ response. This city-level divergence is a key limitation for moving from national
78 air-quality assessment toward more precise city-specific control strategies.

79 Seasonality further complicates the interpretation of O₃ response. In spring and
80 summer, high solar radiation and temperature can enhance photochemical O₃ formation,
81 whereas in winter weak photochemistry and NO titration can lead to opposite
82 relationships between O₃ and primary pollutants (Sillman, 1999; Li et al., 2021).
83 Therefore, long-term changes in O₃ cannot be explained only by annual trends or by
84 reductions in individual pollutants. An analysis is needed that simultaneously considers
85 national trends, regional heterogeneity, seasonal multipollutant relationships and city-
86 level divergence between conventional pollutant reductions and O₃ response.

87 In this study, we analyze the multipollutant evolution of air quality in Chinese cities
88 during 2015–2025 using ground-based observations from the China National
89 Environmental Monitoring Centre (CNEMC) for six major pollutants, namely PM_{2.5},
90 PM₁₀, SO₂, NO₂, CO and O₃. The main objectives of this study are to: (1) quantify
91 national long-term changes in the six pollutants; (2) identify regional differences in
92 multipollutant evolution; (3) characterize seasonal O₃-centred relationships among
93 pollutants; and (4) develop a city-level divergence framework to assess how reductions
94 in conventional pollutants are associated with changes in O₃ across individual cities; (5)
95 quantify the extent to which long-term meteorological changes modify the observed O₃
96 response using city-specific meteorological standardization; and (6) examine recent



97 changes in ozone-precursor conditions using satellite HCHO and tropospheric NO₂
 98 observations and assess their associations with surface O₃ changes. Rather than
 99 providing another assessment of national pollutant trends, the central aim of this study
 100 is to identify and quantify city-level divergence in O₃ responses under broadly
 101 comparable improvements in conventional air pollution, and to evaluate whether this
 102 divergence can be constrained by meteorological variability and recent changes in
 103 precursor conditions.

104 **2. Materials and methods**

105 **2.1 Air-quality data and preprocessing**

106 This study used ground-based air-quality observations from the national monitoring
 107 network of the China National Environmental Monitoring Centre (CNEMC) for the
 108 period from 1 January 2015 to 31 December 2025. The original data had hourly
 109 resolution and covered six major air pollutants: PM_{2.5}, PM₁₀, SO₂, NO₂, CO and O₃. All
 110 measurements were conducted in accordance with the national ambient air-quality
 111 standard (GB3095-2012). For subsequent analysis, hourly observations were converted
 112 into daily values. Daily mean concentrations were calculated for PM_{2.5}, PM₁₀, SO₂, NO₂
 113 and CO, whereas O₃ was represented by the maximum daily 8-hour mean (MDA8).

114 For the annual analysis, daily data were aggregated into annual mean values
 115 separately for each city, year and pollutant. An annual value was included in the
 116 analysis only when at least 274 valid daily observations were available in the
 117 corresponding year, corresponding to approximately 75% annual coverage. The same
 118 principle was applied to the seasonal analysis: a seasonal value for a given city and year
 119 was retained only when at least 68 valid daily observations were available in the
 120 corresponding season.

121 **2.2 Regional classification of cities**

122 To analyze spatial heterogeneity, cities were divided into eight regions: Beijing–
 123 Tianjin–Hebei (BTH), Yangtze River Delta (YRD), Pearl River Delta (PRD), Fenwei
 124 Plain, Sichuan Basin, Northeast China, Northwest China and Other China. This
 125 classification accounted for the major air-pollution regions of China, city location,
 126 climatic conditions, topography and characteristic pollution sources. Cities belonging
 127 to well-known polluted regions were assigned to the corresponding groups based on
 128 predefined city lists. The remaining cities were assigned to Northeast China, Northwest



China or Other China according to their geographical location. Regional mean values were calculated as averages across cities within each region.

2.3 Estimation of national trends

Long-term pollutant trends were evaluated using annual city-level concentrations for 2015–2025. For each city and pollutant, the trend magnitude was calculated using Sen’s slope estimator. Only cities with at least eight valid annual values over the 11-year period were included in the trend analysis. The statistical significance of monotonic trends was assessed using the Mann–Kendall test. Trends were considered statistically significant at $p < 0.05$.

National annual mean concentrations were calculated as averages across all available cities in each year. For the national summary assessment, pollutant changes were determined between 2015 and 2025. For the regional and city-level analyses, two multi-year periods were used: the early period of 2015–2017 and the late period of 2023–2025. This approach reduces the influence of individual anomalous years and allows a more robust assessment of changes between the beginning and the end of the study period.

2.4 Regional changes and O₃ penalty ratio

For each region, relative changes in PM_{2.5}, PM₁₀, SO₂, NO₂, CO and O₃ were calculated between the early period (2015–2017) and the late period (2023–2025). A change was considered valid only when at least two annual values were available in each of the two periods. These calculations were used to assess how consistently conventional pollutants decreased and how O₃ changed against this background.

To quantify the increase in O₃ relative to the reduction of each conventional pollutant, an O₃ penalty ratio was calculated. This indicator reflects the increase in O₃ per 10% reduction in a given pollutant. For example, if PM_{2.5} decreased while O₃ increased simultaneously, the penalty ratio indicates how strongly the increase in O₃ “offset” the reduction in PM_{2.5}. The calculation was performed as follows:

$$Penalty\ ratio_p = 10 \times \frac{\Delta O_3}{R_p}$$

where ΔO_3 is the relative change in O₃ between 2015–2017 and 2023–2025, expressed as a percentage, and R_p is the reduction of the corresponding pollutant (p), expressed as a positive value. For example, if PM_{2.5} changed by –40%, then $R_{2.5} =$



160 40%. Positive penalty-ratio values indicate that O₃ increased against the background of
 161 a reduction in the given pollutant; negative values indicate that O₃ decreased despite a
 162 reduction in the conventional pollutant.

163 2.5 Seasonal correlation analysis

164 Seasonal relationships among pollutants were analyzed using city-season-year mean
 165 values. Four seasons were defined: spring (March–May), summer (June–August),
 166 autumn (September–November) and winter (December–February). For each season,
 167 pairwise Spearman rank correlations were calculated among PM_{2.5}, PM₁₀, SO₂, NO₂,
 168 CO and O₃. Spearman correlation was used because relationships among pollutants may
 169 be nonlinear and sensitive to seasonal and regional differences.

170 To control for multiple testing, *p*-values were adjusted using the Benjamini–
 171 Hochberg false discovery rate procedure. Associations were considered statistically
 172 significant at $q < 0.05$. This analysis was used only to evaluate statistical associations
 173 among pollutants and was not interpreted as direct evidence of causal relationships.

174 2.6 City-level divergence framework

175 To assess the divergence between reductions in conventional pollutants and the O₃
 176 response, a city-level divergence framework was developed. For each city, relative
 177 changes in PM_{2.5}, PM₁₀, SO₂, NO₂, CO and O₃ were calculated between the early period
 178 (2015–2017) and the late period (2023–2025).

179 Based on these changes, a five-pollutant reduction index (R5) was calculated for
 180 each city, representing the mean reduction of the five conventional pollutants:

$$181 \quad R5 = \text{mean}(R_{PM2.5}, R_{PM10}, R_{SO2}, R_{NO2}, R_{CO})$$

182 where R_p is the reduction of each pollutant expressed as a positive value. For
 183 example, a 40% reduction in PM_{2.5} corresponds to $R_{PM2.5} = 40\%$. Higher R5 values
 184 therefore indicate stronger overall reductions in conventional pollutants in a given city.

185 The O₃ response was assessed separately as the relative change in O₃ between the
 186 early and late periods (ΔO_3). Cities were then classified into four types based on the
 187 median values of R5 and ΔO_3 .

188 Type I represents strong reductions in conventional pollutants ($R5 \geq \text{median}$) and a
 189 weak O₃ response ($\Delta O_3 < \text{median}$). Type II represents strong reductions in conventional
 190 pollutants ($R5 \geq \text{median}$) and a strong O₃ increase ($\Delta O_3 \geq \text{median}$). Type III represents
 191 weaker reductions in conventional pollutants ($R5 < \text{median}$) and a strong O₃ increase



($\Delta O_3 \geq \text{median}$). Type IV represents weaker reductions in conventional pollutants ($R5 < \text{median}$) and a weak O_3 response ($\Delta O_3 < \text{median}$).

The relationship between $R5$ and ΔO_3 was additionally evaluated using Spearman's rank correlation coefficient. This typology was interpreted as an empirical city-level classification rather than a direct diagnosis of O_3 chemical regimes, because the available data do not allow direct determination of NO_x –VOC sensitivity or aerosol radiative effects.

2.7 Meteorological data and O_3 standardization

Ground-based meteorological observations for 2015–2025 were obtained from weather stations archived in the NOAA Integrated Surface Database (ISD) (Smith et al., 2011). Three-hourly observations were converted from UTC to China Standard Time (UTC+8) before aggregation to local calendar days. Daily variables included mean near-surface air temperature (T_{mean}), daily maximum air temperature (T_{max}), relative humidity (RH) and wind speed (WS). Meteorological stations were matched to cities according to geographic proximity and temporal completeness, and up to three eligible stations were combined using inverse-distance weighting. The median distance to the nearest selected station was approximately 43 km; sensitivity analyses excluding cities represented by stations >100 km away were also performed.

Meteorological changes were evaluated using the same early (2015–2017) and late (2023–2025) periods as the pollutant analysis. City-year values required at least 274 valid daily observations, and at least two valid annual values were required in each period. A total of 318 cities had sufficient paired O_3 and meteorological coverage. To quantify the influence of meteorological variability on O_3 , city-specific daily models were fitted in which MDA8 O_3 was represented as a nonlinear function of temperature, RH and WS, together with seasonal harmonic terms, day-of-week and year effects. Nonlinear meteorological effects were represented using spline terms for temperature, RH and WS, while annual seasonality was represented by harmonic sine and cosine terms. The primary model used T_{mean} , while a sensitivity model used T_{max} .

For each city, 31-day seasonal climatologies of the meteorological variables were constructed from the 2015–2025 observations. Meteorologically standardized O_3 was calculated as

$$O_{3,std} = O_{3,obs} - \hat{O}_3(M_{actual}) + \hat{O}_3(M_{clim})$$



224 where the two model predictions correspond to observed and climatological
225 meteorological conditions, respectively. Early–late changes in observed and
226 standardized O₃ were then compared using the same completeness criteria. This
227 procedure was used to quantify the influence of the meteorological variables
228 represented in the model rather than to provide causal attribution of the remaining O₃
229 change.

230 **2.8 Satellite HCHO and NO₂ analysis**

231 TROPOMI HCHO (De Smedt et al., 2018) and tropospheric NO₂ (van Geffen et al.,
232 2022) observations were analysed for May–September during 2019–2025 to
233 characterize recent changes in ozone-precursor conditions across Chinese cities. The
234 analysis used Sentinel-5P/TROPOMI OFFL products distributed through the
235 Copernicus satellite data services. Daily satellite observations were spatially averaged
236 within 25-km buffers centred on each city. Retrievals with cloud fraction > 0.3 were
237 excluded before aggregation, and daily city-level values were subsequently averaged to
238 May–September seasonal means. Satellite HCHO columns have previously been used
239 to investigate spatial and temporal variability in VOC emissions over China (Shen et
240 al., 2019). HCHO provided a satellite-based indicator of VOC-related atmospheric
241 variability that was not available from the ground monitoring network, while
242 TROPOMI NO₂ provided a concurrent column-based precursor measure
243 complementary to the surface NO₂ observations. For each city and year, seasonal mean
244 HCHO and NO₂ columns were calculated when at least 60 valid daily retrievals were
245 available; 98.4% of city–year records satisfied this joint sampling criterion.

246 City-level temporal tendencies were quantified using Sen’s slope applied to annual
247 May–September means, with nominal significance assessed using the Mann–Kendall
248 test ($p < 0.05$). Trend estimates required at least five valid annual values during 2019–
249 2025. To characterize recent precursor changes independently of a monotonic-trend
250 assumption, mean HCHO and tropospheric NO₂ conditions during 2019–2021 were
251 compared with those during 2023–2025, requiring at least two valid years in each
252 period. Changes were expressed as percentage differences relative to the 2019–2021
253 mean.

254 Surface O₃ was aggregated over the same May–September periods, and Δ O₃ was
255 calculated as the absolute difference between the 2023–2025 and 2019–2021 means.
256 Associations between precursor changes and concurrent O₃ changes were evaluated



257 using Spearman rank correlations. Correlations between city-specific Sen slopes were
258 additionally examined as a robustness test. HCHO was interpreted as a satellite proxy
259 for VOC-related atmospheric variability rather than as a direct measure of VOC
260 emissions.

261

262 **3. Results and discussion**

263 **3.1 National spatiotemporal evolution of six pollutants**

264 National trends of the six major pollutants reveal substantial but uneven changes in
265 urban air quality across China during 2015–2025 (Figure 1). According to Table 1, the
266 annual mean concentration of PM_{2.5} decreased from 50.4 to 29.8 µg/m³ (–40.8%), while
267 PM₁₀ declined from 88.8 to 57.5 µg/m³ (–35.3%). Among gaseous pollutants, the
268 strongest relative decrease was observed for SO₂, from 25.7 to 7.6 µg/m³ (–70.3%),
269 whereas CO and NO₂ decreased by 45.0% and 36.2%, respectively. These results
270 indicate a substantial weakening of particulate and primary gaseous pollution in
271 Chinese cities, consistent with previous studies (Silver et al., 2018; Zheng et al., 2018;
272 Zhai et al., 2019;).

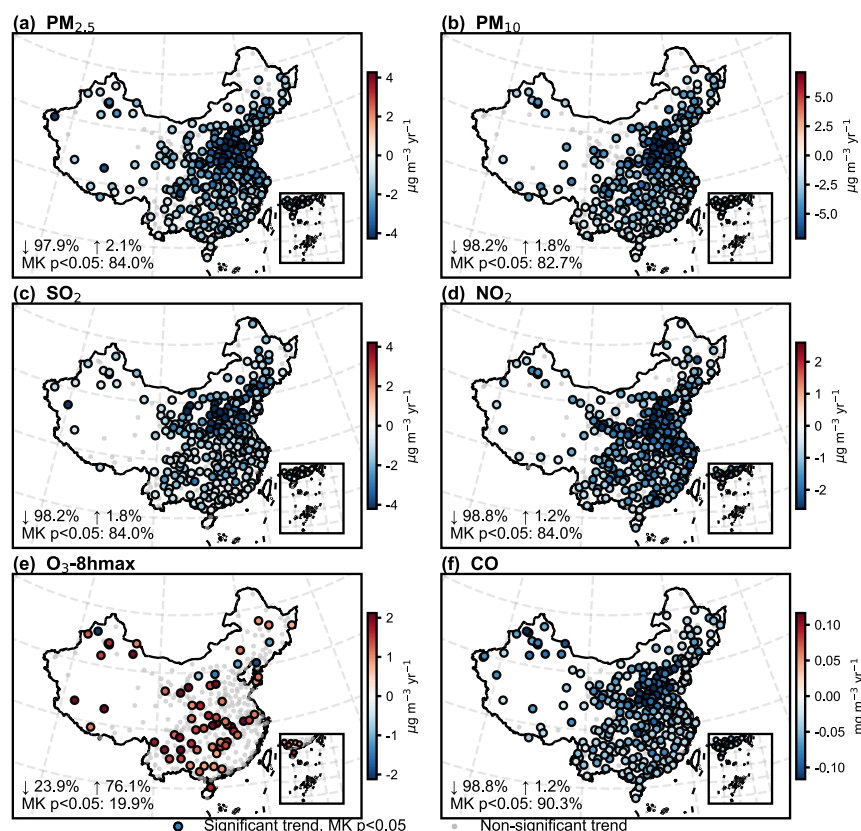


Figure 1. Long-term trends of six major air pollutants across Chinese cities during 2015–2025.

The spatial distribution of Sen's slope shows that decreases in $PM_{2.5}$, PM_{10} , SO_2 , NO_2 and CO were widespread across the country (Figure 1a–d, f). For these five pollutants, more than 97% of cities exhibited negative trends, and most of these trends were statistically significant ($p < 0.05$): 84.0% for $PM_{2.5}$, SO_2 and NO_2 , 82.7% for PM_{10} , and 90.3% for CO . This spatial coherence indicates that the decline in concentrations represented a robust national trend rather than a result of local changes (Li et al., 2020; Geng et al., 2024; Zheng et al., 2024).

The decreases in SO_2 , CO and NO_2 reflect an overall reduction in combustion-related and primary emission sources. SO_2 and CO are closely associated with fuel combustion, whereas NO_2 largely reflects contributions from transport, industry and power generation. Their simultaneous decline indicates a weakening of primary pollution burden. The decrease in $PM_{2.5}$ is consistent with this pattern, because fine



particulate matter depends on both direct emissions and gaseous precursors of secondary aerosols, including SO₂ and NO₂. Median Sen's slope values for PM_{2.5}, PM₁₀, SO₂, NO₂ and CO were $-1.83 \mu\text{g}/\text{m}^3/\text{yr}$, $-2.97 \mu\text{g}/\text{m}^3/\text{yr}$, $-1.11 \mu\text{g}/\text{m}^3/\text{yr}$, $-1.19 \mu\text{g}/\text{m}^3/\text{yr}$ and $-0.042 \text{ mg}/\text{m}^3/\text{yr}$, respectively, confirming the persistent nature of these decreases.

In contrast to the other pollutants, O₃ showed the opposite overall pattern: its annual mean concentration increased from 83.6 to 92.1 $\mu\text{g}/\text{m}^3$ (+10.3%). Positive Sen's slopes were observed in 76.1% of cities, whereas 23.9% showed negative slopes (Figure 1e). However, the direction of the Sen's slope should be distinguished from statistical significance: only 19.9% of cities exhibited statistically significant O₃ trends according to the Mann–Kendall test ($p < 0.05$). The median Sen's slope for O₃ was $+0.54 \mu\text{g}/\text{m}^3/\text{yr}$, with values ranging from -2.14 to $+5.36 \mu\text{g}/\text{m}^3/\text{yr}$. Thus, although the predominant direction of O₃ change was positive, statistically significant long-term trends were confined to a substantially smaller subset of cities.

This contrast can be explained by differences in the nature of these pollutants. PM_{2.5}, PM₁₀, SO₂, NO₂ and CO are more directly linked to primary emissions or relatively direct secondary processes, and therefore respond more strongly to reductions in emission intensity. O₃ is a secondary photochemical pollutant whose formation depends on nonlinear interactions among NO_x, VOCs, solar radiation, temperature and aerosol loading (Li et al., 2019a; Xie et al., 2023). Therefore, reductions in primary pollutants do not necessarily translate into lower O₃; on the contrary, reduced aerosol loading and changes in NO_x chemistry may create conditions favorable for O₃ increases (Ivatt et al., 2022).

Importantly, O₃ displayed a more heterogeneous statistical pattern: statistically significant trends were detected in only 19.9% of cities, compared with 82.7–90.3% for the other pollutants. This contrast indicates that declines in particulate and primary gaseous pollutants were substantially more spatially coherent than the long-term evolution of O₃. The greater heterogeneity of O₃ responses is consistent with the influence of spatially varying precursor conditions and meteorological and photochemical environments, which are examined further below.

Overall, over more than a decade, air quality in Chinese cities improved substantially for most pollutants, but O₃ evolved in the opposite direction. This highlights the need to move from single-pollutant control toward more coordinated multipollutant management, especially given the increasing role of O₃ in urban air pollution.



Table 1. National mean changes in six major air pollutants across Chinese cities, 2015–2025.

Pollutant	Mean 2015	Mean 2025	Absolute change	Relative change	Median Sen's slope	n cities
PM _{2.5}	50.4 µg/m ³	29.8 µg/m ³	-20.6 µg/m ³	-40.8%	-1.83 µg/m ³ /yr	331
PM ₁₀	88.8 µg/m ³	57.5 µg/m ³	-31.3 µg/m ³	-35.3%	-2.97 µg/m ³ /yr	329
SO ₂	25.7 µg/m ³	7.6 µg/m ³	-18.1 µg/m ³	-70.3%	-1.11 µg/m ³ /yr	331
NO ₂	30.0 µg/m ³	19.2 µg/m ³	-10.9 µg/m ³	-36.2%	-1.19 µg/m ³ /yr	331
O ₃	83.6 µg/m ³	92.1 µg/m ³	+8.6 µg/m ³	+10.3%	+0.54 µg/m ³ /yr	331
CO	1.08 mg/m ³	0.60 mg/m ³	-0.49 mg/m ³	-45.0%	-0.042 mg/m ³ /yr	331

3.2 Regional differences in pollutant evolution

Regional analysis shows that the national pattern of air-quality change manifested unevenly across different regions of China (Figure 2). PM_{2.5}, PM₁₀, SO₂, NO₂ and CO decreased in all regions, but the magnitude of these changes varied substantially, reflecting differences in baseline pollution levels, source structure and local conditions controlling pollutant formation.

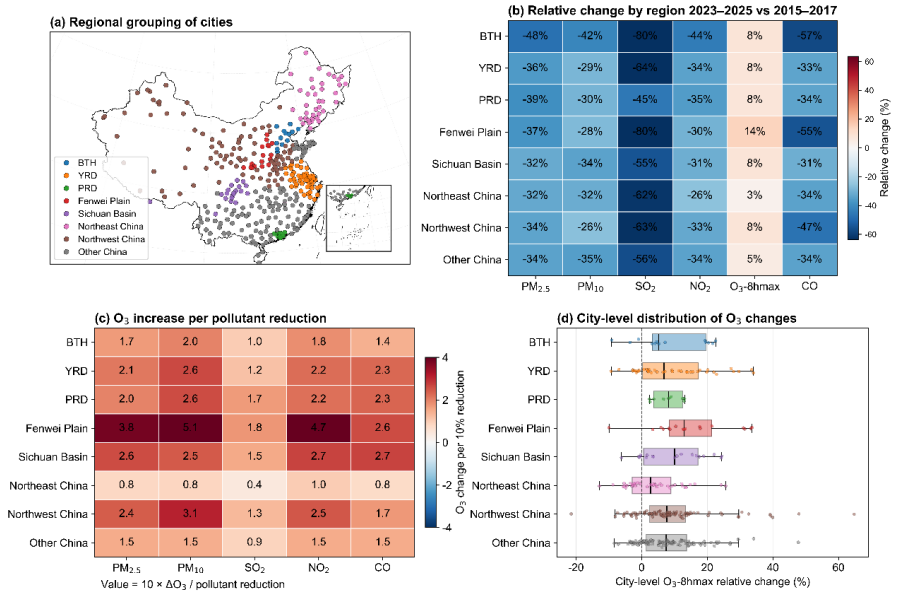


Figure 2. Regional heterogeneity in multipollutant evolution across Chinese cities.



333 The most pronounced decrease in $PM_{2.5}$ was observed in BTH (-47.6%). In the other
 334 regions, the reduction was within a narrower range, from 32.2% in the Sichuan Basin
 335 to 39.1% in PRD, while in Northwest China it reached 33.7% (Figure 2b). Late-period
 336 $PM_{2.5}$ concentrations remained high in BTH ($37.0 \mu\text{g}/\text{m}^3$) and the Fenwei Plain (39.9
 337 $\mu\text{g}/\text{m}^3$), regions with basin-like terrain that limits dispersion, and low in PRD (20.6
 338 $\mu\text{g}/\text{m}^3$), where sea breezes provide effective ventilation.

339 PM_{10} decreased from 41.6% in BTH to 26.5% in Northwest China (Figure 2b). This
 340 difference can be explained by the fact that PM_{10} includes coarse particles associated
 341 with road dust, resuspension and natural sources. In Northwest China, the arid climate
 342 and extensive deserts, including the Taklamakan and Gobi deserts, determine a high
 343 contribution of natural dust; therefore, the anthropogenic reduction in PM_{10} is less
 344 pronounced there (Mu et al., 2023; Filonchyk et al., 2019, 2024). The highest late-
 345 period PM_{10} concentrations remained in the Fenwei Plain ($83.0 \mu\text{g}/\text{m}^3$) and Northwest
 346 China ($77.3 \mu\text{g}/\text{m}^3$).

347 SO_2 showed the strongest decrease among all pollutants: -80.5% in the Fenwei Plain
 348 and -79.7% in BTH, with reductions exceeding 55% in most other regions (Figure 2b).
 349 This reflects targeted reductions in coal-related emissions, which were especially
 350 effective in regions with historically high shares of coal-based energy, including the
 351 Fenwei Plain, the coal belt of Shanxi–Shaanxi, and BTH (Zheng et al., 2018; Zhang et
 352 al., 2019; Tong et al., 2020). The lowest late-period SO_2 concentrations were observed
 353 in PRD ($6.5 \mu\text{g}/\text{m}^3$) and YRD ($6.9 \mu\text{g}/\text{m}^3$). CO decreased by $31\text{--}57\%$ across different
 354 regions, further confirming a broad reduction in combustion-related pollution.

355 NO_2 decreased in all regions, but its reduction was more moderate than that of SO_2 .
 356 The largest decrease was observed in BTH (-44.3%), whereas the smallest decrease
 357 occurred in Northeast China (-25.9% ; Figure 2b). This difference can be explained by
 358 the fact that NO_2 is associated not only with industry and power generation, but also
 359 with transport, where emission reductions occur more gradually. In Northeast China,
 360 extremely cold winters and widespread coal-based heating maintain a persistent NO_2
 361 contribution even under an overall decline in emissions.

362 In contrast to the other pollutants, O_3 increased in all regions, but with different
 363 intensities. The strongest increase was observed in the Fenwei Plain ($+14.3\%$), whereas
 364 in most regions it ranged from 7% to 9% , and was minimal in Northeast China ($+2.7\%$;
 365 Figure 2b). Several factors may contribute to this regional contrast. In the Fenwei Plain,
 366 the sharp reduction in aerosol loading (SO_2 -80.5% , $PM_{2.5}$ -37.4%) may have



367 contributed to more favorable photochemical conditions for O₃ increases (Lou et al.,
 368 2014; Li et al., 2019b; Ivatt et al., 2022). In Northeast China, by contrast, low solar
 369 radiation during the cold season limits photochemistry, while high NO concentrations
 370 may additionally titrate O₃ through the reaction $\text{NO} + \text{O}_3 \rightarrow \text{NO}_2 + \text{O}_2$, explaining the
 371 minimal O₃ increase (Sillman, 1999; Li et al., 2021).

372 Figure 2c shows that the O₃ increase was not uniformly related to reductions in
 373 different pollutants. The highest O₃ penalty ratios were observed in the Fenwei Plain,
 374 reaching 5.1 for PM₁₀, 3.8 for PM_{2.5} and 4.7 for NO₂, whereas values in Northeast China
 375 and PRD were much lower (0.8–2.4). This indicates that in the Fenwei Plain, the
 376 relationship between conventional pollutant reductions and O₃ increase was strongest
 377 precisely where primary emissions declined most sharply and photochemical
 378 conditions became most favorable for O₃ formation.

379 The city-level distribution of O₃ changes shows substantial intra-regional
 380 heterogeneity (Figure 2d). The largest spread was observed in Northwest China, with a
 381 standard deviation of 12.7% and a range from –21.5% to +64.8%, reflecting the
 382 heterogeneity of cities in the arid zone, including the combined influence of natural dust
 383 sources, local industry and differences in photochemical conditions. The broad spread
 384 in Other China, with a standard deviation of 9.4% and a range from –8.4% to +47.8%,
 385 partly reflects the heterogeneous composition of this residual group. In contrast, PRD
 386 (4.4%) and Northeast China (8.7%) showed more compact distributions, indicating a
 387 more homogeneous regional O₃ response.

388 Overall, the regional evolution of air pollution in China was characterized by two
 389 main features. First, PM_{2.5}, PM₁₀, SO₂, NO₂ and CO decreased in all regions, although
 390 the magnitude of these reductions varied depending on source structure and climatic
 391 conditions. Second, O₃ increased almost everywhere, but the strongest increase
 392 occurred in regions where the reduction in aerosol loading was most pronounced,
 393 particularly the Fenwei Plain, and was minimal where photochemical activity is
 394 climatically limited, such as Northeast China. This points to a fundamental difference
 395 between controlling primary and particulate pollution and managing secondary
 396 photochemical pollution.

397 **3.3 Seasonal O₃-centred multipollutant relationships**

398 Seasonal correlation analysis shows that relationships among pollutants changed
 399 substantially throughout the year (Figure 3). Unlike PM_{2.5} and PM₁₀, which exhibited



consistently strong correlations across all seasons ($\rho = 0.85\text{--}0.91$), the relationships between O_3 and other pollutants were strongly season-dependent. This indicates that O_3 is not a simple component of the overall pollution burden; rather, its behavior is linked to other pollutants through seasonally varying photochemical and meteorological conditions (Liao et al., 2024; Yang et al., 2024). However, it should be emphasized that the correlation analysis does not establish causal relationships. The results presented here describe statistical associations, which may reflect both common sources and indirect chemical interactions.

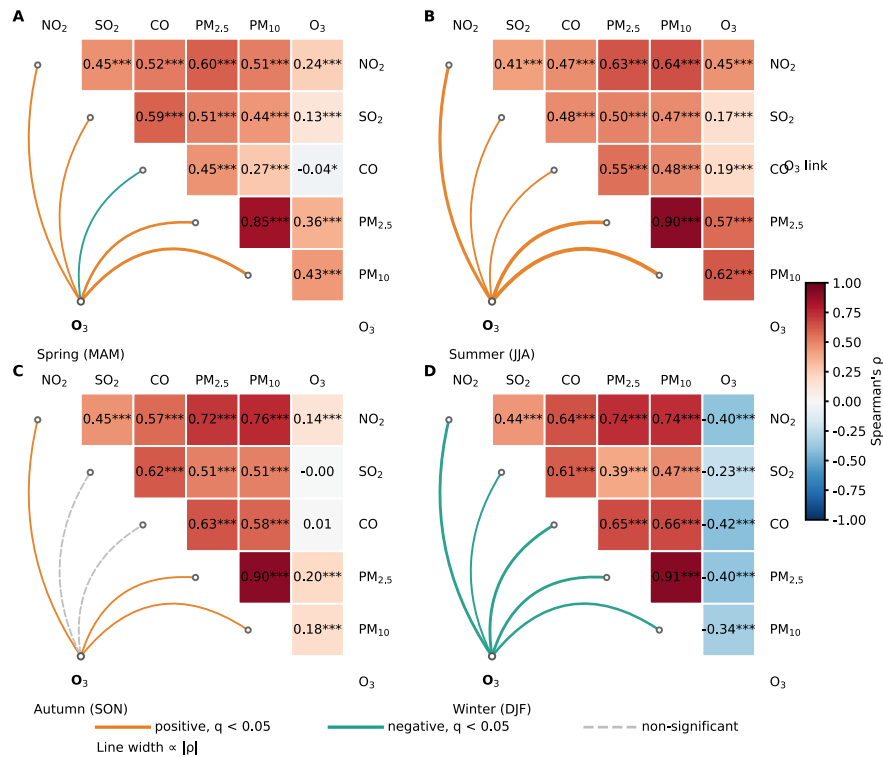


Figure 3. Seasonal relationships between O_3 and other air pollutants across Chinese cities.

As shown in Figure 3, O_3 was positively associated with most pollutants in spring. The strongest relationships were observed with PM_{10} ($\rho = 0.43$, $q < 0.001$) and $\text{PM}_{2.5}$ ($\rho = 0.36$, $q < 0.001$), whereas the relationship with NO_2 was more moderate ($\rho = 0.24$, $q < 0.001$). The relationship with SO_2 was weak but statistically significant ($\rho = 0.13$, $q < 0.001$), while the relationship with CO was statistically significant but practically very weak ($\rho = -0.04$, $q = 0.014$). At the same time, $\text{PM}_{2.5}$ and PM_{10} showed a very



417 strong mutual correlation ($\rho = 0.85$, $q < 0.001$), indicating common sources and
418 processes. This pattern is consistent with the view that, during spring, O_3 remains partly
419 linked to the overall pollution burden, especially particulate pollution, although the
420 direction of this relationship requires further investigation.

421 In summer, positive relationships between O_3 and other pollutants became stronger.
422 The highest coefficients were observed for PM_{10} ($\rho = 0.62$, $q < 0.001$), $PM_{2.5}$ ($\rho = 0.57$,
423 $q < 0.001$) and NO_2 ($\rho = 0.45$, $q < 0.001$). Relationships with CO and SO_2 were also
424 positive but weaker, with $\rho = 0.19$ and $\rho = 0.17$, respectively. The correlation between
425 $PM_{2.5}$ and PM_{10} remained very high ($\rho = 0.90$). This pattern is consistent with the
426 expected enhancement of photochemical activity during summer, when high solar
427 radiation and temperature may promote both O_3 formation and secondary aerosol
428 production (Wang et al., 2016; Chen et al., 2024), creating conditions for the joint
429 variability of these pollutants.

430 In autumn, relationships between O_3 and other pollutants weakened markedly. O_3
431 retained weak positive relationships with $PM_{2.5}$ ($\rho = 0.20$, $q < 0.001$), PM_{10} ($\rho = 0.18$,
432 $q < 0.001$) and NO_2 ($\rho = 0.14$, $q < 0.001$), whereas its relationships with SO_2 and CO
433 became statistically insignificant. The correlation between $PM_{2.5}$ and PM_{10} remained
434 high ($\rho = 0.90$). This indicates that, during autumn, O_3 was less closely associated with
435 combustion-related pollutants, which may reflect changes in photochemical conditions
436 and reduced solar radiation.

437 The most pronounced contrast was observed in winter. During this season,
438 relationships between O_3 and all conventional pollutants became negative and
439 statistically significant. The strongest negative correlations were observed with CO (ρ
440 $= -0.42$, $q < 0.001$), NO_2 ($\rho = -0.40$, $q < 0.001$) and $PM_{2.5}$ ($\rho = -0.40$, $q < 0.001$).
441 Relationships with PM_{10} and SO_2 were also negative, with $\rho = -0.34$ and $\rho = -0.23$,
442 respectively. The correlation between $PM_{2.5}$ and PM_{10} remained high ($\rho = 0.91$). This
443 winter structure indicates opposite behavior of O_3 and primary/particulate pollutants
444 during the cold season. It is consistent with the known mechanism of NO titration, in
445 which high NO concentrations associated with NO_2 can destroy O_3 , as well as with
446 weakened photochemical O_3 formation under low solar radiation (Sillman, 1999; Li et
447 al., 2021).

448 Overall, Figure 3 shows a clear seasonal reversal in pollutant relationships: from
449 positive relationships between O_3 and other pollutants in spring and summer, to weaker
450 relationships in autumn and negative relationships in winter. At the same time,



relationships between PM_{2.5} and PM₁₀ remained consistently strong across all seasons, confirming their common nature. The seasonal dynamics of O₃ partly explain why its increase, identified at the national and regional levels, does not follow the dynamics of PM_{2.5}, PM₁₀, SO₂, NO₂ and CO. Although correlation analysis cannot establish causal relationships, the observed pattern is consistent with the idea that seasonal photochemical and meteorological conditions substantially modify the relationship between O₃ and primary and particulate pollutants. This highlights that O₃ management requires a separate approach and cannot be reduced to the control of particulate and primary gaseous pollution.

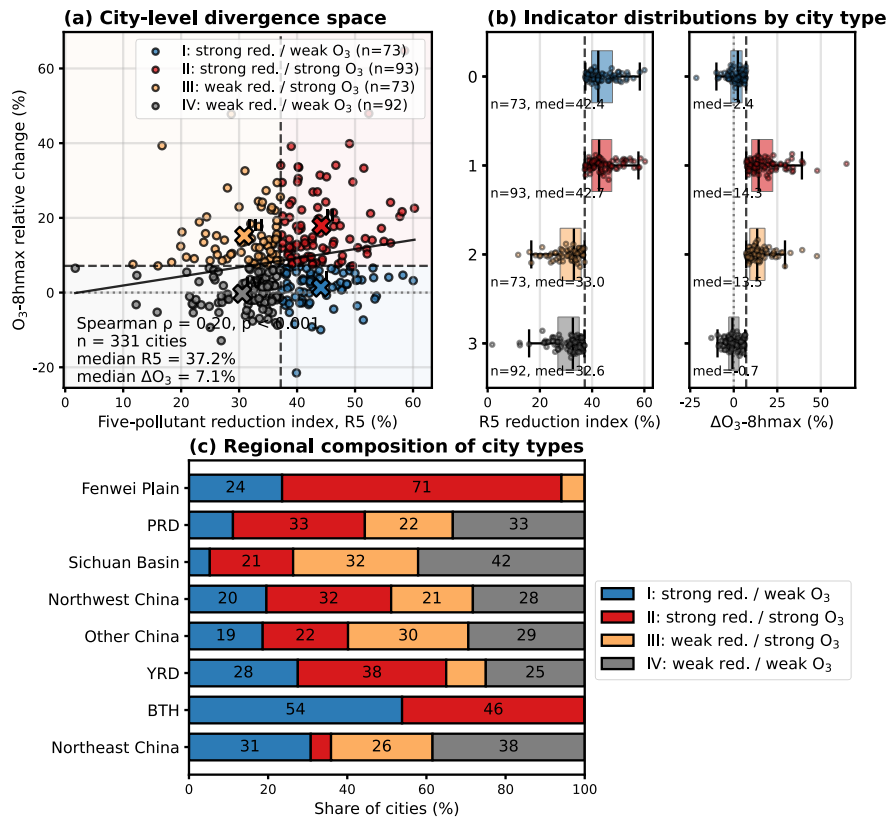
3.4 City-level divergence between conventional pollutant reduction and O₃ response

The use of the city-level divergence framework shows that reductions in conventional pollutants were not accompanied by a uniform O₃ response at the level of individual cities (Figure 4). For 331 cities with sufficient observations in the early and late periods, the relationship between the five-pollutant reduction index and the change in O₃ was positive but weak (Spearman's $\rho = 0.197$, $p = 0.0003$; Figure 4a). The spread of values was substantial: cities with comparable reductions in PM_{2.5}, PM₁₀, SO₂, NO₂ and CO could exhibit either weak O₃ changes or pronounced O₃ increases. This indicates that the magnitude of conventional pollutant reduction alone does not explain the city-level O₃ response.

Based on the median values of the five-pollutant reduction index (37.2%) and O₃ change (7.1%), cities were divided into four types (Figure 4b). Type I cities (22.1% of cities) showed strong reductions in conventional pollutants, with a median reduction of 42.4%, and a weak O₃ response, with a median O₃ change of -2.4%. Type II cities (28.1%) also showed strong reductions, with a median reduction of 42.7%, but experienced strong O₃ increases, with a median change of 14.3%. Type III cities (22.1%) showed weaker reductions in conventional pollutants, with a median reduction of 33.0%, but strong O₃ increases, with a median change of 13.5%. Type IV cities (27.8%) showed weaker reductions, with a median reduction of 32.6%, and a weak O₃ response, with a median change of -0.7%. The key difference among the types was not the magnitude of conventional pollutant reduction, since Type I and Type II were nearly identical in their reduction index, but rather the O₃ response under comparable improvements in PM and primary gaseous pollutants.



484



485

486 **Figure 4.** City-level divergence between conventional pollutant reduction and O₃
487 response.

488 The physicochemical interpretation of these types should be regarded as hypothetical,
489 because the available data do not allow direct diagnosis of the chemical regime of O₃
490 formation (Jin and Holloway, 2015; Jin et al., 2017). Type I may represent cities where
491 strong reductions in conventional pollutants were not accompanied by enhanced O₃
492 increases, which may be related to more effective ventilation, lower photochemical
493 sensitivity or more balanced changes in precursor-related pollution (Ivatt et al., 2022).
494 Type II characterizes cities where strong reductions in PM, SO₂ and other conventional
495 pollutants were accompanied by pronounced O₃ increases; this pattern may be related
496 to reduced aerosol-related attenuation of solar radiation and enhanced photochemical
497 conditions (Verstraeten et al., 2015; Wang et al., 2022), although the role of NO_x-VOC
498 sensitivity cannot be established without additional data. Type III shows that strong O₃



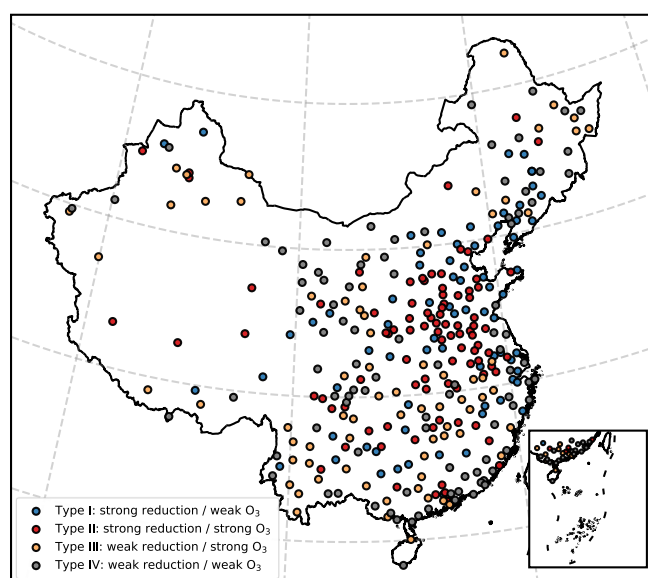
499 increases can occur even under more moderate reductions in conventional pollutants,
500 which may indicate the importance of local meteorological conditions, insolation,
501 ventilation and regional transport (Qu et al., 2021). Type IV, by contrast, includes cities
502 with weaker reductions in conventional pollutants and a weak O₃ response, which may
503 reflect more limited photochemical activity, different baseline pollution levels or less
504 pronounced changes in precursor conditions (Yang et al., 2025). Therefore, these types
505 should not be interpreted as a direct diagnosis of O₃ chemical regimes, but rather as an
506 empirical city-level typology indicating different trajectories of O₃ response under
507 reductions in conventional pollutants. Confirmation of the underlying chemical
508 mechanisms would require direct VOC measurements, speciated emission inventories,
509 radiation and photolysis observations, and chemical-transport modelling capable of
510 diagnosing local NO_x–VOC sensitivity.

511 The regional composition of the types shows pronounced spatial heterogeneity
512 (Figure 4c). The Fenwei Plain was most clearly skewed toward Type II. Here, strong
513 reductions in conventional pollutants were often accompanied by strong O₃ increases.
514 This is consistent with the regional analysis in Section 3.2, where the sharp reduction
515 in aerosol loading in the coal belt may have contributed to enhanced photochemical
516 conditions under basin-like topography (Lin et al., 2023). BTH also showed a high
517 share of Type II cities, although Type I remained slightly more common. Northeast
518 China, by contrast, was more strongly represented by Type IV and Type I, indicating
519 weak O₃ increases under different levels of conventional pollutant reduction, likely
520 reflecting limited photochemical activity in a cold climate and NO titration (Li et al.,
521 2021). Northwest China and Other China showed a more mixed structure, including
522 notable shares of both ozone-increase types and weak-response types, reflecting the
523 heterogeneity of cities within these large regions.

524 The spatial distribution of city-level divergence types further shows that the
525 identified types were not randomly distributed across China (Figure 5). Type II cities,
526 where strong reductions in conventional pollutants were accompanied by strong O₃
527 increases, formed the most pronounced cluster in the North China and central-eastern
528 belt, including BTH, the Fenwei Plain and adjacent inland areas. Individual Type II
529 cities also occurred in Northwest China and some inland southern regions. Type I cities,
530 characterized by strong reductions in conventional pollutants and a weak O₃ response,
531 were more common in eastern and coastal China and also accounted for a substantial
532 share of BTH cities. Type III cities, where strong O₃ increases occurred under weaker



533 reductions in conventional pollutants, were more spatially mosaic and occurred in
534 Northwest China, southern regions and inland areas. Type IV cities, characterized by
535 weaker reductions in conventional pollutants and a weak O₃ response, were more
536 frequently observed in Northeast China, western regions and other peripheral cities.
537 Importantly, different types often coexisted within the same broad regions, indicating
538 that the city-level O₃ response was determined not only by regional background
539 conditions but also by local factors.



540

541 **Figure 5.** Spatial distribution of city-level divergence types across Chinese cities.

542 This analysis complements the seasonal results in Section 3.3. Seasonal correlations
543 show that relationships between O₃ and other pollutants vary substantially throughout
544 the year, whereas the city-level divergence analysis identifies the cities and conditions
545 under which O₃ increased despite the overall decline in conventional pollutants.
546 Together, these results indicate that air-quality improvement in Chinese cities was not
547 a single coherent process: in many cities, reductions in PM_{2.5}, PM₁₀, SO₂, NO₂ and CO
548 diverged from the O₃ response.

549 The results show that reductions in conventional pollutants were not necessarily
550 accompanied by weak changes in O₃. Some cities exhibited relatively balanced
551 improvement, whereas others experienced strong O₃ increases under comparable
552 reductions in PM and primary gaseous pollutants. This highlights the need to move
553 beyond assessing air quality only through reductions in particulate and primary gaseous



pollution toward city-specific and regionally differentiated multipollutant control strategies that account for the risk of O₃ enhancement.

3.5 Meteorological influence on O₃ change

Meteorological conditions changed appreciably between the early (2015–2017) and late (2023–2025) periods (Figure 6a). Across 318 cities with sufficient paired coverage, mean near-surface air temperature increased by 0.73 °C (95% CI: 0.68–0.77 °C) and daily maximum temperature increased by 0.88 °C (95% CI: 0.83–0.93 °C). Mean relative humidity decreased by 1.65 percentage points (95% CI: –1.92 to –1.37 percentage points), whereas the mean change in wind speed was small (–0.029 m s^{–1}, 95% CI: –0.066 to +0.012 m s^{–1}). Thus, the late period was characterized by widespread warming and generally drier conditions, while no comparably coherent national change in wind speed was evident.

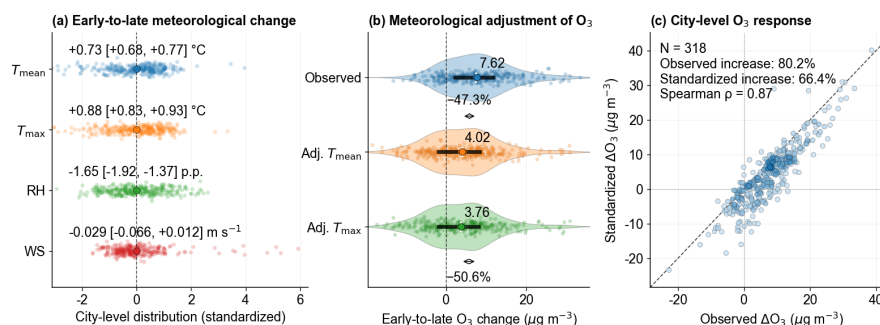


Figure 6. Meteorological changes and meteorologically standardized O₃ responses across Chinese cities. **a**, City-level early-to-late changes in mean air temperature (T_{mean}), maximum air temperature (T_{max}), relative humidity (RH) and wind speed (WS) between 2015–2017 and 2023–2025; values indicate national mean changes and 95% confidence intervals. **b**, Distributions of observed and meteorologically standardized early-to-late O₃ changes using T_{mean} - and T_{max} -based models. **c**, Relationship between observed and T_{mean} -standardized city-level O₃ changes; the dashed line denotes the 1:1 relationship.

City-level temperature changes showed only weak associations with the O₃ response. The absolute early–late O₃ change was positively correlated with changes in T_{mean} (Spearman’s $\rho = 0.129$, $p = 0.022$) and T_{max} ($\rho = 0.144$, $p = 0.010$), whereas the association with RH was negative ($\rho = -0.190$, $p < 0.001$). However, after T , RH and WS changes were considered simultaneously, neither T_{mean} (standardized $\beta = 0.033$, $p = 0.554$) nor T_{max} ($\beta = 0.022$, $p = 0.746$) showed an independent association with the city-level O₃ response. The RH change remained significant ($\beta \approx -0.16$, $p = 0.006$),



581 although the combined meteorological changes explained only a small fraction of the
 582 between-city variation in O_3 change ($R^2 \approx 0.03$). This is consistent with previous China-
 583 wide attribution studies showing that meteorological variability can substantially
 584 modulate surface O_3 , but cannot fully explain its sustained increase (Li et al., 2020; Liu
 585 et al., 2023).

586 Meteorological standardization nevertheless showed that interannual meteorological
 587 variability materially affected the magnitude of the national early–late O_3 difference
 588 (Figure 6b). Across the 318 cities, the observed mean O_3 increase was $7.62 \mu\text{g}/\text{m}^3$. After
 589 standardization using T_{mean} , RH and WS, the increase was reduced to $4.02 \mu\text{g}/\text{m}^3$,
 590 corresponding to a 47.3% reduction in the observed early–late difference. The T_{max} -
 591 based sensitivity model produced a similar standardized increase of $3.76 \mu\text{g}/\text{m}^3$,
 592 corresponding to a 50.6% reduction. The proportion of cities with increasing O_3
 593 decreased from 80.2% in the observed data to 66.4% after T_{mean} -based meteorological
 594 standardization (Figure 6c), and to 65.7% using T_{max} . Excluding cities represented by
 595 meteorological stations more than 100 km away yielded nearly identical results.
 596 Meteorological normalization has similarly been used to separate part of the
 597 meteorological influence on O_3 variability and thereby improve the interpretation of
 598 changes associated with precursor conditions (Qiu et al., 2025).

599 These findings indicate that the magnitude of the observed early–late O_3 difference
 600 was substantially reduced after standardizing the meteorological variables represented
 601 in the models, but was not fully accounted for by them. Approximately one-half of the
 602 mean early–late O_3 difference was removed by standardizing the meteorological
 603 variables included in the model, while a substantial positive O_3 increase remained
 604 across most cities. The residual increase is therefore consistent with an important role
 605 for non-meteorological changes, including changes in precursor emissions, aerosol
 606 loading and atmospheric chemistry. Because solar radiation, boundary-layer dynamics
 607 and VOC concentrations were not explicitly represented in the meteorological
 608 standardization, the remaining O_3 change should not be interpreted as a direct
 609 quantitative measure of emission-driven or chemical effects.

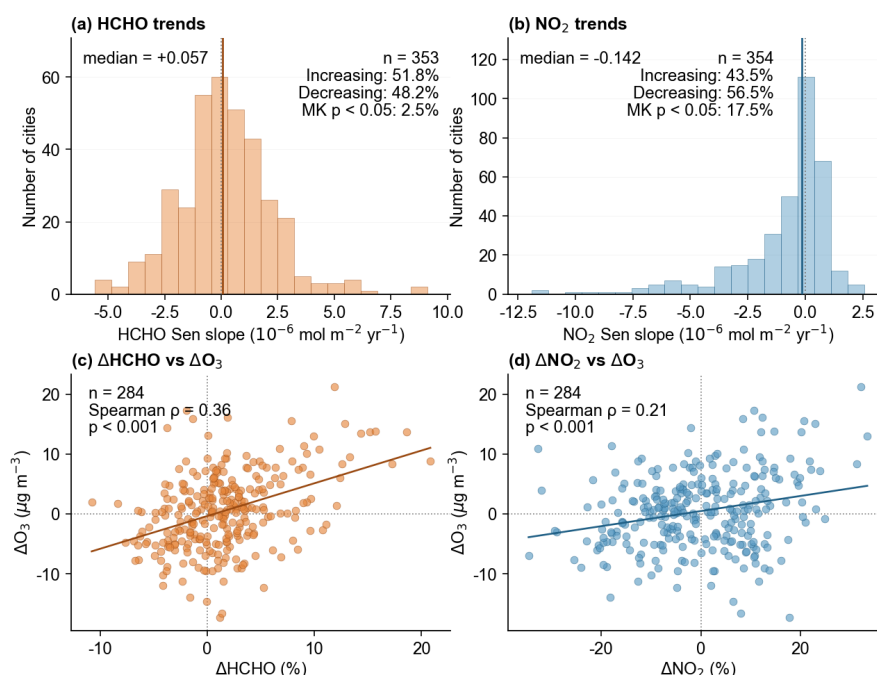
610 **3.6 Satellite evidence of changing ozone-precursor conditions**

611 Satellite observations provided an additional constraint on recent changes in ozone-
 612 precursor conditions across Chinese cities. TROPOMI HCHO and tropospheric NO_2
 613 were analysed for May–September during 2019–2025, with 98.4% of city–year records
 614 satisfying the joint sampling criterion of at least 60 valid retrieval days for both species.



615 Warm-season HCHO showed little evidence of a coherent nationwide monotonic trend.
 616 Among 353 cities with sufficient temporal coverage, city-level Sen slopes were positive
 617 in 51.8% and negative in 48.2%, with a near-zero median slope of 5.7×10^{-8} mol/m²/yr;
 618 only 2.5% of cities showed nominally significant Mann–Kendall trends (Fig. 7a).
 619 Nevertheless, comparison of 2019–2021 with 2023–2025 indicated a modest shift
 620 toward higher HCHO, with a median increase of 1.33% and increases occurring in 63.6%
 621 of evaluated cities. Consistent with previous satellite studies using HCHO columns to
 622 constrain VOC-related variability and NMVOC emissions over China (Shen et al., 2019;
 623 Feng et al., 2024), these results indicate substantial spatial heterogeneity rather than a
 624 coherent nationwide HCHO trend.

625 Tropospheric NO₂ exhibited a more pronounced tendency toward decline. Negative
 626 Sen slopes occurred in 56.5% of 354 cities, compared with 43.5% showing positive
 627 slopes, and the median slope was -1.42×10^{-7} mol/m²/yr (Fig. 7b). Nominally significant
 628 Mann–Kendall trends occurred in 17.5% of cities. The median change between 2019–
 629 2021 and 2023–2025 was -0.72% , although the direction and magnitude of change
 630 remained spatially heterogeneous.



631
 632 **Figure 7.** Satellite-derived precursor trends and their associations with surface O₃
 633 changes across Chinese cities. **a, b,** Distributions of city-level Sen slopes in May–



634 September TROPOMI HCHO and tropospheric NO₂ during 2019–2025. Dotted vertical
 635 lines indicate zero trend and solid coloured lines indicate median Sen slopes.
 636 Percentages summarize cities with increasing and decreasing slopes and nominally
 637 significant Mann–Kendall trends ($p < 0.05$). **c, d**, Relationships between percentage
 638 changes in city-level HCHO and tropospheric NO₂ and the concurrent absolute change
 639 in May–September surface O₃ between 2019–2021 and 2023–2025. Δ HCHO and Δ NO₂
 640 are expressed relative to their respective 2019–2021 means, whereas Δ O₃ is expressed
 641 as the absolute concentration difference ($\mu\text{g}/\text{m}^3$). Solid lines show linear fits for visual
 642 guidance; reported statistics are Spearman rank correlations.

643 Importantly, spatial differences in precursor changes were associated with
 644 concurrent changes in surface O₃. Between 2019–2021 and 2023–2025, city-level
 645 Δ HCHO was positively associated with Δ O₃ (Spearman $\rho = 0.36$, $p < 0.001$, $n = 284$;
 646 Fig. 7c), indicating that cities with larger positive HCHO changes generally
 647 experienced larger increases, or smaller decreases, in warm-season O₃. Δ NO₂ was also
 648 positively associated with Δ O₃, although the relationship was weaker ($\rho = 0.21$, $p <$
 649 0.001 , $n = 284$; Fig. 7d). This association should be interpreted cautiously because the
 650 O₃ response to NO_x is nonlinear and regime-dependent, varying among VOC-limited,
 651 transitional and NO_x-limited conditions and potentially changing over time within the
 652 same region (Sillman, 1999; Li et al., 2019a; Wang et al., 2025b). The positive spatial
 653 association between Δ NO₂ and Δ O₃ therefore should not be interpreted as evidence that
 654 increasing NO₂ directly enhances O₃ formation. As a robustness check, associations
 655 based on city-specific Sen slopes over 2019–2025 were also positive for HCHO ($\rho =$
 656 0.34 , $p < 0.001$) and NO₂ ($\rho = 0.26$, $p < 0.001$). Together, these results indicate spatially
 657 heterogeneous precursor evolution, with HCHO variability showing the clearest city-
 658 level association with recent O₃ changes rather than a coherent nationwide increase in
 659 HCHO. The comparatively stronger HCHO association is consistent with previous
 660 evidence that VOC-related variability can remain important for the evolution of surface
 661 O₃ in China even as NO_x and particulate pollution decline (Li et al., 2019b; Li et al.,
 662 2020).

663 4. Conclusions

664 This study analyzed the multipollutant evolution of air quality in Chinese cities
 665 during 2015–2025 based on long-term observations of PM_{2.5}, PM₁₀, SO₂, NO₂, CO and
 666 O₃. The results show that substantial air-quality improvement occurred in Chinese cities



667 over the study period. Annual mean concentrations of PM_{2.5}, PM₁₀, SO₂, NO₂ and CO
 668 decreased by 40.8%, 35.3%, 70.3%, 36.2% and 45.0%, respectively. These changes
 669 were spatially coherent and statistically significant in most cities, indicating a persistent
 670 national decline in particulate and primary gaseous pollution.

671 However, this improvement was not accompanied by a simultaneous decrease in O₃.
 672 In contrast to the other pollutants, O₃ increased by 10.3% at the national level. Positive
 673 Sen's slopes were observed in 76.1% of cities, although statistically significant O₃
 674 trends were detected in only 19.9%, demonstrating that the predominant direction of O₃
 675 change was positive but considerably less spatially coherent than the declines in
 676 conventional pollutants. This shows that air-quality improvement in China was not a
 677 uniform decline of all pollutants. Instead, reductions in PM and primary gaseous
 678 pollutants were accompanied by an intensification of secondary photochemical
 679 pollution.

680 Meteorological changes contributed importantly to this O₃ increase. Across 318
 681 cities with sufficient paired data, mean near-surface air temperature increased by 0.73 °C
 682 and relative humidity decreased by 1.65 percentage points between 2015–2017 and
 683 2023–2025. Meteorological standardization reduced the observed mean O₃ increase
 684 from 7.62 to 3.76–4.02 µg/m³, but did not eliminate it: approximately two-thirds of
 685 cities still showed increasing O₃ after standardization. Thus, changing meteorological
 686 conditions amplified the observed O₃ difference, but substantial O₃ divergence
 687 persisted after accounting for the meteorological variability represented in the analysis.

688 Satellite observations provided complementary evidence that recent O₃ evolution
 689 was also associated with spatially heterogeneous changes in precursor conditions.
 690 Warm-season TROPOMI HCHO showed no coherent nationwide trend during 2019–
 691 2025, whereas tropospheric NO₂ exhibited a greater tendency toward decline.
 692 Nevertheless, city-level changes in HCHO were positively associated with concurrent
 693 changes in O₃ between 2019–2021 and 2023–2025 (Spearman $\rho = 0.36$, $p < 0.001$),
 694 with a weaker positive association for tropospheric NO₂ ($\rho = 0.21$, $p < 0.001$). These
 695 relationships indicate that recent O₃ divergence cannot be attributed to meteorology
 696 alone and was accompanied by spatially heterogeneous precursor evolution,
 697 particularly VOC-related variability represented by HCHO. Because satellite HCHO is
 698 an indirect proxy rather than a direct measure of VOC emissions, these associations
 699 should not be interpreted as evidence of a uniform nationwide increase in VOC
 700 emissions.



701 Regional analysis showed that ozone divergence exhibited pronounced spatial
702 heterogeneity. The Fenwei Plain experienced the strongest O₃ increase despite
703 substantial decreases in SO₂, CO and PM, whereas Northeast China showed the
704 smallest O₃ increase. Seasonal analysis further showed that relationships between O₃
705 and other pollutants varied substantially throughout the year: in spring and summer, O₃
706 was positively associated with PM and NO₂-related pollution, whereas these
707 relationships weakened in autumn and became negative in winter. This highlights the
708 seasonal dependence of O₃ response and its distinction from conventional pollutants.

709 At the city level, the proposed city-level divergence typology showed that
710 comparable reductions in conventional pollutants can be accompanied by
711 fundamentally different O₃ responses. Type I and Type II cities had nearly identical
712 median values of the five-pollutant reduction index but differed sharply in O₃ change:
713 weak decrease or almost no increase in Type I, and strong increase in Type II. The
714 spatial distribution of the types showed that ozone-divergent cities were more
715 frequently concentrated in northern and inland regions, whereas balanced-improvement
716 and weak-response types were more common in eastern, northeastern and peripheral
717 areas. This confirms that O₃ divergence is not only a national or regional issue, but also
718 a city-specific problem.

719 Overall, the results show that reductions in PM_{2.5}, PM₁₀, SO₂, NO₂ and CO do not
720 guarantee mitigation of O₃ pollution. The next stage of air-quality management in China
721 should move beyond single-pollutant or PM-focused control and shift toward city-
722 specific and regionally differentiated multipollutant management. Particular attention
723 should be paid to the coordinated management of PM, NO_x and VOC precursors,
724 seasonal photochemical conditions and local meteorological constraints. Because the
725 present analysis remains observational, future work should combine direct VOC
726 measurements, speciated emission inventories, radiation and photolysis observations,
727 and chemical-transport modelling to distinguish the relative contributions of precursor
728 changes, aerosol–radiation effects and meteorological variability to city-level O₃
729 divergence.

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734 **Author contributions.** V.H: Formal analysis, Investigation, Writing-Original Draft.
 735 M.F: Conceptualization, Methodology, Funding Acquisition, Writing-Review &
 736 Editing. T.T: Methodology, Writing-Review & Editing. L.Z: Conceptualization,
 737 Writing-Review & Editing, Supervisor. All authors discussed the results, contributed
 738 to the final manuscript, and approved the submitted version.

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