

Research on Coordinated Control Strategies of PM_{2.5} and Ozone in Urban Areas of China

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Abstract. China's clean-air campaign has now run for over a decade. By 2025, the third and most recent action plan had driven national PM_{2.5} concentrations down by roughly 60% based on level in the year of 2013. Yet the same period saw a different story for ozone. While PM_{2.5} fell, ground-level ozone (O₃) kept climbing, and by 2025 it had overtaken most other pollutants to become the country's second-biggest air quality concern. The reason why these pollutants behave so differently under the same emission controls are not fully investigated. The link of these two pollutants is not straightforward. PM_{2.5} can suppress ozone by shading the surface from solar radiation and by scavenging free radicals on aerosol surface. Yet when PM_{2.5} drops, both brakes are released. Volatile organic compounds (VOCs) are the core of this problem. They fuel ozone through photochemical chain reactions with NO_x, and they also condense into secondary organic aerosols (SOA). Summer megacity atmosphere is typically VOC-limited. In this regime, cutting NO_x without touching VOCs is risky for less NO means less titration of O₃, so ozone can rise even as NO_x emission fall. This nonlinearity makes single-pollutant control not effective as expected. Between 2015 and 2019, PM_{2.5}-related deaths fell by roughly 410,000, but ozone-related deaths rose by 80,000 over the same period. What is needed is not a single pollution abatement plan but a patchwork of approaches. Each region needs its own VOC-NO_x mixing control policy for the different photochemical regimes. The findings of this study are grounded in Chinese data, but the underlying chemistry and PM_{2.5}-O₃ coupling works globally. Countries now grappling with both particulate and ozone pollution face the same issue. So, the control strategy in China can provide reference for these countries.

Keywords: PM_{2.5}, ozone (O₃), coordinated control, volatile organic compounds (VOC), nonlinear response

1. Introduction

1.1. China's achievements in controlling fine particulate matter in the air

Since 2013, the Chinese government has launched three successive nationwide clean air initiatives: the *Air Pollution Prevention and Control Action Plan* (2013–2017) [1], the *Three-Year Action Plan for Winning the Blue-Sky Defense Battle* (2018–2020) [2] and the *Action Plan for Continuously Improving Air Quality* (2023–2025) [3, 4].

By 2025, the average annual $PM_{2.5}$ concentration across 339 prefecture-level cities in China had fallen by approximately 59%. Meanwhile, the frequency of heavily polluted days dropped by more than 90% [5]. Beijing serves as a representative example: its annual $PM_{2.5}$ concentration declined from $89.5 \mu\text{g}/\text{m}^3$ in 2013 to $32 \mu\text{g}/\text{m}^3$ in 2025. This clearly proves the effectiveness of regional joint air pollution prevention and control. These policies have resulted in a substantial cumulative reduction in national $PM_{2.5}$ concentrations, potentially preventing hundreds of thousands of premature deaths [6]. Even with such progress, challenges remain. The national average $PM_{2.5}$ concentration in April 2026 was $24.0 \mu\text{g}/\text{m}^3$ —still above the World Health Organization's (WHO) Interim Target-1. It means stricter pollution control actions are still necessary [7].

1.2. Ground-level O_3 has emerged as a major air pollutant

While $PM_{2.5}$ pollution still fails to meet relevant standards, ground-level ozone (O_3) has emerged as the second major air pollutant, and its concentration shows a slow rising trend [8].

As stated by the Ministry of Ecology and Environment of the People's Republic of China at an environmental press conference, O_3 pollution frequently reaches its peak in late spring on clear, cloudless days. Intense solar radiation, high temperatures, and low humidity make pollution even worse throughout summer and early autumn [9]. Between January and May in 2025, the national O_3 concentration reached $140 \mu\text{g}/\text{m}^3$, up 0.7% from the same period in 2024. Over 80 cities exceeded the national O_3 standard during this period, and pollution was generally more severe in the north than in the south [10].

The 2025 *China Environmental and Ecological Status Report* shows that the share of cities with O_3 concentrations between 120 and to $150 \mu\text{g}/\text{m}^3$ increased by 49.9% compared to 2024, as illustrated in Figure1. Cities that exceeded the daily maximum 8-hour O_3 limits ($160 \mu\text{g}/\text{m}^3$) were also more numerous than in 2024. These cities are mainly located in the Beijing-Tianjin-Hebei region, the Yangtze River Delta, and the Pearl River Delta regions.

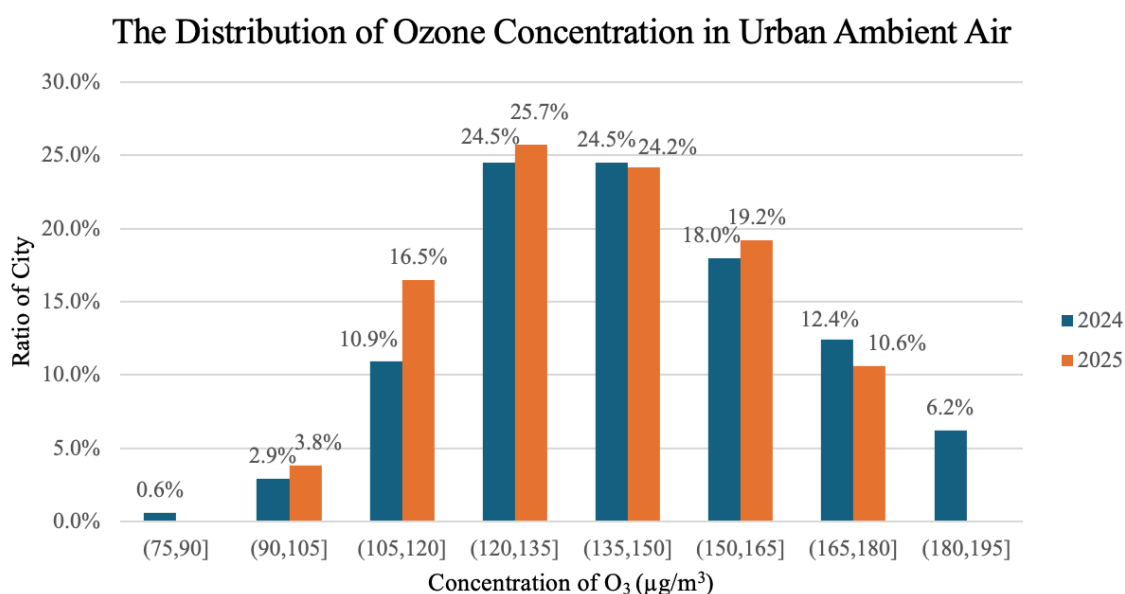


Figure 1. The distribution of ambient ozone concentrations across Chinese cities in 2025, with year-on-year changes relative to 2024. Data are derived from the 2025 China Environmental and Ecological Status Report [11]

The divergent trends in $PM_{2.5}$ and O_3 cannot be explained by emission reductions alone. Their shared precursor sources and complex atmospheric chemistry link the two pollutants. The *2025 China Environmental and Ecological Status Report* indicates that ambient $PM_{2.5}$ concentrations in China have exhibited a sustained decline, while ground-level O_3 concentration rising at the same time. This dual-pollutant challenge-simultaneous $PM_{2.5}$ reduction and O_3 control-represents the most pressing issue in China's currently faced by China in terms of improving air quality [11]. A global air pollution study noted that China's ambient air quality standards for $PM_{2.5}$ and O_3 are less stringent than those of many developed countries [12]. For instance, WHO recommends an annual mean $PM_{2.5}$ guideline value of $5 \mu\text{g}/\text{m}^3$, whereas China's national standard is $35 \mu\text{g}/\text{m}^3$ -higher than the U.S. EPA's standard ($12 \mu\text{g}/\text{m}^3$) and the European Union's target value ($25 \mu\text{g}/\text{m}^3$) EU [12]. In addition, Table1 shows that the maximum 8-hour average daily concentration of O_3 level in China, at $160 \mu\text{g}/\text{m}^3$, is also higher than the $100 \mu\text{g}/\text{m}^3$ standard set by the U.S. EPA, the $120 \mu\text{g}/\text{m}^3$ standard set by the EU, and the $60 \mu\text{g}/\text{m}^3$ standard set by the WHO. Thus, for China, the concurrent challenges of $PM_{2.5}$ and O_3 pollution remain critically severe.

Table 1. Ambient air quality standards for major pollutants across different countries and organizations

Pollutant ($\mu\text{g}/\text{m}^3$)	EU	U.S. EPA	WHO	Russian Federation	India	China	Japan	Australia
PM_{10}	40	-	15	40	60	50	-	-
$PM_{2.5}$	25	12	5	25	40	35	15	8
NO_2	40	30	10	40	40	40	-	60
Ozone	120	100	60	-	100	160	-	170

1.3. $PM_{2.5}$ and O_3 combined pollution and common precursor substance

The combined pollution of $PM_{2.5}$ and O_3 is mainly driven by the emissions of a precursor pollutants, particularly volatile organic compounds (VOCs). VOCs are not only key precursors of organic aerosols in $PM_{2.5}$ but also the primary drivers of photochemical O_3 formation. When VOCs are oxidized in the atmosphere, they produce reactive intermediates and oxidation products that drive the formation of $PM_{2.5}$ and O_3 pollution. The two pollutants also interact: aerosols can slow O_3 formation by blocking solar radiation, yet they can also redirect reactions through heterogeneous chemistry and by changing the photolysis of key species. This is why VOCs reduction have become a critical target for the control of $PM_{2.5}$ and ground-level O_3 together. China's air quality policies have moved from pollutant-by-pollutant control toward joint management of multiple pollutants [13]. Putting this joint control into practice is still difficult. The atmospheric chemistry is not fully understood, meteorological conditions strongly modulate both pollutants, pollution is routinely transported across regions, and climate mitigation strategies remain poorly integrated with air pollution control.

1.4. Aim of this research

This paper provides a systematic review of coordinated $PM_{2.5}$ and O_3 control in urban China, with five main objectives: (1) examining the physicochemical basis of $PM_{2.5}$ - O_3 coupling; (2) interpreting the divergent trends observed-including both synchronous reductions and opposing patterns such as falling $PM_{2.5}$ alongside rising O_3 ; (3) analyzing the spatiotemporal distribution of both pollutants

across China's major city clusters; (4) assessing whether $\text{PM}_{2.5}$ -focused policies have generated co-benefits or unintended O_3 concentration increases; (5) identifying critical uncertainties and outlining future research priorities to advance integrated management. The findings are intended not only to support China's ongoing policy refinement but also to offer transferable lessons for other developing countries confronting similar air quality trade-offs.

2. Scientific basis of the interaction between $\text{PM}_{2.5}$ and O_3

2.1. Characteristics of ambient air pollutants ($\text{PM}_{2.5}$, O_3 , VOCs)

In China, $\text{PM}_{2.5}$ as the main air pollutant of fine particulate matter, has seen a significant reduction in pollutant concentration after 10 years of treatment. According to the global air quality guidelines set by the WHO and the domestic research findings, both long-term and short-term $\text{PM}_{2.5}$ pollution have adverse effects on human health, such as respiratory diseases and cardiovascular diseases [6].

Ozone (O_3) is a highly reactive gas composed of three oxygen atoms. It is both a natural and a man-made product that occurs in the Earth's upper atmosphere. Breathing ground-level O_3 can have numerous health effects, such as the induction of respiratory symptoms, a decrease in lung function, and inflammation of the airways [14, 15]. The U.S. EPA has proposed that in the Northern Hemisphere, O_3 pollution will be transported regionally, moving the ground-level O_3 from the cold north to the warmer south. Therefore, the actual range of ozone's influence extends far beyond its source of emission.

VOCs are precursors for O_3 and $\text{PM}_{2.5}$. VOCs, which are volatile organic compounds, have a wide range of sources. These sources include both natural and anthropogenic process. Anthropogenic sources mainly refer to pollutants emitted during manufactured products and daily activities [16]. In urban areas, these mainly include vehicle exhaust emissions, oil product volatilization leakage, industrial emissions, solvent usage emissions, liquefied petroleum gas (LPG) usage, and household products. Under the ultraviolet radiation of strong sunlight, through photochemical reactions, they will generate O_3 can be generated via these sources. The higher the temperature and the longer the exposure time, the greater the amount of volatilization, and the higher the reaction extent, so the O_3 concentration is often higher [17].

2.2. Relationship between $\text{PM}_{2.5}$ and O_3

The relationship between $\text{PM}_{2.5}$ and O_3 concentrations is highly nonlinear and bidirectional. Meteorological factors-including temperature, relative humidity, solar radiation, sunshine duration, wind speed, and planetary boundary layer height-significantly modulate the $\text{PM}_{2.5}$ - O_3 nonlinearity. This is because these variables govern the formation, loss, and dispersion of both pollutants [18]. Generally, under low $\text{PM}_{2.5}$ loading, these particles reduce O_3 primarily via heterogeneous reactions on their surfaces, which suppress ozone production. When the concentration of $\text{PM}_{2.5}$ is high, the particulate matter mainly reduces O_3 by slowing down the photolysis of NO_2 (and other photochemical precursors) and consequently suppressing O_3 formation [19].

2.3. The core role of VOCs in atmospheric chemistry

2.3.1. VOCs affect the atmospheric oxidation capacity

VOCs play a pivotal role in determining the atmospheric oxidation capacity. The atmospheric oxidation capacity is dominated by hydroxyl radicals (OH), and the reaction between VOCs and OH

radicals is the main pathway for the consumption of OH. Meanwhile, the peroxy radicals (RO_2) generated during the oxidation of VOCs participate in the HO_x cycle and affects the regeneration of OH [20]. The photolysis of oxygen-containing volatile organic compounds (OVOCs) represents a crucial source of OH radicals, and this process further amplifies the atmospheric oxidation capacity [21]. The emissions and spatial distribution of VOCs profoundly affect the self-purification ability of the atmosphere and the generation rate of secondary pollutants particularly O_3 by modulating the sources and sinks of OH radicals.

2.3.2. Photochemical chain reactions of VOCs and NO_x in O_3 formation

O_3 forms through a photochemical chain reaction: OH radicals oxidize VOCs into peroxy radicals, which then convert NO to NO_2 , and photolysis of NO_2 produces O_3 [22]. How much O_3 forms depends on the relative abundance of VOCs and NO_x , not on VOC levels alone. The relationship between O_3 and its precursors is therefore non-linear [23]. In the VOC-limited regime, O_3 formation is restricted by VOCs availability. Under these conditions, O_3 levels may rise for less NO is available to titrate O_3 .

2.3.3. Generation of secondary organic aerosols (SOA) via the oxidation of VOCs

Some oxygen-containing organic compounds with relatively low volatility produced during rapid oxidation by hydroxyl radicals (OH) and O_3 promote the formation of both ground-level O_3 and SOA [22, 24, 25]. SOA makes up a large share of $\text{PM}_{2.5}$ mass: 20%-60% globally, and 30%-78% in Chinese cities during pollution episodes [26]. SOA and O_3 also share a chemical origin: both arise from the atmospheric oxidation of VOCs through different pathways.

2.4. Mechanisms linking $\text{PM}_{2.5}$ reduction to O_3 elevation

2.4.1. Radiative effects of $\text{PM}_{2.5}$ on photolysis

$\text{PM}_{2.5}$ also modulates O_3 levels through its radiative effects on photolysis. High $\text{PM}_{2.5}$ loading weakens the solar radiation reaching the ground, resulting in a significant decrease in the photolysis rates of NO_2 and O_3 near the surface. During a haze event in the Pearl River Delta, aerosols reduced near-surface NO_2 and O_3 by 22% and 29%, respectively. Surface O_3 dropped by more than 10 ppb on average, and by as much as 30 ppb in the most polluted area [27]. $\text{PM}_{2.5}$ also scatters sunlight upward, which can speed up photolysis above the planetary boundary layer and alter O_3 at higher altitudes [27]. As $\text{PM}_{2.5}$ declines, stronger surface radiation speeds up photolysis and drives the VOC- NO_x chain reactions to generate more O_3 .

2.4.2. Heterogeneous chemistry on $\text{PM}_{2.5}$ surfaces

$\text{PM}_{2.5}$ also provides a large surface area on which trace gases including NO_x , N_2O_5 , and HO_2 radicals are taken up and heterogeneously converted. These heterogeneous reactions change how radicals are produced and consumed in the atmosphere, thereby affecting the concentration of O_3 [19]. Gaseous HO_2 for example, is readily taken up onto particle surfaces and converted into H_2O_2 or H_2O through self-reaction or catalytic decomposition. This reaction suppresses O_3 formation by disrupting the photochemical chain reactions [28]. So when $\text{PM}_{2.5}$ levels fall, it weakens the heterogeneous reactions, thereby leading to more O_3 production.

2.4.3. The nonlinear response characteristics of O₃ formation to precursor substances

The reduction in O₃ precursor emissions (NO_x and VOCs) does not linearly translate to a decrease in ambient O₃ concentrations, owing to the inherently nonlinear nature of photochemical O₃ formation [29]. The chemical formula for the formation of ground-level ozone is as follows:



The photochemical formation of O₃ proceeds via the following chain reaction mechanism [29, 30]. First, NO₂ is photolyzed by solar radiation (hv) to produce NO and ground-state oxygen atoms O. The generated O atoms rapidly combine with O₂ to form O₃. However, the newly formed O₃ rapidly reacts with NO to regenerate NO₂, thereby consuming O₃. A closed cycle is formed, and the net production of O₃ is zero. In the presence of VOCs, OH radicals initiate VOC oxidation, generating alkyl radicals (R). The R radicals rapidly add O₂ to form peroxy radicals (RO₂) which then oxidize NO to NO₂ and are themselves converted to alkoxy radicals (RO). This step matters because it converts NO to NO₂ without going through the O₃-consuming pathway (Reaction 3). The RO radicals then react with O₂, or decompose and isomerize, yielding carbonyl compounds and regenerating HO₂, which keeps the radical chain running.

This chemistry defines two sensitivity regimes. In summer, the urban air of many megacities falls in the VOC-limited regime (i.e., NO_x-saturated conditions) [23]: RO₂ production is capped by VOC availability, so O₃ formation is limited by VOCs rather than NO_x. Meanwhile, the abundant NO keeps removing O₃ through titration (Reaction 3). Cutting NO_x alone in this regime lowers both NO and NO₂. Less NO₂ means slightly slower O₃ production (Reactions 1-2), but less NO means a much weaker titration sink. And since VOCs are untouched, RO₂ production does not rise, so Reaction 5 (RO₂ + NO → NO₂ + RO) cannot make up the difference. The titration loss shrinks more than the photolysis source, and ambient O₃ goes up, not down. This is why NO_x-only cuts in VOC-limited cities can backfire, and why NO_x and VOCs have to be controlled together.

3. Pollution characteristics of PM_{2.5} and O₃ in major urban agglomerations of China

3.1. Ambient air pollution characteristics of the Beijing-Tianjin-Hebei region

The Beijing-Tianjin-Hebei (BTH) region and its surrounding areas (i.e., the "2+36" cities) are among the most severely PM_{2.5}-polluted regions in China. These regions are characterized by both persistently high PM_{2.5} levels and pronounced seasonal variations. In January 2026, the average PM_{2.5} concentration in this region was 68.8 µg/m³, an increase of 13.2% compared to the previous year, far exceeding the national average (45.2 µg/m³) [31]. In contrast, the O₃ concentration was relatively low in winter (77 µg/m³), but it reached as high as 175 µg/m³ in summer, a 0.6% year-on-year increase, making it one of the regions with the most prominent O₃ pollution in the country [32].

3.2. Ambient air pollution characteristics of the Yangtze River Delta region

The Yangtze River Delta region exhibits a "multi-center, belt-shaped pollution pattern" [33]. In January 2026, the average $PM_{2.5}$ concentration was $51.4 \mu\text{g}/\text{m}^3$, which was down 11.2% compared to the previous year, and average O_3 concentration was $91 \mu\text{g}/\text{m}^3$, which was a year-on-year decrease of 11.7% [10]. However, the O_3 pollution was more prominent in the summer. In July 2025, the average O_3 concentration reached $160 \mu\text{g}/\text{m}^3$, up 11.1% compared to the previous year [32]. The Yangtze River Delta region is undergoing rapid economic development. Dense emissions of VOCs and NO_x are released by industrial facilities and motor vehicles, which drive vigorous photochemical reactions, resulting in the typical characteristic of high O_3 pollution in summer and rapid increase for this region.

3.3. Ambient air pollution characteristics of the Pearl River Delta region

From 2000 to 2022, the concentration of $PM_{2.5}$ in the Pearl River Delta region showed a downward trend, while the concentration of O_3 increased. The concentrations of $PM_{2.5}$ and O_3 in the Pearl River Delta exhibited significant seasonal variations. $PM_{2.5}$ pollution mainly occurred in winter, while the concentration was relatively lower in summer. O_3 pollution was more severe in autumn, and the concentration was the lowest in winter [34]. For spatial distribution, $PM_{2.5}$ shows a decreasing pattern from the northwest to the southeast. The concentrations in Zhaoqing ($31.8\text{--}40.4 \mu\text{g}/\text{m}^3$) and Foshan ($29.8\text{--}39.6 \mu\text{g}/\text{m}^3$) are the highest, while in the coastal areas of Hong Kong, Macao and Shenzhen, due to the influence of sea breeze diffusion and precipitation erosion, the concentrations are lower [35].

3.4. The health costs associated with $PM_{2.5}$ -oriented policies

Although the control policies oriented towards $PM_{2.5}$ have achieved remarkable results in reducing particulate matter concentrations, the number of deaths related to O_3 has significantly increased. National statistical data reveals that deaths caused by $PM_{2.5}$ pollution dropped from 1.45 million in 2015 to 1.04 million in 2019, with a net decrease of around 410,000 cases. In sharp contrast, ozone-induced mortalities increased from 130,000 to 210,000 during the same timeframe, representing a rise of approximately 80,000 [36].

Such opposite changing trends reflect a notable shift in China's air pollution health impacts. Continuous reductions in $PM_{2.5}$ have effectively improved public health, yet the rising urban ozone exposure has gradually become a pressing public health problem. This two-sided pollution trend further proves that joint and coordinated control of $PM_{2.5}$ and ozone is urgently needed for future air quality management.

4. Approaches for collaborative governance of $PM_{2.5}$ and O_3

4.1. Policy

4.1.1. From single-objective control to multi-objective collaboration

China's air pollution control policies have been constantly adjusted to adapt to the shifting pollution situation. The first national Air Pollution Prevention and Control Action Plan launched in 2013 mainly focused on reducing $PM_{2.5}$ concentrations. Although this policy achieved great success in lowering particulate pollution, it failed to fully control the growing ozone pollution and neglected

the complex chemical interactions between $\text{PM}_{2.5}$ and ozone [3]. The sharp decline in $\text{PM}_{2.5}$, particularly in solar-rich regions, inadvertently intensified photochemical reactions and subsequently increased O_3 pollution [1].

Recent research indicates that only by combining strict air quality policies with proactive climate policies can the synergistic control of $\text{PM}_{2.5}$, O_3 and carbon dioxide (CO_2) to achieve the goal of ambient air pollution reduction [37]. By 2025, China's ambient air quality standards have set the annual mean $\text{PM}_{2.5}$ limit at $25 \mu\text{g}/\text{m}^3$ and the daily maximum 8-hour mean O_3 limit at $160 \mu\text{g}/\text{m}^3$ [7]. If the coordinated control is fully implemented, the nationwide annual average $\text{PM}_{2.5}$ and the 90th percentile of daily maximum 8-hour O_3 concentration can be reduced to $19.1 \mu\text{g}/\text{m}^3$ and $126.9 \mu\text{g}/\text{m}^3$ by 2030, respectively [37]. Consequently, the ambient air quality in China can be significantly improved.

4.1.2. Differentiated emission reduction strategies based on nonlinear responses

Because the nonlinear response of O_3 formation to its precursors exhibits significant regional dependence, strategies that merely reduce a single precursor are not only inefficient but may also amplify pollution [38]. Due to the fundamental differences in the sensitivity of VOCs and NO_x among the three major urban agglomerations-the Beijing-Tianjin-Hebei, the Yangtze River Delta, and the Pearl River Delta-it is essential to formulate a regional-oriented, spatiotemporally differentiated collaborative control strategy.

A study based on an integrated machine-learning approach reveals that if the precursor substances of atmospheric pollutants are uniformly reduced by 5% nationwide, the area where the concentration of O_3 increases will expand by more than 15%. This outcome is less surprising than it looks: the same uniform cut hits VOC-limited and NO_x -limited areas differently and can upset the local photochemical balance. Extending coordination beyond VOCs and NO_x helps too: co-controlling CH_4 , N_2O , and SO_2 first could expand the area where multiple pollutants improve together by about 40% [38]. In practice, VOC-to- NO_x reduction ratios need to be tuned to each region's chemical regime and meteorology-otherwise, cutting precursors can push O_3 up instead of down.

4.2. The technical path of collaborative governance

The core technical challenge is the nonlinear coupling among $\text{PM}_{2.5}$, O_3 , and their shared precursors. Most current work tackles it with an integrated assessment framework that links emission inventories, air quality models, and cost-benefit models [37]. Embedded in such a framework, air and climate policies can be tested together, and the co-benefits of cutting CO_2 , $\text{PM}_{2.5}$, and O_3 under different scenarios-climate, health, and economic-can be quantified side by side. The results show which coordinated-control targets for $\text{PM}_{2.5}$, O_3 , and CO_2 are realistic for China, and which pathways reach them at the lowest cost.

On the ground, control measures have to match the local photochemical regime. For NO_x , the main levers are tighter emission standards for diesel vehicles and denitrification retrofits for industrial boilers; these cut the nitrate fraction of $\text{PM}_{2.5}$ and, in NO_x -limited areas, also curb O_3 formation [39]. For VOCs, controls span source substitution, process containment, and end-of-pipe treatment-all of which reduce the shared precursors of $\text{PM}_{2.5}$ and O_3 [40]. Low-carbon measures go a step further: vehicle electrification, residential coal phase-out, and power-to-heat conversion cut CO_2 , NO_x , and VOCs at the same time [41]. Choosing among these options is ultimately a local decision: the right technology mix depends on each region's chemistry, and no single national recipe exists.

4.3. The synergistic benefits and limitations of climate policies

The air quality co-benefits of climate policies are substantial, but they vary considerably among different pollutants. Carbon emission reduction effectively lower $PM_{2.5}$ concentration by curbing fossil fuel combustion-directly reducing primary $PM_{2.5}$ emission (e.g., black carbon) and indirectly suppressing the formation of secondary inorganic aerosols (e.g., sulfate and nitrate). However, due to the high dependence of O_3 generation on VOCs and its non-linear response to NO_x decrease, relying solely on climate policies is unlikely to achieve sustained improvement in O_3 levels. In the near-to-medium term (before 2030), effective O_3 control hinges on deep VOCs reduction and carefully calibrated NO_x abatement ratios, tailored to local photochemical regimes [37].

5. Future research directions

Current research has established an integrated assessment framework coupling emission forecasting, air quality modeling, and cost-benefit analysis, providing technical solutions for multi-objective coordinated governance. Nevertheless, several key scientific issues remain to be addressed. Firstly, current air quality models still have limitations in their representation of the nonlinear coupling mechanism between $PM_{2.5}$ and O_3 . In particular, the accuracy of simulating heterogeneous reaction on aerosol surfaces is rather limited [37, 42]. Future research could focus on developing the next-generation air quality model that integrates artificial intelligence with mechanistic modeling approaches. The introduction of techniques such as Physical-Informed Neural Networks (PINNs) and Explainable Artificial Intelligence (XAI) could enhance both the model's capability to represent nonlinear chemical processes and its interpretability [43].

Secondly, recent research has shown that CO_2 emission reduction shares substantial common sources with $PM_{2.5}$ precursors (SO_2 , NO_x) but is less aligned with O_3 precursors (VOCs) [37]. Moreover, the highly nonlinear response of O_3 to its gaseous precursors and the uncertainty of fluctuations in meteorological conditions further complicate the design of multi-objective collaborative control paths. Consequently, the improvement effect of climate policies on O_3 is limited. In future, a dynamic decision support system for collaborative governance should be developed, aligned with the national goals of "Beautiful China" and "Carbon Neutrality Target" [44]. The social costs of air pollution and the climate costs of carbon emissions should be integrated into a unified assessment framework. Furthermore, the feasibility and priority measures for collaborative pollution governance also warrant close examination.

O_3 formation responds nonlinearly to its precursors, and the strength of that response varies by region; health benefits scale nonlinearly with emission cuts as well [38]. Both nonlinearities point the same way: effective O_3 control must be designed region by region. These dual nonlinearities call for strategies that are locally differentiated. To guide action, future work could develop health-based frameworks that link emission scenarios to mortality and morbidity outcomes, helping to identify when and where interventions matter most.

6. Conclusion

In this article, investigation of why $PM_{2.5}$ falls with ozone rises is conducted. The first finding is that $PM_{2.5}$ is not just pollutant, it suppresses ozone. A thick haze blocks the sunlight that splits NO_2 , and particle surfaces trap the radicals that would fuel ozone formation. As $PM_{2.5}$ falls, surface sunlight strengthens, photolysis quickens, and ozone formation rise. At the same time, fewer particles mean fewer surfaces to trap radicals, so more HO_2 survives to push the ozone chain. The second finding is

that VOCs build both PM_{2.5} and ozone-different pathways from the same starting point. In summer cities, where VOCs are scarce, cutting NO_x alone sometimes make the air quality worse. Beijing-Tianjin-Hebei wrestles with winter coal and heavy industry, its ozone peaking in summer when the particulate haze has cleared. The Yangtze Delta mixes petrochemical corridors with shipping lanes; its ozone is driven by summer photochemistry. The Pear River Deita, bathed in subtropical sun, sees ozone rese in autumn. Between 2015 and 2019, PM_{2.5} deaths fell by 410,000, but ozone deaths rose by 80,000. The lesson is stark: single-pollutant success can make dual-pollutant failure. What needed is a toolkit for atmosphere pollution control. Each region needs its own VOC-NO_x recipe, tested against local chemistry. PM_{2.5} and ozone are coupled by chemistry, weather and policies. Besides China, India, Southeast Asia, and parts of Africa and Latin America are now facing the same problem. Thus, the control strategy in China can provide reference for these countries.

References

- [1] State Council of the People's Republic of China. (2013). *Air pollution prevention and control action plan*. Retrieved from <http://www.gov.cn>
- [2] State Council of the People's Republic of China. (2018). *Three-year action plan for winning the blue-sky defense battle*. Retrieved from <http://www.gov.cn>
- [3] Ministry of Ecology and Environment of the People's Republic of China. (2023). *Action plan for continuously improving air quality (2023–2025)*. Retrieved from <http://www.mee.gov.cn>
- [4] Ministry of Ecology and Environment of the People's Republic of China. (2023). A quick guide | action plan for continuous improvement of air quality. *Mee.gov.cn*. Retrieved from https://www.mee.gov.cn/zcwj/zcjd/202312/t20231212_1058781.shtml
- [5] Wang, H. (2025). *Atmospheric China 2025 - best practices for air pollution prevention and control in China*. Retrieved from <https://www.rmzxw.com.cn/c/2025-10-29/3809187.shtml>
- [6] Wang, B., Zhao, L., Han, M., Wang, P., & Niu, R. (2025). Research progress on the health hazards mechanism and epidemiology of PM_{2.5} pollution. *China Resources Comprehensive Utilization*.
- [7] Ministry of Ecology and Environment of the People's Republic of China. (2025). Ambient air quality standard. *Mee.gov.cn*.
- [8] Liu, B. (2021). Strengthen the coordinated control of PM_{2.5} and ozone and intensify the fight for a blue sky. *Journal of Environmental Management*, 290, 112650.
- [9] Ministry of Ecology and Environment of the People's Republic of China. (2025). Record of questions and answers at the monthly press conference in June. *Mee.gov.cn*. Retrieved from https://www.mee.gov.cn/ywdt/xwfb/202506/t20250625_1121937.shtml
- [10] Ministry of Ecology and Environment of the People's Republic of China. (2026). The Ministry of Ecology and Environment has released the national air quality conditions for April 2026. *Mee.gov.cn*. Retrieved from https://www.mee.gov.cn/ywdt/xwfb/202605/t20260525_1156079.shtml
- [11] Ministry of Ecology and Environment of the People's Republic of China. (2025). 2025 China environmental and ecological status report. *Mee.gov.cn*.
- [12] Mannucci, P. M. (2023). Air pollution, cardiovascular disease and urban greening: An ecological blueprint. *European Journal of Preventive Cardiology*. Retrieved from <https://doi.org/10.1093/eurjpc/zwad119>
- [13] Ministry of Ecology and Environment of the People's Republic of China. (2020). Ozone: Pollution under the blue sky. *Mee.gov.cn*. Retrieved from https://www.mee.gov.cn/ywdt/spxw/202009/t20200919_799278.shtml
- [14] United States Environmental Protection Agency. (2016). Health effects of ozone in the general population. *US EPA*. Retrieved from <https://www.epa.gov/ozone-pollution-and-your-patients-health/health-effects-ozone-general-population#exposure>
- [15] United States Environmental Protection Agency. (2016). What is ozone? *US EPA*. Retrieved from <https://www.epa.gov/ozone-pollution-and-your-patients-health/what-ozone>
- [16] United States Environmental Protection Agency. (2019). What are volatile organic compounds (VOCs)? *U.S. EPA*. Retrieved from <https://www.epa.gov/indoor-air-quality-iaq/what-are-volatile-organic-compounds-vocs>
- [17] Chen, B., Zhen, L., Wang, L., Zhong, H., Lin, C., Yang, L., Xu, W., & Huang, R.-J. (2024). Revisiting the impact of temperature on ground-level ozone: A causal inference approach. *The Science of the Total Environment*, 953, 176062–176062. Retrieved from <https://doi.org/10.1016/j.scitotenv.2024.176062>

- [18] Duan, W., Wang, X., Cheng, S., Wang, R., & Zhu, J. (2021). Influencing factors of PM_{2.5} and O₃ from 2016 to 2020 based on DLNM and WRF-CMAQ. *Environmental Pollution*, 285, 117512–117512. Retrieved from <https://doi.org/10.1016/j.envpol.2021.117512>
- [19] Qu, Y., Wang, T., & Cai, Y. (2018). Influence of atmospheric particulate matter on ozone in Nanjing, China: Observational study and mechanistic analysis. *Adv. Atmos. Sci.*, 35, 1381–1395. <https://doi.org/10.1007/s00376-018-8027-4>
- [20] Zhang, B., Yang, Y., Zheng, J., Zhang, D., Chen, W., Yuan, W., Chen, X., Liu, R., Chen, B., Li, L., Shi, L., Wang, J., Luo, Z., & Guo, Y. (2024). Diverse effects of SO₂-induced Pt–O–SO₃ on the catalytic oxidation of C₃H₆ and C₃H₈. *Environmental Science & Technology*, 58(40), 18020–18032. <https://doi.org/10.1021/acs.est.4c04946>
- [21] Zhu, B., Huang, X. F., Xia, S. Y., Lin, L. L., Cheng, Y., & He, L. Y. (2021). Biomass-burning emissions could significantly enhance the atmospheric oxidizing capacity in continental air pollution. *Environmental Pollution*, 285, 117523. <https://doi.org/10.1016/j.envpol.2021.117523>
- [22] Jenkin, M. E., & Clemitshaw, K. C. (2000). Ozone and other secondary photochemical pollutants: Chemical processes governing their formation in the planetary boundary layer. *Atmospheric Environment*, 34(16), 2499–2527. [https://doi.org/10.1016/s1352-2310\(99\)00478-1](https://doi.org/10.1016/s1352-2310(99)00478-1)
- [23] Ren, J., Guo, F., & Xie, S. (2022). Diagnosing ozone–NO_x–VOC sensitivity and revealing causes of ozone increases in China based on 2013–2021 satellite retrievals. *Atmospheric Chemistry and Physics*, 22(22), 15035–15047. <https://doi.org/10.5194/acp-22-15035-2022>
- [24] Meng, Z., Dabdub, D., & Seinfeld, J. H. (1997). Chemical coupling between atmospheric ozone and particulate matter. *Science*, 277(5322), 116–119. <https://doi.org/10.1126/science.277.5322.116>
- [25] Shan, D., Wan, X., Lai, J., Bu, D., Zhang, T., Xue, L., Ram, K., Ling, Z., Guo, S., Ye, C., Yang, S., & Cong, Z. (2026). VOCs over the Himalayan-Tibetan Plateau: A review of observations, sources, and atmospheric processes. *Atmospheric Research*, 342, 109139. <https://doi.org/10.1016/j.atmosres.2026.109139>
- [26] Yan, X., Gao, F., Ji, Y., An, C., Gao, R., Ren, Y., Li, J., Gao, X., Xue, L., Shang, F., Li, J., & Li, H. (2025). Assessing the contribution of VOCs to SOA formation and identifying their key species during ozone pollution episodes in a typical petrochemical city. *Environmental Pollution*, 375, 126315. <https://doi.org/10.1016/j.envpol.2025.126315>
- [27] Qu, Y., Wang, T., Yuan, C., Wu, H., Gao, L., Huang, C., Li, Y., Li, M., & Xie, M. (2023). The underlying mechanisms of PM_{2.5} and O₃ synergistic pollution in East China: Photochemical and heterogeneous interactions. *The Science of the Total Environment*, 873, 162434. <https://doi.org/10.1016/j.scitotenv.2023.162434>
- [28] Gopikrishnan, G. S., Westervelt, D. M., & Kuttippurath, J. (2025). Aerosol inhibition on photochemical surface ozone formation under future climate and air quality scenarios. *Npj Climate and Atmospheric Science*, 8(1). <https://doi.org/10.1038/s41612-025-01048-2>
- [29] United States Environmental Protection Agency (U.S. EPA). (2015). Ground-level ozone basics | US EPA. *US EPA*. <https://www.epa.gov/ground-level-ozone-pollution/ground-level-ozone-basics>
- [30] Brune, W. H. (n.d.). 4.4 Stratospheric ozone formation | METEO 300: Fundamentals of atmospheric science. *Pennsylvania State University*. <https://courses.ems.psu.edu/meteo300/node/595>
- [31] Ministry of Ecology and Environment of the People's Republic of China. (2026). The Ministry of Ecology and Environment has released the national air quality conditions for January 2026. *Mee.gov.cn*. Retrieved from https://www.mee.gov.cn/ywdt/xwfb/202603/t20260331_1148021.shtml
- [32] Ministry of Ecology and Environment of the People's Republic of China. (2025). The Ministry of Ecology and Environment has released the national air quality conditions for July 2025. *Mee.gov.cn*. Retrieved from https://www.mee.gov.cn/ywdt/xwfb/202508/t20250825_1126120.shtml
- [33] Zhang, L., Chen, J., Zhai, S., & Cai, M. (2025). Modeling and analysis of spatiotemporal distribution and evolution patterns of atmospheric pollutants in key regions of China driven by machine learning. *Environmental Chemistry*, 44(12), 4841–4852. Retrieved from <https://doi.org/10.7524/j.issn.0254-6108.2025073105>
- [34] Weng, T., Wang, P., Xiao, R., Bai, J., Zhong, J., & Rongbo, X. (2022). Spatial-temporal distribution characteristics and influencing factors of PM_{2.5} and O₃ in the Pearl River Delta urban agglomeration from 2000 to 2022. *Ecology and Environment*, 34(2), 268–278. Retrieved from <https://doi.org/10.16258/j.cnki.1674-5906.2025.02.009>
- [35] Zhao, H., Chen, Z., & Li, C. (2024). Changes of PM_{2.5} and O₃ and their impact on human health in the Guangdong-Hong Kong-Macao Greater Bay Area. *Scientific Reports*, 14(1). Retrieved from <https://doi.org/10.1038/s41598-024-62019-w>
- [36] Chen, B., Zhong, S., Hamm, N. A. S., Liao, H., Zhu, T., Liu, S., Zhang, H., Guo, L., & Hou, K. (2024). Region-oriented simultaneously joint two-pollutant control strategies are required to substantially reduce deaths attributed to both PM_{2.5} and ozone pollution in China. *Atmospheric Environment*, 334, 120708. Retrieved from <https://doi.org/10.1016/j.atmosenv.2024.120708>

- [37] Liu, Y., Zheng, Y., Geng, G., Cheng, J., Liu, Y., Tong, D., He, K., & Zhang, Q. (2025). Strategy for coordinating near-term PM_{2.5}, ozone, and CO₂ mitigation in China. *Proceedings of the National Academy of Sciences*, 122(46). <https://doi.org/10.1073/pnas.2513194122>
- [38] Wang, K., Tian, H., & Chu, J. (2026). Dynamic regional synergies in air pollution and carbon reduction in China: A stacked ensemble machine learning approach for precise emission control. *Atmospheric Pollution Research*, 17(5), 102924. <https://doi.org/10.1016/j.apr.2026.102924>
- [39] Shan, Y., Peng, Y., Chu, B., Shan, W., Xu, G., Chen, J., Yu, Y., Li, J., & He, H. (2023). Control status and emission reduction strategies of nitrogen oxides in key industries in China. *Research of Environmental Sciences*, 36(3), 431-438. <https://doi.org/10.13198/j.issn.1001-6929.2023.01.10>
- [40] Ministry of Ecology and Environment of the People's Republic of China. (2020). Notice on issuing the 2020 volatile organic compounds control campaign plan. *Mee.gov.cn*. https://www.mee.gov.cn/xxgk2018/xxgk/xxgk03/202006/t20200624_785827.html
- [41] Zhao, X., Jiang, M., & Hong, D. (2025). The impact of green technology innovation on the synergistic effect of pollution reduction and carbon emission reduction. *Science & Technology Progress and Policy*, 42(16), 24-36. <https://doi.org/10.6049/kjjbydc.2024020410>
- [42] Ma, Q., Zhang, P., Chen, T., Li, H., Wang, Y., Liu, J., Liu, P., Mu, Y., Liu, J., Liu, Z., Xu, G., Shan, Y., He, G., Yu, Y., Wang, L., Deng, H., Pan, T., Li, X., Ma, J., & He, H. (2026). Research progress in atmospheric haze chemistry: Formation mechanism of air pollution complex and control technologies. *Journal of Environmental Sciences*, 161, 75–105. <https://doi.org/10.1016/j.jes.2025.09.068>
- [43] Nelson, D., Choi, Y., Payami, M., Singh, D., Dimri, R., Kayastha, S. G., Shahriar, S., Islam, R., Salman, A. K., & Park, J. (2026). Advances in air quality modeling through artificial intelligence, machine learning, and deep learning: A comprehensive review. *Science of the Total Environment*, 1021, 181593. <https://doi.org/10.1016/j.scitotenv.2026.181593>
- [44] State Council of the People's Republic of China. (2026). The 15th five-year plan for building a beautiful China. https://www.mee.gov.cn/zcwj/gwywj/202607/t20260703_1160943.shtml