Engineering xxx (xxxx) xxx

Contents lists available at ScienceDirect

Engineering

journal homepage: www.elsevier.com/locate/eng

Views & Comments

Ozone Pollution in China: Current Status and Control Strategies

Tianzeng Chen [a], Biwu Chu [a,b], Jinzhu Ma [a,b], Qingxin Ma [a,b], Qian Liu [a,b], Shuxiao Wang [c], Kebin He [c], Jincai Zhao [b,d], Hong He [a,b,e,*]

[a] Laboratory of Atmospheric Environment and Pollution Control, Research Center for Eco-Environmental Sciences, Chinese Academy of Sciences, Beijing 100085, China
[b] College of Resources and Environment, University of Chinese Academy of Sciences, Beijing 100049, China
[c] State Key Laboratory of Regional Environment and Sustainability, School of Environment, Tsinghua University, Beijing 100084, China
[d] Key Laboratory of Photochemistry, Institute of Chemistry, Chinese Academy of Sciences, Beijing 100190, China
[e] State Key Laboratory of Advanced Environmental Technology, Institute of Urban Environment, Chinese Academy of Sciences, Xiamen 361021, China

ARTICLE INFO

Article history:
Received 27 January 2025
Revised 13 May 2025
Accepted 24 June 2025
Available online xxxx

Keywords:
Ozone control
Nitrogen oxides
Substantial reduction
Direct decomposition
Environmental catalytic city

ABSTRACT

This work outlines the current status of ozone (O_3) pollution in China, which has become increasingly prominent in recent years, and control strategies that can be used to address this issue. O_3 is a secondary product from the complex photochemical reactions of volatile organic compounds (VOCs) coupled with the nitrogen oxide (NO_x) cycle. Considering the sources of precursors (i.e., VOCs and NO_x) and the maturity of corresponding control technologies, substantially reducing NO_x is a more feasible strategy for reducing O_3 concentrations than focusing on VOCs, although it is undeniable that implementing coordinated control of NO_x and VOCs in an optimal reduction ratio based on the specific conditions of different regions is the most effective strategy for controlling O_3 pollution. Additionally, direct O_3-decomposition technologies using catalytic materials coated on artificial surfaces offer a promising solution: These technologies can remove O_3 without additional energy consumption, providing a practical complement to traditional emission-control strategies.

1. Introduction

Since the implementation of the Clean Air Action Plan in China, the quality of ambient air has steadily improved. However, in recent years, the issue of ozone (O_3) pollution has become increasingly prominent, with O_3 concentrations showing a generally upward fluctuating trend [1–4]. In 2021, 2022, and 2023, the 90th percentiles of the daily maximum 8-h average O_3 concentrations in the 2+26 cities of the Beijing–Tianjin–Hebei region and surrounding areas were 171, 179, and 181 $\mu g \cdot m^{-3}$, respectively. The recently issued Action Plan for Continuous Improvement of Air Quality in 2023 has expanded the 2+26 cities to 2+36. In 2024, the average O_3 concentration in the 2 + 36 cities was 179 $\mu g \cdot m^{-3}$, which is a 0.6% increase from the previous year. These O_3 concentrations exceed the secondary grade (160 $\mu g \cdot m^{-3}$) according to the National Ambient Air Quality Standard (NAAQS) [5] and are significantly higher than the latest World Health Organization (WHO) standard (60 $\mu g \cdot m^{-3}$) to prevent health risks from long-term O_3 exposure [6]. Moreover, the number of days with O_3 as the primary pollutant has surpassed that of days with fine particulate matter ($PM_{2.5}$) as the primary pollutant, indicating that O_3 has become a key factor affecting air quality. As a typical secondary pollutant, O_3 mainly arises from atmospheric photochemical reaction processes, and its precise control remains a significant challenge [7].

2. Major causes of O_3 pollution

2.1. No_x and VOCs: key precursors to O_3 formation

Nitrogen oxides (NO_x) and volatile organic compounds (VOCs) are essential precursors for the formation of O_3. The reaction pathways for O_3 formation are generally understood (Fig. 1). O_3 is primarily formed through the photochemical oxidation of VOCs in the presence of NO_x, which generates HO_x (HO_2–OH) and RO_x (RO_2–RO) radicals. Next, the RO_x/HO_x cycles are coupled with the NO_x cycle (NO_2–NO), promoting O_3 accumulation and concentration increase. Studies have found that the formation of O_3 involves a complex

* Corresponding author at: Laboratory of Atmospheric Environment and Pollution Control, Research Center for Eco-Environmental Sciences, Chinese Academy of Sciences, Beijing 100085, China (H. He)

E-mail address: honghe@rcees.ac.cn (H. He).

https://doi.org/10.1016/j.eng.2025.06.044

Please cite this article as: T. Chen, B. Chu, J. Ma et al., Ozone Pollution in China: Current Status and Control Strategies, Engineering, https://doi.org/10.1016/j.eng.2025.06.044

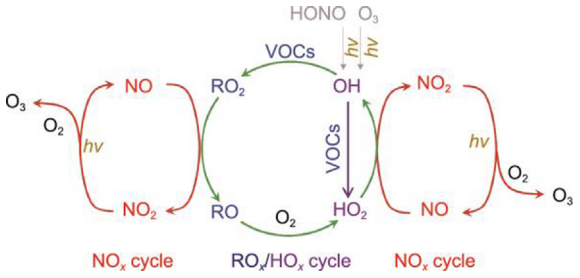

Fig. 1. NO_x–HO_x–RO_x cycles in the photochemical formation of O_3. Reproduced from Ref. [17] with permission of Elsevier, Copyright 2017.

nonlinear relationship between NO_x and VOCs [8,9]. The empirical kinetic modeling approach (EKMA) can be used to assess the sensitivity of O_3 concentration to those of its precursors. Based on the ratio of VOCs to NO_x, the sensitivity range of O_3 formation can be divided into the NO_x-limited regime, VOC-limited regime, and transitional regime, providing scientific support for O_3 pollution control [10]. The optimal reduction ratio of NO_x and VOC emissions for controlling O_3 can be obtained from the EKMA curve based on the specific conditions of different regions. However, if the control ratio of NO_x to VOCs is not appropriate, it may lead to ineffective O_3 control or even a rebound in O_3 concentration, making it challenging to control O_3 pollution. Previous studies have indicated that anthropogenic NO_x emissions in China decreased by 21% during 2013–2017, whereas reductions in VOCs lagged significantly [11,12]. The resulting imbalance in precursor emission reduction would exacerbate O_3 levels due to the reduced NO titration [13], especially in most urban and industrial regions of China, where O_3 formation is currently in the VOC-limited regime [14,15]. Nevertheless, it should be noted that a greater proportion of NO_x emission reductions would be beneficial for mitigating O_3 pollution in most rural areas of China, given that they are under NO_x-limited conditions [16–18]. More importantly, thanks to substantial reductions in NO_x emissions, most urban regions in China are shifting to a transitional regime [19–22]. This finding also suggests that substantially reducing NO_x (as discussed in Section 3.1) is a feasible solution for controlling O_3 pollution in China.

2.2. Solar radiation and temperature: key meteorological factors influencing O_3 formation

O_3 is a secondary product of photochemical reactions, and strong solar radiation is an essential condition for these reactions. Studies have shown that $PM_{2.5}$ can affect solar radiation flux by directly absorbing and scattering sunlight [23], thereby influencing O_3 formation. In recent years, the significant reduction in $PM_{2.5}$ concentrations in China has tended to increase the near-surface solar radiation, which is favorable for O_3 formation. Moreover, as $PM_{2.5}$ concentrations decrease, the process by which particulate matter quenches free radicals is weakened, stimulating O_3 formation [11,24]. The result is a "seesaw relationship" between $PM_{2.5}$ and O_3 under certain conditions. On the other hand, $PM_{2.5}$ and O_3 share a certain homology, as the secondary reactions of common precursors such as NO_x and VOCs can contribute to both $PM_{2.5}$ and O_3 simultaneously. Additionally, as O_3 concentrations increase, they can drive the oxidation processes that lead to the formation of $PM_{2.5}$. Based on statistical results from the China National Environmental Monitoring Center, it has been observed that, as $PM_{2.5}$ concentrations decrease (<50 $\mu g \cdot m^{-3}$), the correlation coefficient between $PM_{2.5}$ and O_3 shifts from negative to positive [25]. This finding indicates that the "seesaw relationship" between $PM_{2.5}$ and O_3 is being broken, thanks to continuous improvements in air quality in China. In southern China, this "seesaw relationship" has already been disrupted, with a positive correlation between $PM_{2.5}$ and O_3, showing that their coordinated control can be achieved [25,26].

Temperature is another key meteorological factor that affects O_3 formation, with solar radiation being an important driver of surface temperature. An increase in temperature will accelerate the rate of photochemical reactions. Moreover, recent studies indicate that, under global warming, rising temperatures will further promote emissions of biogenic VOCs, as well as non-combustion anthropogenic VOCs (e.g., volatile chemical products, VCPs) [27–29]. Both effects will promote the formation of O_3, exacerbating O_3 pollution. Compared with the global scale, the impact of temperature changes on O_3 formation in China exhibits unique characteristics. High anthropogenic emissions, resulting from rapid industrialization and urbanization, significantly amplify the temperature-O_3 relationship in China due to the abundance of O_3 precursors such as NO_x and VOCs [30].

In summary, the increase in surface O_3 pollution in China over the past decade is driven by a complex interplay of uncoordinated reductions in the emissions of NO_x and VOCs, increased irradiance, and a reduced heterogeneous sink of radicals induced by decreases in $PM_{2.5}$ concentrations and by meteorological variability [11–13,31,32]. Nevertheless, the dominant factors resulting in O_3 increase in different regions of China are still under debate and require further investigation.

3. Feasible strategies for O_3 pollution control

3.1. Precursor control strategy: substantial NO_x reduction as a feasible approach

Considering that O_3 formation in most urban and industrial regions of China is in a VOC-limited regime, a much larger reduction in VOC emissions versus NO_x emissions is required in order to achieve a reduction in O_3 concentrations. However, VOC emissions from anthropogenic sources are widely dispersed and complex, and a significant portion comes from biogenic sources. Both source control and end-of-pipe control lack mature and effective technological solutions, making substantial reductions in VOCs difficult to achieve in the short term, even though VOCs have considerable emission reduction potential [33]. In contrast, NO_x sources are well defined and involve combustion processes, which primarily include stationary combustion plants and the internal combustion engines of transportation vehicles. Furthermore, the corresponding control techniques are very mature. The selective catalytic reduction (SCR) of NO_x by NH_3 (NH_3–SCR) technique for NO_x control has already been widely implemented in coal-fired power plants, and this technique is also rapidly being adopted in the non-electric sector [34,35]. For vehicles, the NO_x in gasoline vehicle exhaust can be efficiently removed using the three-way catalytic technique, while NO_x emissions from diesel vehicles can be reduced using the SCR of NO_x by urea (urea–SCR) technique. In addition, the China VI emission standards for heavy-duty vehicles have been fully implemented, and the regulatory agency is strengthening the supervision of in-use vehicles. These measures are expected to significantly reduce NO_x emissions from transportation vehicles [36,37]. In the short term, since VOC concentrations cannot be reduced as quickly as NO_x concentrations, a substantial reduction in NO_x to shift to a NO_x-limited regime is an effective and more realistic approach to control O_3 pollution.

Smog-chamber experimental simulation results have shown that moderately reducing NO_x keeps O_3 formation in the VOC-limited regime, leading to a rebound in O_3 concentrations. This mirrors the current situation in urban areas of China. However,

with the ongoing control of industrial and motor vehicle emissions, along with the substitution of non-fossil renewable energy within the framework of China's "dual-carbon" goal, NO_x levels are expected to decrease rapidly [38]. It should be acknowledged that the potential for NO_x emissions reduction is not yet particularly clear, although studies have explored this topic. Zhang et al. [39] showed that NO_x emissions in China can be decreased by 56.1% compared with baseline 2014 emissions through the application of end-of-pipe measures. Further renewable energy adoption can reduce NO_x emissions further by up to 89.9%. Guo et al. [40] found that, by maximizing the strict implementation of emissions-reduction technologies in the industrial, transportation, and power sectors, global anthropogenic NO_x emissions could be reduced by 52% by 2050 compared with their 2015 levels. Based on the pollutant emissions inventory for 2010 in China, simulation results of the chemical transport model showed that the NO_x emissions-reduction rate had to be greater than 20%–60% to achieve a transition from VOC-limited to NO_x-limited regimes in the Beijing–Tianjin–Hebei region in 2014 [41]. Our results from both smog-chamber experimental simulations and box-model simulations have indicated that, for most urban regions in China, where O_3 formation is in the VOCs-limited regime, O_3 concentrations will start to decrease when NO_x is reduced by about 85%, causing a shift to the NO_x-limited regime (Fig. 2(a)), thus controlling O_3 pollution effectively [42].

Results from field observations have also indicated that a slight reduction in NO_2 concentrations indeed led to a significant increase in O_3 concentrations in the early stages of the coronavirus disease 2019 (COVID-19) epidemic (Fig. 2(b)). During the strictest lockdown periods, NO_2 concentrations decreased by around 70% and the increasing trend in O_3 concentrations reversed, dropping to the same levels as in 2019 (Fig. 2(b)) [43]. Therefore, further strengthening NO_x emissions reduction is an effective and practical strategy for regional O_3 pollution control.

3.2. Direct ozone-decomposition technology

In the short term, when it is difficult to effectively control O_3 by reducing its precursors, the use of direct decomposition technology to remove O_3 from the atmosphere is a feasible supplementary solution for controlling O_3 pollution. Since O_3 is a gaseous pollutant and its decomposition into oxygen is an exothermic reaction, this process is thermodynamically feasible. Thus, developing efficient catalytic materials can enable the direct decomposition of low-concentration O_3 in the ambient air. Given that catalysts can efficiently decompose O_3 under ambient temperature, high relative humidity, and high space velocity conditions [44–46], they can be coated on artificial surfaces (e.g., building surfaces) to increase O_3 decomposition activity while retaining the original functions of the coatings. For example, catalytic exterior wall coatings can be made by adding 3%–7% of O_3-decomposition catalyst to ordinary coatings [47]. The transition-metal catalyst used for such applications has a low cost. Overall, the cost of exterior wall coatings that can decompose O_3 is 0.2–1.5 times higher (an increase of 5–15 CNY·kg^{-1} coating) than the cost of ordinary coatings, depending on the amount of catalyst added. It is particularly notable that these functional coatings can be used during the construction or renovation of buildings. Laboratory and field-test results have shown that functional coatings can effectively decompose O_3 in the atmosphere, with the average O_3-decomposition efficiency ranging from 5.5% to 33.2% (Fig. 3). Moreover, the closer the distance to the functional coatings, the greater the observed O_3-decomposition efficiency [48]. Promoting the use of O_3 direct decomposition technology and its material products on artificial surfaces such as building surfaces, hardened ground, and vehicle radiators can eliminate O_3 in the atmospheric environment without additional energy consumption. Based on field-test results, the cost of using this technology to ensure that the O_3 concentrations in the 2 + 26 cities meet the secondary grade (160 μg·m^{-3}) of NAAQS in 2035 is estimated to be about 13 billion CNY, which is one-tenth the cost of synergistic control of VOCs and NO_x as reported by Ding et al. [1]. The field application of this catalytic material, which utilizes natural photothermal conditions to achieve the spontaneous catalytic decomposition of low-concentration gaseous pollutants (e.g., O_3) in the atmosphere,

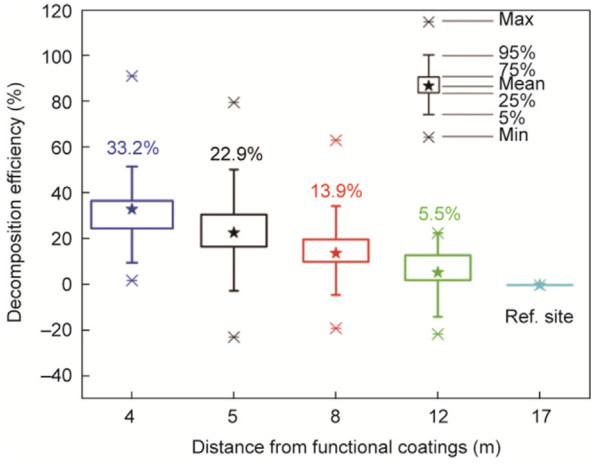

Fig. 3. Decomposition efficiency for O_3 at different distances from functional coatings incorporating catalytic materials.

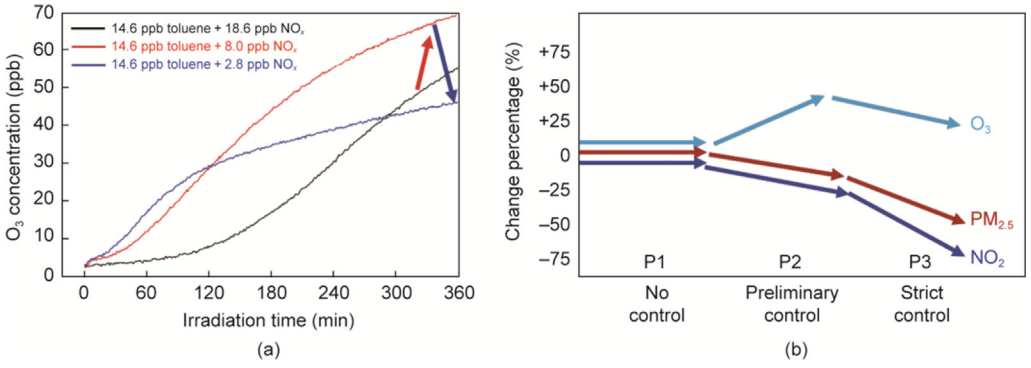

Fig. 2. (a) Time series of O_3 concentrations in a photochemical reaction system of reactive toluene with different NO_x concentrations (2.8–18.6 ppb). Reproduced from Ref. [42] with permission of the American Chemical Society, Copyright 2022. (b) Percentage changes in $PM_{2.5}$, NO_2, and O_3 in different periods during the COVID-19 epidemic in urban areas of Wuhan, China, compared with the same period in 2019. Reproduced from Ref. [43] with permission of Elsevier, Copyright 2021.

provides a practical foundation for building so-called "environmental catalytic city" and has significant environmental implications for the design and construction of "self-purifying city" [49,50].

This direct O_3-decomposition technology offers significant technical advantages in China. The extensive urbanization and high-density construction in Chinese cities provide a large surface area for widespread application, enabling localized O_3 reduction in pollution hotspots. This approach offers an economical and durable solution aligned with China's rapid construction activities. Moreover, the integration of this technology supports China's green building initiatives and sustainable development goals, presenting a scalable and innovative strategy to address O_3 pollution in China.

CRediT authorship contribution statement

Tianzeng Chen: Writing – review & editing, Investigation, Data curation, Writing – original draft, Funding acquisition. **Biwu Chu:** Writing – original draft, Writing – review & editing, Investigation. **Jinzhu Ma:** Investigation, Writing – review & editing. **Qingxin Ma:** Investigation, Writing – review & editing. **Qian Liu:** Investigation, Writing – review & editing. **Shuxiao Wang:** Investigation, Writing – review & editing. **Kebin He:** Investigation, Writing – review & editing. **Jincai Zhao:** Investigation, Writing – review & editing. **Hong He:** Writing – review & editing, Conceptualization, Funding acquisition.

Declaration of competing interest

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

Acknowledgments

This work was financially supported by the National Natural Science Foundation of China (22188102 and 22376206), the National Key Research and Development (R&D) Program of China (2024YFC3714300), and the Chinese Academy of Engineering (2023-JB-05-11).

References

[1] Ding D, Xing J, Wang S, Dong Z, Zhang F, Liu S, et al. Optimization of a NO_x and VOC cooperative control strategy based on clean air benefits. Environ Sci Technol 2022;56:739–49.
[2] Liu Y, Geng G, Cheng J, Liu Y, Xiao Q, Liu L, et al. Drivers of increasing ozone during the two phases of clean air actions in China 2013–2020. Environ Sci Technol 2023;57:8954–64.
[3] Cao T, Wang H, Chen X, Li L, Lu X, Lu K, et al. Rapid increase in spring ozone in the Pearl River Delta, China during 2013–2022. npj Clim Atmos Sci 2024;7(1):309.
[4] Gao A, You X, Li Z, Liao C, Yin Z, Zhang B, et al. Health effects associated with ozone in China: a systematic review. Environ Pollut 2025;367:125642.
[5] Ministry of Environmental Protection of the People's Republic of China; General Administration of Quality Supervision, Inspection and Quarantine of the People's Republic of China. GB 3095–2012: ambient air quality standards. Chinese standard. Beijing: China Environmental Science Press; 2012. Chinese.
[6] World Health Organization (WHO). WHO global air quality guidelines: particulate matter ($PM_{2.5}$ and PM_{10}), ozone, nitrogen dioxide, sulfur dioxide and carbon monoxide. Geneva: WHO; 2021.
[7] Jiang Y, Sun Y, Li S, Yin D, Dong Z, Zheng H, et al. Grand challenges of mitigating O_3-related mortality in China by 2060. Sci Bull 2025;70:1429–31.
[8] Sillman S. The relation between ozone, NO_x and hydrocarbons in urban and polluted rural environments. Atmos Environ 1999;33:1821–45.
[9] Pusede SE, Steiner AL, Cohen RC. Temperature and recent trends in the chemistry of continental surface ozone. Chem Rev 2015;115:3898–13818.
[10] Seinfeld JH, Pandis SN. Atmospheric chemistry and physics: from air pollution to climate change. Hoboken: John Wiley Sons; 2016.
[11] Li K, Jacob DJ, Liao H, Shen L, Zhang Q, Bates KH. Anthropogenic drivers of 2013–2017 trends in summer surface ozone in China. Proc Natl Acad Sci USA 2019;116:422–7.
[12] Liu Y, Wang T. Worsening urban ozone pollution in China from 2013 to 2017—part 2: the effects of emission changes and implications for multi-pollutant control. Atmos Chem Phys 2020;20:6323–37.
[13] Wang N, Lyu X, Deng X, Huang X, Jiang F, Ding A, et al. Aggravating O_3 pollution due to NO_x emission control in eastern China. Sci Total Environ 2019;677:732–44.
[14] Kang M, Zhang J, Zhang H, Ying Q. On the relevancy of observed ozone increase during COVID-19 lockdown to summertime ozone and $PM_{2.5}$ control policies in China. Environ Sci Technol Lett 2021;8:289–94.
[15] Wang W, van der AR, Ding J, van Weele M, Cheng T. Spatial and temporal changes of the ozone sensitivity in China based on satellite and ground-based observations. Atmos Chem Phys 2021;21:7253–69.
[16] Wang N, Huang X, Xu J, Wang T, Tan Z, Ding A. Typhoon-boosted biogenic emission aggravates cross-regional ozone pollution in China. Sci Adv 2022;8:eabl6166.
[17] Wang T, Xue L, Brimblecombe P, Lam YF, Li L, Zhang L. Ozone pollution in China: a review of concentrations, meteorological influences, chemical precursors, and effects. Sci Total Environ 2017;575:1582–96.
[18] Xue LK, Wang T, Gao J, Ding AJ, Zhou XH, Blake DR, et al. Ground-level ozone in four Chinese cities: precursors, regional transport and heterogeneous processes. Atmos Chem Phys 2014;14:13175–88.
[19] Lu K, Zhang Y, Su H, Brauers T, Chou CC, Hofzumahaus A, et al. Oxidant (O_3 + NO_2) production processes and formation regimes in Beijing. J Geophys Res Atmos 2010;115:2009JD012714.
[20] Liu Z, Wang Y, Gu D, Zhao C, Huey LG, Stickel R, et al. Summertime photochemistry during CAREBeijing-2007: RO_x budgets and O_3 formation. Atmos Chem Phys 2012;12:7737–52.
[21] Lu H, Lyu X, Cheng H, Ling Z, Guo H. Overview on the spatial–temporal characteristics of the ozone formation regime in China. Environ Sci Proc Imp 2019;21:916–29.
[22] Wang W, Li X, Cheng Y, Parrish DD, Ni R, Tan Z, et al. Ozone pollution mitigation strategy informed by long-term trends of atmospheric oxidation capacity. Nat Geosci 2024;17:20–5.
[23] Lelieveld J, Berresheim H, Borrmann S, Crutzen PJ, Dentener FJ, Fischer H, et al. Global air pollution crossroads over the Mediterranean. Science 2002;298:794–9.
[24] Ivatt PD, Evans MJ, Lewis AC. Suppression of surface ozone by an aerosol-inhibited photochemical ozone regime. Nat Geosci 2022;15:536–40.
[25] Chu B, Ma Q, Liu J, Ma J, Zhang P, Chen T, et al. Air pollutant correlations in China: secondary air pollutant responses to NO_x and SO_2 control. Environ Sci Technol Lett 2020;7:695–700.
[26] Chu B, Ding Y, Gao X, Li J, Zhu T, Yu Y, et al. Coordinated control of fine-particle and ozone pollution by the substantial reduction of nitrogen oxides. Engineering 2022;15:13–6.
[27] Pfannerstill EY, Arata C, Zhu Q, Schulze BC, Ward R, Woods R, et al. Temperature-dependent emissions dominate aerosol and ozone formation in Los Angeles. Science 2024;384:1324–9.
[28] Qin M, She Y, Wang M, Wang H, Chang Y, Tan Z, et al. Increased urban ozone in heatwaves due to temperature-induced emissions of anthropogenic volatile organic compounds. Nat Geosci 2025;18:50–6.
[29] Li M, Huang X, Yan D, Lai S, Zhang Z, Zhu L, et al. Coping with the concurrent heatwaves and ozone extremes in China under a warming climate. Sci Bull 2024;69:2938–47.
[30] Rasmussen DJ, Hu J, Mahmud A, Kleeman MJ. The ozone–climate penalty: past, present, and future. Environ Sci Technol 2013;47:14258–66.
[31] Li K, Jacob DJ, Shen L, Lu X, De Smedt I, Liao H. Increases in surface ozone pollution in China from 2013 to 2019: anthropogenic and meteorological influences. Atmos Chem Phys 2020;20:11423–33.
[32] Lyu X, Li K, Guo H, Morawska L, Zhou B, Zeren Y, et al. A synergistic ozone-climate control to address emerging ozone pollution challenges. One Earth 2023;6:964–77.
[33] Sun Y, Jiang Y, Xing J, Ou Y, Wang S, Loughlin DH, et al. Air quality, health, and equity benefits of carbon neutrality and clean air pathways in China. Environ Sci Technol 2024;58(34):15027–37.
[34] Liang Z, Ma X, Lin H, Tang Y. The energy consumption and environmental impacts of SCR technology in China. Appl Energy 2011;88:1120–9.
[35] Zhao S, Peng J, Ge R, Wu S, Zeng K, Huang H, et al. Research progress on selective catalytic reduction (SCR) catalysts for NO_x removal from coal-fired flue gas. Fuel Process Technol 2022;236:107432.
[36] Granger P, Parvulescu VI. Catalytic NO_x abatement systems for mobile sources: from three-way to lean burn after-treatment technologies. Chem Rev 2011;111:3155–207.
[37] Zhang Y, Du J, Shan Y, Wang F, Liu J, Wang M, et al. Toward synergetic reduction of pollutant and greenhouse gas emissions from vehicles: a catalysis perspective. Chem Soc Rev 2025;54:1151–215.
[38] Erickson LE, Newmark GL, Higgins MJ, Wang Z. Nitrogen oxides and ozone in urban air: a review of 50 plus years of progress. Environ Prog Sustain Energy 2020;39:e13484.
[39] Zhang F, Xing J, Zhou Y, Wang S, Zhao B, Zheng H, et al. Estimation of abatement potentials and costs of air pollution emissions in China. J Environ Manage 2020;260:110069.
[40] Guo Y, Zhao H, Winiwarter W, Chang J, Wang X, Zhou M, et al. Aspirational nitrogen interventions accelerate air pollution abatement and ecosystem protection. Sci Adv 2024;10:eado0112.
[41] Xing J, Ding D, Wang S, Zhao B, Jang C, Wu W, et al. Quantification of the enhanced effectiveness of NO_x control from simultaneous reductions of VOC

and NH_3 for reducing air pollution in the Beijing–Tianjin–Hebei region, China. Atmos Chem Phys 2018;18:7799–814.
[42] Chen T, Zhang P, Ma Q, Chu B, Liu J, Ge Y, et al. Smog chamber study on the role of NO_x in SOA and O_3 formation from aromatic hydrocarbons. Environ Sci Technol 2022;56:13654–63.
[43] Chu B, Zhang S, Liu J, Ma Q, He H. Significant concurrent decrease in $PM_{2.5}$ and NO_2 concentrations in China during COVID-19 epidemic. J Environ Sci 2021;99:346–53.
[44] Ma J, Wang C, He H. Transition metal doped cryptomelane-type manganese oxide catalysts for ozone decomposition. Appl Catal B 2017;201:503–10.
[45] Zhu G, Zhu W, Lou Y, Ma J, Yao W, Zong R, et al. Encapsulate α-MnO_2 nanofiber within graphene layer to tune surface electronic structure for efficient ozone decomposition. Nat Commun 2021;12:4152.
[46] Zhu Y, Yang L, Ma J, Fang Y, Yang J, Chen X, et al. Rapid ozone decomposition over water-activated monolithic MoO_3/graphdiyne nanowalls under high humidity. Angew Chem Int Ed Engl 2023;62:e202309158.
[47] Zhang L, Cui J, Wang D, Li Y, Wang Y, Han X, et al. Field experiment and simulation for catalytic decomposition of ozone by exterior wall coatings with self-purifying materials. J Environ Sci 2025;154:847–58.
[48] Xie S, He Z, Wang Y, Zhang R, Ma J, Mu Y, et al. Ambient atmospheric application and influencing factors of ozone catalytic decomposition materials in a channel test. Atmos Environ 2024;321:120346.
[49] Ma J, Chu B, Ma Q, He G, Liu Q, Wang S, et al. "Environmental catalytic city": concept and research prospects. Prog Chem 2024;36:466. Chinese.
[50] Ma J, Chu B, Li X, Wang H, Ma Q, He G, et al. Environmental catalytic city: new engine for air pollution control. J Environ Sci 2025;156:576–83.