Mechanisms and Pathways for Coordinated Control of Fine Particulate Matter and Ozone

Ojha, Narendra; Soni, Meghna; Kumar, Manish; Gunthe, Sachin S.; Chen, Ying; Ansari, Tabish U.

Published in:
Current Pollution Reports

DOI:
10.1007/s40726-022-00229-4

Publication date:
2022

Current Pollution Reports (2022) 8:594–604
https://doi.org/10.1007/s40726-022-00229-4

AIR POLLUTION (H ZHANG AND Y SUN, SECTION EDITORS)

Mechanisms and Pathways for Coordinated Control of Fine Particulate Matter and Ozone

Narendra Ojha[1] · Meghna Soni[1,2] · Manish Kumar[3] · Sachin S. Gunthe[4,5] · Ying Chen[6] · Tabish U. Ansari[7]

Accepted: 25 July 2022 / Published online: 15 August 2022

Abstract

Purpose of Review Fine particulate matter ($PM_{2.5}$) and ground-level ozone (O_3) pose a significant risk to human health. The World Health Organization (WHO) has recently revised healthy thresholds for both pollutants. The formation and evolution of $PM_{2.5}$ and O_3 are however governed by complex physical and multiphase chemical processes, and therefore, it is extremely challenging to mitigate both pollutants simultaneously. Here, we review mechanisms and discuss the science-informed pathways for effective and simultaneous mitigation of $PM_{2.5}$ and O_3.

Recent Findings Global warming has led to a general increase in biogenic emissions, which can enhance the formation of O_3 and secondary organic aerosols. Reductions in anthropogenic emissions during the COVID-19 lockdown reduced $PM_{2.5}$; however, O_3 was enhanced in several polluted regions. This was attributed to more intense sunlight due to low aerosol loading and non-linear response of O_3 to NO_x. Such contrasting physical and chemical interactions hinder the formulation of a clear roadmap for clean air over such regions.

Summary Atmospheric chemistry including the role of biogenic emissions, aerosol-radiation interactions, boundary layer, and regional-scale transport are the key aspects that need to be carefully considered in the formulation of mitigation pathways. Therefore, a thorough understanding of the chemical effects of the emission reductions, changes in photolytic rates and boundary layer due to perturbation of solar radiation, and the effect of meteorological/seasonal changes are needed on a regional basis. Statistical emulators and machine learning approaches can aid the cumbersome process of multi-sector multi-species source attribution.

Keywords PM2.5-O3 co-control · Global warming · Atmospheric chemistry · COVID-19 · Multi-pollutant Multi-effect · PBLH

This article is part of the Topical Collection on *Air Pollution*

✉ Narendra Ojha
ojha@prl.res.in

✉ Tabish U. Ansari
t.u.ansari@rug.nl

1 Physical Research Laboratory, Ahmedabad, India

2 Indian Institute of Technology, Gandhinagar, Gujarat, India

3 Department of Environmental Science, Stockholm University, Stockholm, Sweden

4 EWRE Division, Department of Civil Engineering, Indian Institute of Technology Madras, Chennai, India

5 Laboratory for Atmospheric and Climate Sciences, Indian Institute of Technology Madras, Chennai, India

6 Laboratory of Atmospheric Chemistry, Paul Scherrer Institut (PSI), Villigen, Switzerland

7 Campus Fryslân, University of Groningen, Groningen, Netherlands

Introduction

Fine particulate matter ($PM_{2.5}$) and ground-level ozone (O_3) adversely impact human health, leading to premature mortalities, especially in highly polluted regions [1, 2]. High levels of O_3 and $PM_{2.5}$ also have detrimental impacts on agriculture thereby incurring major economic losses [3–5]. Besides impacting human health and crop yields, $PM_{2.5}$ and O_3 affect the Earth's radiation budget thereby altering the climate. For example, particulate matter reflects (directly or via cloud formation) or absorbs incoming solar radiation depending on its chemical composition and physical properties [6], whereas ozone is an effective greenhouse gas (GHG) that traps outgoing terrestrial radiation and thereby increases surface temperatures [7–9]. Considering the impacts on air quality, health, economy, and climate, the coordinated control of both $PM_{2.5}$ and O_3 is highly desirable. Notably, achieving several of the sustainable development goals (SDGs) set by the United Nations [10] is not possible

without mitigating air pollution including both $PM_{2.5}$ and O_3. Several regions of the world have already been experiencing pollution levels exceeding the air quality standards set by the World Health Organization (WHO). Recently revised standards [11, 12] for $PM_{2.5}$ (24-h mean: 15 μg m^{-3}) and O_3 (8-h mean: 100 μg m^{-3}) require even more stringent interventions to mitigate air pollution.

Anthropogenically sourced $PM_{2.5}$, besides having primary emissions from the combustion of fossil and biomass-based fuels, is also formed through reactions of inorganics (e.g. SO_2, NO_x, NH_3, etc.) and volatile-organic precursors. Ozone (O_3) on the other hand does not have direct emission and is formed in the atmosphere through a complex set of chemistry involving the oxidation of carbon monoxide (CO), methane (CH_4), and volatile organic compounds (VOCs) and photolytic dissociation of NO_2 to NO. In presence of O_3, higher concentrations of NO_x or SO_2 can produce nitrate or sulphate and thereby enhance the $PM_{2.5}$ concentrations in the atmosphere [13, 14].

The dependence of O_3 on its precursors is highly non-linear and reductions in some precursors (which are also precursors for particulate matter) may enhance O_3 pollution [15, 16]. $PM_{2.5}$, with an atmospheric lifetime of about a week, can readily be transported over local to regional scales while surface O_3 (with a lifetime of more than a month) can be transported over longer, inter-continental, and hemispheric scales depending on the prevailing meteorological conditions [17]. Historically, the problems of elevated levels of $PM_{2.5}$ and O_3 have been confined to specific seasons [18]. In general, wintertime stagnation and meteorological inversions cause severe haze contributed by $PM_{2.5}$ [19–21], whereas, in contrast, during hot summer conditions, the intense solar radiation favours the photochemical formation of ozone. However, in addition to seasonal trends, several major cities around the globe experience higher levels of both $PM_{2.5}$ and O_3 in spring as well as summer [22]. Additionally, due to the aforementioned roles of chemistry and atmospheric dynamics, air pollution is no longer confined to highly urbanised regions but is being experienced in rural and remote locations as well [23, 24]. Higher $PM_{2.5}$ levels are seen in parts of South Asia, East Asia, the Middle East, Southeast Asia, and Africa regions throughout the year [25]. However, the widespread enhancements are most pronounced typically during winter over the Indo-Gangetic Plain (IGP) and the North-China Plain (NCP) [21] The studies show higher NO_2 levels ($\geq$ 16 ppbv) generally near the source regions, i.e. over the IGP, eastern China and North America, whereas O_3-rich airmasses are seen to be distributed more uniformly also over remote mountains and oceanic regions [17, 24, 26, 27].

Dramatic reductions in aerosol loadings can allow more sunlight to reach near the Earth's surface resulting in more ozone production [28]. Heterogeneous losses of O_3 and precursors are also lower due to less aerosol surface area under lower $PM_{2.5}$ conditions, which can also contribute to the enhanced O_3 pollution [29, 30]. Such enhancements in ground-level O_3 were profound during the recent socio-economic slowdown caused by the COVID-19 (coronavirus disease-2019) lockdown restrictions [31]. Furthermore, from an air quality perspective, both local/regional and inter-continental sources impose additional burden on pollution episodes during unfavourable meteorological conditions [21, 32, 33]. The winter-time stagnant conditions substantially increase the severity of haze episodes, whereas the heatwaves during summer lead to excess biogenic emissions of VOCs with escalated chemistry of tropospheric O_3 production [34].

Figure 1 highlights the key connections of $PM_{2.5}$ and O_3 concentrations with emissions and meteorology. An increase in $PM_{2.5}$ reduces incoming solar radiation and photolysis rates thereby reducing O_3 production. This decrease in incoming solar radiation also reduces surface temperature leading to a general decrease in planetary boundary layer height (PBLH). Reduced PBLH means decreased turbulent mixing of all pollutants in the boundary layer which leads to increased surface concentrations of both $PM_{2.5}$ and O_3. Daily, seasonal, and long-term variations in the PBLH, as well as levels of pollutants, are strongly governed by synoptic and seasonal weather patterns. For example, substantial build-up in $PM_{2.5}$ levels corresponding to reduced BLH due to synoptic forcing is seen in several megacities in NCP, Sichuan Basin, and Central China [35]. On a seasonal basis, severe $PM_{2.5}$ pollution is evident during winters when the average PBLH is lowest [25, 36].

Designing science-driven strategies to simultaneously reduce $PM_{2.5}$ and O_3 is a win–win solution toward clean air and climate change mitigation. Nevertheless, highly complex and non-linear chemical processes and atmospheric dynamics make it very challenging to pinpoint the driving sources/mechanisms over local-to-regional scales. In recent years, the availability of the state-of-the-art ground-based instrumentation [37, 38], more reliable satellite retrievals [39, 40], and models accounting for the detailed atmospheric chemistry [41, 42] have provided valuable insights into the atmospheric budgets of $PM_{2.5}$ and O_3.

Here, we review the latest state-of-the-science of $PM_{2.5}$-O_3 interactions to inform strategies for their co-reduction, leading to improved air quality and mitigating the health impacts.

Fine Particulates ($PM_{2.5}$): Importance, Processes, and Reduction Challenges

The fine airborne particulate matter with an aerodynamic diameter of 2.5 μm or less ($PM_{2.5}$) is a leading climate and environmental health risk factor. These tiny solid or liquid airborne particles, ranging from a few nanometres

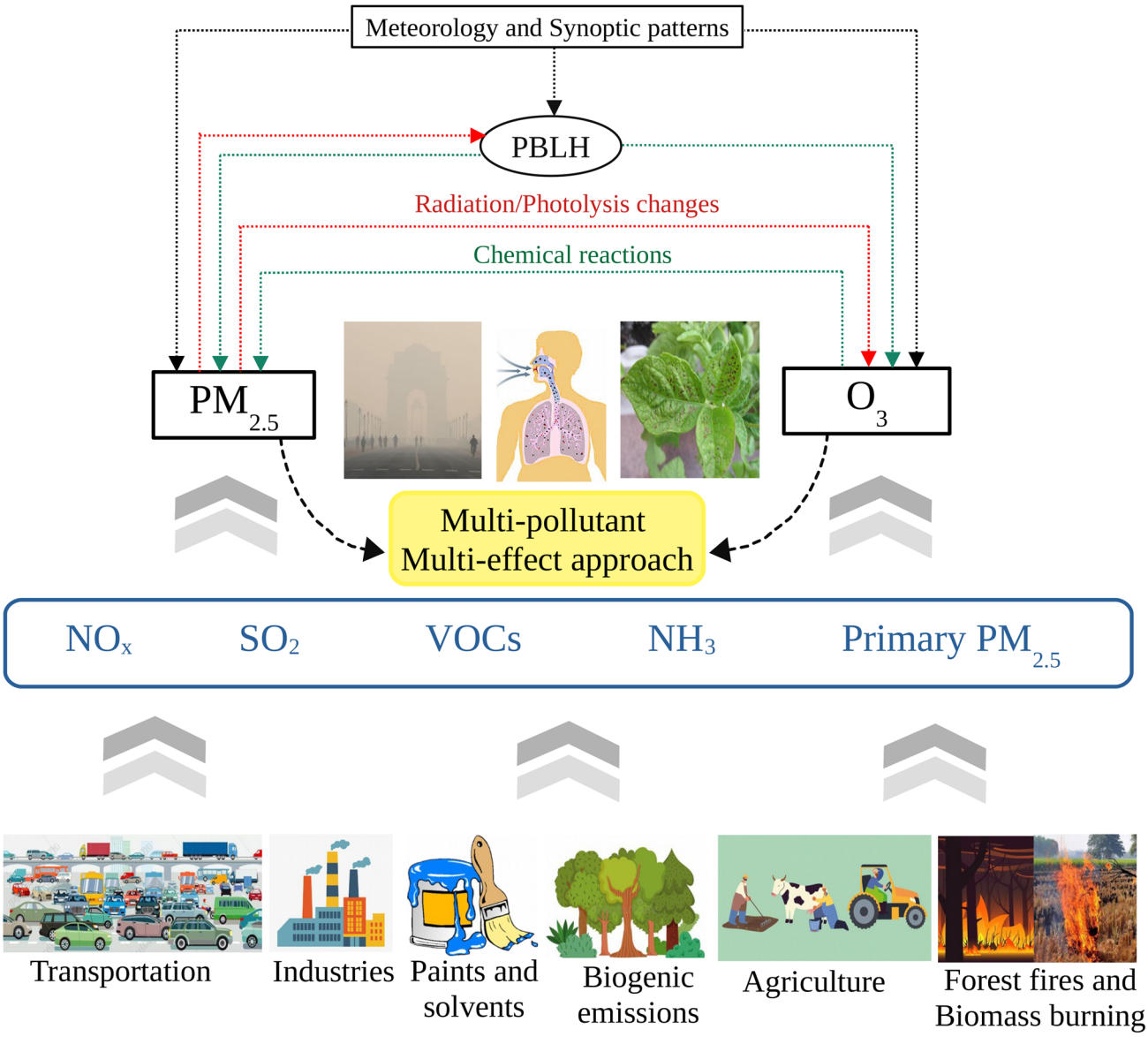

Fig. 1 A summary of key processes governing the variations in O_3 and $PM_{2.5}$ in the atmosphere. Red lines denote decreasing effects with an increase in $PM_{2.5}$ while green lines denote increasing effects

to several micrometres in size, influence the energy budget of the Earth [43, 44] and boundary layer dynamics through scattering and absorption of solar radiation. Further, aerosols act as cloud condensation nuclei (CCN) and can increase or decrease rainfall thus modifying the hydrologic cycle [45, 46]. Being emitted from a variety of sources and with multiple societal impacts, $PM_{2.5}$ is a key target under the United Nations (UN) Agenda 2030. The motivation behind such expedited moves toward $PM_{2.5}$ reduction is considering its impacts on climate, health, and economy. Recent estimates of the World Health Organization (WHO) suggest that the $PM_{2.5}$-induced annual global mortality is as high as 4.1 million in 2016. The major causes of mortalities from $PM_{2.5}$ include stress on the cardiovascular and respiratory organ systems. The cases of morbidity and mortality are seen especially higher (> 90%) in developing countries [47].

A recent synthesis from the State of Global Air [48] suggests an explicit geographical heterogeneity has been observed in the impacts of $PM_{2.5}$ on health and mortality patterns showing the highest burden over Asia and Africa. The combination of dense population and poor air quality has led to higher pre-mature mortalities in China and India. The episodic nature of air pollution in some regions of China additionally elevates the health risks associated with $PM_{2.5}$ [49]. Steep enhancement in $PM_{2.5}$-induced mortality is also predicted for sub-Saharan, North Africa, and the Middle

East [48]. The burden of diseases has high economic costs associated with hospitalisation, loss of working days, treatment costs, and premature mortalities. The potential increase in $PM_{2.5}$ and pollutants like tropospheric ozone is projected to raise the global healthcare costs from USD 21 billion in 2015 to USD 176 billion in 2060 [50].

The $PM_{2.5}$ composition is chiefly constituted by inorganic ions, carbonaceous compounds, and some fractions of mineral dust. Besides direct emissions through, $PM_{2.5}$ is also formed in the atmosphere via chemical reactions involving precursors through gas-to-particle conversion [51]. New particle formation (NPF) via gas-to-particle conversion can occur under diverse environmental conditions such as urban locations, forest areas, marine/coastal regions, and the remote or free troposphere [52]. A recent study has suggested a major role of NPF (~65% of the number concentration of haze particles from NPF) in causing present-day haze events in Beijing [53]. Among various sources, solid-fuel combustion (mostly coal) constitutes nearly 27% of the total global $PM_{2.5}$ burden, eliminating which can help in avoiding 1.05 million deaths annually [54]. In the last decade, about 55% of the global population was exposed to higher levels of $PM_{2.5}$ with varying spatial concentrations and impacts across the world. A decline in the population-weighted $PM_{2.5}$ concentration from 12.4 to 9.8 μgm^{-3} was observed over North America and Europe but a sharp increase from 54.8 to 61.5 μgm^{-3} was noticed over Central and Southern Asia [55, 56]. Among major megacities in the world, Toronto (Canada), Miami and New York (USA), and Madrid (Spain) were identified among the least polluted with concentrations ranging between 7 and 10 $\mu g\ m^{-3}$, while Delhi (India), Cairo (Egypt), Xi'an, Tianjin, and Chengdu (China) had experienced highest annual average $PM_{2.5}$ (89–143 $\mu g\ m^{-3}$) [57]. A systematic time-series analysis [58] found population, urban ratio, and vegetation greenness as key socio-economic drivers of $PM_{2.5}$. Studies have revealed that the effects of different sources on human health can be different since all the components of $PM_{2.5}$ are not equally toxic [59]. Out of all major sources such as biomass burning, diesel vehicles, and dust to $PM_{2.5}$, biomass burning was positively co-related with respiratory illness in the St. Louis metropolis [60]. Besides PM mass, nowadays, $PM_{2.5}$ oxidative potential (OP) is considered a metric to understand PM toxicity [59]. The largest contribution to the cellular OP was from SOA (>54%) in the urban region but from agricultural activities (62%) in the rural region of the mid-west US. However, these sources which were the largest contributors to the cellular OP were not the major contributors (≤21%) to the $PM_{2.5}$ mass.

An econometric study of China suggests positive associations between $PM_{2.5}$ concentrations and the size of the urban sprawl, population density, share of industry, and developmental indices like gross developmental product (GDP). The urban sprawl and increased economic change have resulted in enhanced $PM_{2.5}$ emissions over a large region of China [61]. An interesting observation was made that the expansion of urban set-ups in developed countries results in the reduction in $PM_{2.5}$ while it increases the $PM_{2.5}$ load in developing countries [58]. Such patterns in the developing world have many climatic implications, particularly over the tropical and Northern Hemisphere mid-latitude regions due to a significant drop in the wet deposition associated with less large-scale precipitation over land [62]. Although with increased awareness and stringent policy implementations, steep declines in $PM_{2.5}$ are being recognised in some of the emission hotspots in Asia. An emerging dipole has been observed in the columnar aerosol loading with decreasing patterns over China while increasing over India [63, 64].

To tackle the impacts of $PM_{2.5}$ on the climate, health, and socio-economic sustainability, worldwide efforts have been observed toward the reduction of ambient concentration as well as minimising human exposure. The U.S. Clean Air Act, the European Union Clean Air Policy, the Action Plan on Prevention and Control of Air Pollution (APPCAP) in China [15], and the Indian National Clean Air Program (NCAP) are some examples of major policy-oriented efforts. Even with the unequivocal global consensus on reducing $PM_{2.5}$ reductions, the spatiotemporal heterogeneity in their sources and complex formation chemistry makes it a cumbersome task. The total mass concentration and chemical composition of $PM_{2.5}$ depend on the local meteorological conditions and the nature and strength of emission sources. Moreover, air pollution is a transboundary phenomenon and does not adhere to geopolitical boundaries. Developing a precise understanding of the spatio-temporal trends of $PM_{2.5}$ and its drivers and chemical and meteorological interactions of its precursors is hence of utmost importance before aiming to design effective mitigation measures.

Ozone: Importance, Processes, and Control Pathways

Higher concentrations of O_3 pose adverse impacts on human health including respiratory illnesses such as asthma exacerbation, chronic obstructive pulmonary disease (COPD), lung and cardiovascular diseases, etc [65]. Furthermore, O_3 damages terrestrial vegetation by reducing photosynthesis, altering carbon allocation, and impairing stomatal function causing visible foliar injury and disturbing whole-plant level responses [66, 67]. Besides tropospheric ozone's role as a criteria pollutant, it is also a potent greenhouse gas which significantly contributes to global warming [11]. The photochemical reactions involving VOCs, CO, and methane (CH_4) in presence of NO_x result in the production of O_3 [6]. The efficiency of O_3 production depends on the reactivity of VOC with hydroxyl

radical (OH) and the ratio of VOC/NO_x concentration [6]. O_3 is lost through chemical pathways involving its reactions with water vapour (after photolysis), radicals (HO, HO_2, NO), and through dry deposition at the surface. Besides gas-phase sinks, heterogeneous reactions of O_3 with aerosols contribute to O_3-sink processes [68]. The global and regional scale meteorology (e.g. temperature, relative humidity (RH), wind speed and direction, and precipitation) affects the chemical conditions for O_3 production.

Recent investigations revealed a positive correlation between O_3 and temperature, driven by faster chemistry and enhanced natural emissions, e.g. isoprene [69, 70]. Such deterioration of air quality due to global warming—even without an increase in anthropogenic emissions—is considered the "climate penalty" [71]. Therefore, efforts to reduce anthropogenic emissions for controlling O_3 pollution could be offset by the projected rise in global surface temperatures in the future. In addition, enhanced biogenic VOCs due to higher temperatures can also increase the production of secondary organic aerosols [72]. This increase in aerosols may increase cloud condensation nuclei (CCN) and affect cloud albedo [72]. Such feedback, starting with climate warming (higher temperatures) either intensifies or reduces the warming depending on various atmospheric factors such as meteorology, location, topography, properties of aerosols, and concentration of reactive trace gases in the region. Stronger NO_x emissions through soil microbes have been observed to contribute nearly half of the O_3 increase in a rural site in the USA with increasing temperature [73]. Additionally, meteorological variability could affect O_3 production by modulating NO_x emissions by lightning [74], wildfire emissions [75], and methane emissions from wetlands [76] and shale gas [77].

The multiphase processes associated with the emissions of reactive halogen species also alter the ground-level O_3 chemistry [65]. Over the oceans, the halogen species (mainly involving iodine and bromine) catalytically react with O_3 resulting in lower O_3 levels [78, 79]. On the contrary, in rural-urban environments of as northern China and India, chlorine-related species are reported to accelerate the oxidation of several VOCs resulting in the enhancement of O_3 [80, 81]. Additionally, the oxidation of VOCs by chlorine radicals could be a potent source of secondary organic aerosol formation [82]. Therefore, it is suggested that reducing chlorine emissions along with reducing VOCs and NO_x could be beneficial in reducing O_3 as well as SOA formation and thus reducing $PM_{2.5}$.

Challenges and Mitigation Pathways for Co-controlling $PM_{2.5}$ and O_3

The effects of particulate-precursor chemistry must be carefully considered for designing an effective policy for $PM_{2.5}$ and O_3 control. The relative reduction of fine particles and the reduced rate of hydroperoxyl radical removal help in the escalation of ozone formation [83]. Further, attempts to reduce oxides of nitrogen (NO_x) can exacerbate the ambient O_3 pollution in urban regions, while equally controlling NO_x and volatile organic compounds (VOCs) may have minimal effect on air quality improvement.

In view of highly complex atmospheric chemistry, Xiang et al. [84] suggested a stepwise strategy with focussing first on VOCs and then on NO_x. Such attempts have resulted in the reduction of anthropogenic VOC emissions by ~60% and NO_x emissions by ~20% in the first stage, while the removal of the rest of the VOCs and NO_x emissions in the second stage in China. Several South Asian urban centres also experience the influence of local traffic emissions and regionally transported pollution affecting the ambient concentrations of $PM_{2.5}$ and O_3. In such environments, only reducing the local particulate emissions will not be effective in controlling the overall air pollution. A study over Delhi [42] showed that reducing the regional emissions along with the local emissions can significantly reduce the local $PM_{2.5}$ by 25–30% without increasing O_3 over Delhi.

To address the $PM_{2.5}$ burden with O_3 in highly polluted regions, global efforts involving both developed and developing megacities are needed. Tropospheric Ozone Assessment Report (TOAR; https://igacproject.org/activities/TOAR) and Monitoring, Analysis, and Prediction of Air Quality (MAP-AQ; https://igacproject.org/activities/map-aq) are few important initiatives toward assessment and mitigation of air pollution. Further works also aiming at energy optimisation and application of advanced emission-reducing technologies globally are needed. Since $PM_{2.5}$ and O_3 have some common precursors and O_3 itself affects chemistry leading to $PM_{2.5}$ formation, a multi-pollutant multi-effect approach should be applied for co-control of O_3 and $PM_{2.5}$. Akimoto et al. [85] recommended a stepwise approach in which first the anthropogenic emissions of NO_x and SO_2 are to be controlled simultaneously. This step will help reduce nitrate (NO_3^-) and sulphate (SO_4^{2-}), the key constituents of inorganic $PM_{2.5}$. As a precursor of O_3, reducing NO_x will also reduce O_3 in NO_x-limited environments. Additionally, VOCs may get reduced when common sources of NO_x and SO_2 (e.g. fossil fuel burning) are controlled. In the next step, NO_x and VOC emissions should be reduced simultaneously to directly reduce O_3 significantly. This will also reduce secondary $PM_{2.5}$ since both NO_x and VOCs are important sources of secondary aerosols. This strategy can reduce $PM_{2.5}$ and O_3 simultaneously as it focuses on preferential control of the secondary pollutants. Few studies reported that NO_x reductions in the first step may exacerbate the O_3 pollution in polluted urban environments. For example, Xiang et al. [84] targeted the precursors contributing to both O_3 and $PM_{2.5}$ precursors, i.e. NO_x and VOCs, over the Beijing-Tianjin-Hebei region and observed that

reducing NO_x enhances O_3 due to VOC-limited chemistry [86, 87], especially where O_3 pollution is severe. However, reducing NO_x and VOC emissions together led to small improvements in air quality. In this view, they proposed a "first VOC-focused, then NO_x-focused" strategy. In the first step, reducing anthropogenic VOC emissions by 60% and NO_x emissions by 20% and in the next step reducing more VOCs and NO_x. This policy was suggested to control both $PM_{2.5}$ and O_3 pollution to a greater extent.

An analysis over Delhi [42] showed that local traffic emissions from within the city and transport of pollution from the National Capital Region (NCR) surrounding Delhi were the dominant factors influencing $PM_{2.5}$ and O_3. Reducing traffic emissions by 50% in Delhi alone reduces $PM_{2.5}$ by 15–20% but increases O_3 by 20–25%. Reducing domestic emissions can decrease $PM_{2.5}$, although lesser than the reduction achieved by reducing traffic emissions but without increasing O_3. This could be possible since domestic emissions might not be a major source of NO_x as traffic emissions. Moreover, VOCs are reduced more than NO_x when controlling domestic emissions i.e. VOC/NO_x ratio was 1.8 in contrast to a ratio of 0.4 for traffic emissions. Greater reductions of VOCs suppress the increase in O_3 in Delhi, which is a VOC-limited environment [88–90]. However, the O_3 formation regime may vary for different sites within Delhi depending on VOCs/NO_x [28, 91]. Reducing NCR regional emissions at the same time reducing traffic emissions in Delhi by 25–30% would further reduce the $PM_{2.5}$ in Delhi by 5–10% and avoid an O_3 increase [42].

Therefore, reducing VOC and NO_x simultaneously with a larger reduction in VOCs compared to NO_x can be an effective step to co-mitigate the $PM_{2.5}$ and O_3. Conclusively, reducing common sources of $PM_{2.5}$ and O_3 will help reduce both together. Therefore, studies with more detailed information on specific emission sectors and their contribution to the polluted regions are required for effective mitigation policies. Since modelling of sector-specific and species-specific source contributions is computationally expensive, statistically trained reduced-form models as well as machine learning-based models can be used for exploring potentials of multi-sector and multi-species emission reductions [92, 93].

A holistic model evaluation approach is required to develop a correct understanding of the co-control of $PM_{2.5}$ and O_3. Model simulations with reduced NO_x and VOC emissions for O_3 control should evaluate not only the modelled O_3 but also the modelled NO_x and VOC against observations. This highlights the need for continuous monitoring especially covering the periods when emissions were also reduced in reality. Since reducing NO_x and VOCs will also reduce secondary $PM_{2.5}$, therefore, in addition to validating the simulated $PM_{2.5}$ against the observations, several individual SOA should also be validated—this requires chemically-resolved monitoring of aerosol components. Additionally, quantification of the primary and secondary fractions of the $PM_{2.5}$ burden for different regions is highly recommended. The reduction of primary aerosols is relatively straightforward and can ensure rapid improvement of air quality depending on its contribution to the total $PM_{2.5}$ mass.

COVID-19 Lockdown: a Reality Check

Despite large emission reductions during the COVID-19 lockdown, unexpected changes in O_3 and $PM_{2.5}$ were experienced in different regions of the world (Table 1). Mean $PM_{2.5}$ levels generally showed reductions by about 30–40% but dust transport, biomass burning, and secondary PM formation caused enhancements in some parts of Europe and Asia [94, 95]. O_3 exhibited diverse trends with insignificant change or slight increase over Europe but 25–30% higher levels over East Asia and South America [94]. Enhanced relative humidity by 30–50% and reduced wind speed under lower PBL height during winter promoted the multiphase chemistry which enhanced SOA levels during lockdown in China [96]. Additionally, the reduced NO_x contributed to the enhancement in O_3 levels due to non-linear chemistry. The higher O_3 levels increased the atmospheric oxidising capacity and facilitated enhanced SOA and SO_4 formation and further contributed to $PM_{2.5}$ loading [96]. While there was a significant reduction in $PM_{2.5}$ (13–29%) as well as NO_2 (39–53%), O_3 levels were found to be higher by a factor of 1.6–2 in northern China [97, 98].In the USA, changes in NO_2 from 5 to 49% are mainly due to lower transportation and utility demands [99]. Significant reductions (up to 45%) in $PM_{2.5}$ were seen in Northeast and California/Nevada metropolises, where NO_2 declined strongly. Minor changes (within ± 20%) in O_3 concentration were seen. The Sao Paulo region in Brazil experienced extreme reductions in NO_x (~54–77%), and $PM_{2.5}$ (~29.8%) but ~30% increase in the O_3 concentrations, compared to the 5-year monthly mean [100]. Here, lesser changes in the levels of pollutants were observed in the industrial areas, as industries were not restricted to shut down, but they were partially affected by the decreased demand.

Satellite-based observations revealed up to 50% reductions in boundary layer NO_2 over urban areas of Europe [101]. The aircraft-based measurements under the BLUESKY campaign showed large variabilities in reactive nitrogen and aerosols (20–70%) over German cities during the lockdown as compared to multi-year average data [101]. In situ measurements across the UK also showed reductions in NO_2 (up to 48%) but a slight increase in O_3 by ~11% [102]. In some other European sites, the NO_x and $PM_{2.5}$ concentrations were observed to be reduced by 50–71% and

Table 1 A list of some studies from different regions of the world describing the changes in the levels of various pollutants during the COVID-19 lockdown period

Location	Pollutants		
	NO_2 (% reduction)	$PM_{2.5}$ (% change)	O_3 (% change)
Global [94, 95]	48–72	17–45	−2 to 10
North China [97, 110]	39–53	13–39	100–160
USA [99]	5–49	−45 to 50	±20
Sao Paulo, Brazil [100]	54–77	29.8	30
UK [102]	41–48	–	11
Spain [104]	50–69	13–50	2.4–50
Nice, France [103]	63	3	24
Delhi [105, 106]	40–60	40–60	20
Ahmedabad [107]	43–55	–	41
Kolkata [108]	–	20–66	11–91
Thiruvananthapuram [109]	40	–	−36

3–50%, respectively, while O_3 increased by ~ 2.4–50% [103, 104]. Such reductions were also attributed to the restrictions on population mobility leading to reduced road traffic, and industrial operations.

The reductions in some of the key air pollutants were also pronounced over India during the COVID-19 lockdown exhibiting a sharp decline in $PM_{2.5}$ (~25–60%) and mixed changes in O_3 across different regions [105, 106]. In Delhi, O_3 concentrations showed more or less site-specific trends with a mean increase of ~ 20%. This increase could be attributed to the significant reductions (~40–60%) in NO_2, PM_{10}, and $PM_{2.5}$ levels [106], which enhanced photochemical O_3 production via non-linear chemistry and more solar radiation in conditions of lower aerosol burden. Ahmedabad, an urban hotspot in western India experienced ~41% enhancement in O_3 levels in contrast to the reduction in NO_x (~43–55%), which was attributed to chemistry (25%) and meteorology (16%) [107]. Here, enhancement in NO_2/NO ratio during the lockdown (3.3) in comparison to the pre-lockdown period (2.6) results in the lesser titration of O_3 as during the lockdown period [107]. Kolkata over the eastern region of the Indo-Gangetic Plain showed a decrease in $PM_{2.5}$ levels by 20–66%, but an increase in O_3 levels by 11–91% during five different phases of lockdown [108]. In contrast, Thiruvananthapuram, a tropical coastal site in India experienced reduced daytime O_3 (~36%) accompanied by reduced daytime NO_2 (~40%) [109]. Similar reductions in the O_3 were also seen in the south and central regions of India [106]. Both PM and NO_2 show a strong reduction due to less fuel consumption in various economic sectors i.e. transport (50–60%), aviation (90%), industries (40%), and construction activities (70%), even with a 12% increase in household fuel consumption, while O_3 was increased in the populated areas of India [111].

Heterogeneities in the levels of pollutants were attributed to the complex air chemistry, meteorology, and episodic events such as dust, biomass burning, and crop fertilising [94, 95]. Such insights from the societal slowdown and reduced anthropogenic activities during the COVID-19 restrictions, albeit not intended for co-mitigation of $PM_{2.5}$ and O_3 but for containing the pandemic, suggest the achievable improvements in the air quality by curbing major emissions, which were not economically and socially viable otherwise.

Conclusion

The challenges to reducing air pollution greatly depend on the sources of specific pollutants, the effects of atmospheric dynamics and complex chemical interactions under various meteorological conditions. Exposure to both $PM_{2.5}$ and ground-level O_3 has been reported to have deleterious impacts on human and plant health and the built environment. Hence, a strong scientific consensus is building globally for co-controlling both $PM_{2.5}$ and ground-level O_3 for achieving several overarching air quality benefits. The complex interplay between the reduction in the hydroperoxyl radical removal helps in the escalation of ozone formation while efforts to reduce NO_x exacerbate ambient O_3 (in several polluted urban environments). Further, equally controlling NO_x and VOCs results in the marginal improvement of air quality. $PM_{2.5}$ and O_3 have some common precursors; therefore, a multi-pollutant multi-effect approach to co-controlling them is urgently needed including a stepwise approach for controlling NO_x and SO_2 emissions simultaneously from the anthropogenic sources first, which will lower down NO_3^- and SO_4^{2-} while as a precursor of O_3, reducing NO_x will further reduce the levels of O_3. The societal slowdown during COVID-19 with consequent lowered energy consumption provided a natural experiment and evidence for orienting future research on co-control of both pollutants. Further, it highlighted a strong prerequisite for further scientific investigation to understand the $PM_{2.5}$-O_3 interactions over the polluted regions.

Compliance with Ethical Standards

Conflict of Interest On behalf of all authors, the corresponding authors state that there is no conflict of interest.

Human and Animal Rights and Informed Consent This article does not contain any studies with human or animal subjects performed by any of the authors.

References

1. Lelieveld J, Evans JS, Fnais M, Giannadaki D, Pozzer A. The contribution of outdoor air pollution sources to premature mortality on a global scale. Nature. 2015;525(7569):367–71. https://doi.org/10.1038/nature15371.
2. Ghude SD, Chate DM, Jena C, et al. Premature mortality in India due to PM2.5 and ozone exposure. Geophys Res Lett. 2016. https://doi.org/10.1002/2016GL068949.
3. Zhou L, Chen X, Tian X. The impact of fine particulate matter (PM2.5) on China's agricultural production from 2001 to 2010. J Clean Prod. 2018;178:133–41. https://doi.org/10.1016/J.JCLEPRO.2017.12.204.
4. Avnery S, Mauzerall DL, Liu J, Horowitz LW. Global crop yield reductions due to surface ozone exposure: 1. Year 2000 crop production losses and economic damage. Atmos Environ. 2011. https://doi.org/10.1016/j.atmosenv.2010.11.045.
5. Sharma A, Ojha N, Pozzer A, Beig G, Gunthe SS. Revisiting the crop yield loss in India attributable to ozone. Atmos Environ X. 2019. https://doi.org/10.1016/j.aeaoa.2019.100008.
6. Seinfeld JH, Pandis SN. Atmospheric chemistry and physics: from air pollution to climate change. New York: John Wiley & Sons; 2006.
7. Krishna RK, Panicker AS, Yusuf AM, Ullah BG. On the contribution of particulate matter (PM 2.5) to direct radiative forcing over two urban environments in India. Aerosol Air Qual Res. 2019. https://doi.org/10.4209/aaqr.2018.04.0128.
8. Rowlinson MJ, Rap A, Hamilton DS, et al. Tropospheric ozone radiative forcing uncertainty due to pre-industrial fire and biogenic emissions. Atmos Chem Phys. 2020;20(18):10937–51. https://doi.org/10.5194/acp-20-10937-2020.
9. Jasaitis D, Vasiliauskiene V, Chadyšiene R, Pečiuliene M. Surface ozone concentration and its relationship with UV radiation, meteorological parameters and radon on the eastern coast of the Baltic sea. Atmosphere (Basel). 2016. https://doi.org/10.3390/atmos7020027.
10. United Nations. Resolution adopted by the General Assembly on 25 September 2015. Transforming our world: The 2030 Agenda for Sustainable Development. Published online 2015.
11. Masson-Delmotte V, Zhai P, Pirani A, Connors SL, Péan C, Berger S, et al. (eds. IPCC, 2021: Climate change 2021: the physical science basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change. 2021.
12. World Health Organization. WHO global air quality guidelines: particulate matter (PM2.5 and PM10), ozone, nitrogen dioxide, sulfur dioxide and carbon monoxide. World Health Organization. Published online 2021.
13. Ge B, Sun Y, Liu Y, et al. Nitrogen dioxide measurement by cavity attenuated phase shift spectroscopy (CAPS) and implications in ozone production efficiency and nitrate formation in Beijing, China. J Geophys Res Atmos. 2013;118(16):9499–509. https://doi.org/10.1002/jgrd.50757.
14. Li L, Chen ZM, Zhang YH, Zhu T, Li JL, Ding J. Kinetics and mechanism of heterogeneous oxidation of sulfur dioxide by ozone on surface of calcium carbonate. Atmos Chem Phys. 2006;6(9):2453–64. https://doi.org/10.5194/acp-6-2453-2006.
15. Zhang Q, Zheng Y, Tong D, et al. Drivers of improved PM2.5 air quality in China from 2013 to 2017. Proc Natl Acad Sci U S A. 2019. https://doi.org/10.1073/pnas.1907956116.
16. Hidy GM, Blanchard CL. Precursor reductions and ground-level ozone in the Continental United States. J Air Waste Manage Assoc. 2015;65(10):1261–82. https://doi.org/10.1080/10962247.2015.1079564.
17. Girach IA, Tripathi N, Nair PR, Sahu LK, Ojha N. O3 and CO in the South Asian outflow over the Bay of Bengal: impact of monsoonal dynamics and chemistry. Atmos Environ. 2020;233:117610. https://doi.org/10.1016/J.ATMOSENV.2020.117610.
18. Fowler D, Brimblecombe P, Burrows J, et al. A chronology of global air quality. Philos Trans A Math Phys Eng Sci. 2020;378(2183):20190314. https://doi.org/10.1098/rsta.2019.0314.
19. Ansari TU, Ojha N, Chandrasekar R, Balaji C, Singh N, Gunthe SS. Competing impact of anthropogenic emissions and meteorology on the distribution of trace gases over Indian region. J Atmos Chem. 2016;73:363–80. https://doi.org/10.1007/s10874-016-9331-y.
20. Ansari TU, Wild O, Li J, et al. Effectiveness of short-term air quality emission controls: a high-resolution model study of Beijing during the Asia-Pacific Economic Cooperation (APEC) summit period. Atmos Chem Phys. 2019;19(13):8651–68. https://doi.org/10.5194/acp-19-8651-2019.
21. Ojha N, Sharma A, Kumar M, et al. on the widespread enhancement in fine particulate matter across the Indo-Gangetic Plain towards winter. https://doi.org/10.1038/s41598-020-62710-8.
22. Chen J, Shen H, Li T, Peng X, Cheng H, Ma C. Temporal and spatial features of the correlation between $PM_{2.5}$ and O_3 concentrations in China. Published online 2019. https://doi.org/10.3390/ijerph16234824.
23. Ravishankara AR, David LM, Pierce JR, Venkataraman C. Outdoor air pollution in India is not only an urban problem. Proc Natl Acad Sci U S A. 2020. https://doi.org/10.1073/pnas.2007236117.
24. Soni M, Girach I, Sahu LK, Ojha N. Photochemical evolution of air in a tropical urban environment of India: a model-based study. Chemosphere. 2022;297:134070. https://doi.org/10.1016/J.CHEMOSPHERE.2022.134070.
25. Ojha N, Soni M, Kumar M, Girach I, Sharma SK, Gunthe SS. Air pollution episodes: brief history, mechanisms and outlook. Published online 2022:283–301. https://doi.org/10.1007/978-981-16-7727-4_12.
26. Lawrence MG, Lelieveld J. Atmospheric pollutant outflow from southern Asia: a review. 2010. Atmos Chem Phys Discuss. https://doi.org/10.5194/acp-10-11017-2010.
27. Imran G, Ojha N, Nair P, et al. Variations in O3, CO, and CH4 over the Bay of Bengal during the summer monsoon season: shipborne measurements and model simulations. Atmos Chem Phys. 2017;17:257–75. https://doi.org/10.5194/acp-17-257-2017.
28. Chen Y, Beig G, Archer-Nicholls S, et al. Avoiding high ozone pollution in Delhi, India. Faraday Discuss. 2021. https://doi.org/10.1039/d0fd00079e.
29. Li K, Jacob DJ, Liao H, et al. A two-pollutant strategy for improving ozone and particulate air quality in China. Nat Geosci. 2019. https://doi.org/10.1038/s41561-019-0464-x.
30. 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 U S A. 2019. https://doi.org/10.1073/pnas.1812168116.

31. Korhale N, Anand V, Beig G. Disparity in ozone trends under COVID-19 lockdown in a closely located coastal and hillocky metropolis of India. Air Qual Atmos Heal. 2021. https://doi.org/10.1007/s11869-020-00958-9.
32. Bikkina S, Andersson A, Kirillova EN, et al. Air quality in megacity Delhi affected by countryside biomass burning. Nat Sustain. 2019. https://doi.org/10.1038/s41893-019-0219-0.
33. Kanawade VP, Srivastava AK, Ram K, et al. What caused severe air pollution episode of November 2016 in New Delhi? Atmos Environ. 2020;222:117125. https://doi.org/10.1016/J.ATMOSENV.2019.117125.
34. Ma M, Gao Y, Wang Y, et al. Substantial ozone enhancement over the North China Plain from increased biogenic emissions due to heat waves and land cover in summer 2017. Atmos Chem Phys. 2019;19(19):12195–207. https://doi.org/10.5194/acp-19-12195-2019.
35. Miao Y, Li J, Miao S, et al. Interaction between planetary boundary layer and PM2.5 pollution in megacities in China: a review. Curr Pollut Reports. 2019;5(4):261–71. https://doi.org/10.1007/s40726-019-00124-5.
36. Miao Y, Liu S, Huang S. Synoptic pattern and planetary boundary layer structure associated with aerosol pollution during winter in Beijing, China. Sci Total Environ. 2019;682:464–74. https://doi.org/10.1016/J.SCITOTENV.2019.05.199.
37. Gorai AK, Tchounwou PB, Mitra G. Spatial variation of ground level ozone concentrations and its health impacts in an urban area in India. Aerosol Air Qual Res. 2017. https://doi.org/10.4209/aaqr.2016.08.0374.
38. Wei J, Li Z, Li K, et al. Full-coverage mapping and spatiotemporal variations of ground-level ozone (O3) pollution from 2013 to 2020 across China. Remote Sens Environ. 2020. https://doi.org/10.1016/j.rse.2021.112775.
39. Zhang XY, Zhao LM, Cheng MM, Chen DM. Estimating ground-level ozone concentrations in eastern China using satellite-based precursors. IEEE Trans Geosci Remote Sens. 2020. https://doi.org/10.1109/TGRS.2020.2966780.
40. Wang W, Liu X, Bi J, Liu Y. A machine learning model to estimate ground-level ozone concentrations in California using TROPOMI data and high-resolution meteorology. Environ Int. 2022. https://doi.org/10.1016/j.envint.2021.106917.
41. Xing J, Wang SX, Jang C, Zhu Y, Hao JM. Nonlinear response of ozone to precursor emission changes in China: a modeling study using response surface methodology. Atmos Chem Phys. 2011;11(10):5027–44. https://doi.org/10.5194/acp-11-5027-2011.
42. Chen Y, Wild O, Conibear L, et al. Local characteristics of and exposure to fine particulate matter (PM2.5) in four Indian megacities. Atmos Environ X. 2020. https://doi.org/10.1016/j.aeaoa.2019.100052.
43. Schwartz SE. The whitehouse effect—shortwave radiative forcing of climate by anthropogenic aerosols: an overview. J Aerosol Sci. 1996;27(3):359–82. https://doi.org/10.1016/0021-8502(95)00533-1.
44. Zhang Y, Wen XY, Jang CJ. Simulating chemistry–aerosol–cloud–radiation–climate feedbacks over the continental U.S. using the online-coupled Weather Research Forecasting Model with chemistry (WRF/Chem). Atmos Environ. 2010;44(29):3568–82. https://doi.org/10.1016/J.ATMOSENV.2010.05.056.
45. Ramanathan V, Crutzen PJ, Kiehl JT, Rosenfeld D. Aerosols, climate, and the hydrological cycle. Science. 2001;294(5549):2119–24. https://doi.org/10.1126/science.1064034.
46. Rosenfeld D, Lohmann U, Raga GB, et al. Flood or drought: how do aerosols affect precipitation? Science. 2008;321(5894):1309–13. https://doi.org/10.1126/science.1160606.
47. World Health Organization. Ambient (outdoor) air pollution, Fact sheet 2021. https://www.who.int/news-room/fact-sheets/detail/ambient-(outdoor)-air-quality-and-health. [Accessed: 11-Aug-2022]
48. Keel J, Walker K, Pant P. Air Pollution and its impacts on health in Africa -insights from the State of Global Air 2020. Clean Air J. 2020;30:1–2. http://www.scielo.org.za/scielo.php?script=sci_arttext&pid=S2410-972X2020000200005&nrm=iso. [Accessed: 11-Aug-2022]
49. Gao M, Guttikunda SK, Carmichael GR, et al. Health impacts and economic losses assessment of the 2013 severe haze event in Beijing area. Sci Total Environ. 2015;511:553–61. https://doi.org/10.1016/j.scitotenv.2015.01.005.
50. Lanzi E, Dellink R, Chateau J, et al. The economic consequences of outdoor air pollution. 2016. https://doi.org/10.1787/9789264257474-en.
51. Sharma SK, Kumar M, Rohtash, et al. Characteristics of ambient ammonia over Delhi. India Meteorol Atmos Phys. 2014. https://doi.org/10.1007/s00703-013-0299-8.
52. Lee S-H, Gordon H, Yu H, et al. New particle formation in the atmosphere: from molecular clusters to global climate. https://doi.org/10.1029/2018JD029356.
53. Kulmala M, Dada L, Daellenbach KR, et al. Is reducing new particle formation a plausible solution to mitigate particulate air pollution in Beijing and other Chinese megacities? Faraday Discuss. 2021;226:334–47. https://doi.org/10.1039/D0FD00078G.
54. Mcduffie EE, Martin RV, Spadaro JV, et al. Source sector and fuel contributions to ambient PM 2.5 and attributable mortality across multiple spatial scales. https://doi.org/10.1038/s41467-021-23853-y.
55. Shaddick G, Thomas ML, Mudu P, Ruggeri G, Gumy S. Half the world's population are exposed to increasing air pollution. https://doi.org/10.1038/s41612-020-0124-2.
56. Morawska L, Zhu T, Liu N, et al. The state of science on severe air pollution episodes: Quantitative and qualitative analysis. Environ Int. 2021;156:106732. https://doi.org/10.1016/J.ENVINT.2021.106732.
57. Cheng Z, Luo L, Wang S, et al. Status and characteristics of ambient PM2.5 pollution in global megacities. Environ Int. 2016;89–90:212–21. https://doi.org/10.1016/J.ENVINT.2016.02.003.
58. Lim CH, Ryu J, Choi Y, Jeon SW, Lee WK. Understanding global PM2.5 concentrations and their drivers in recent decades (1998–2016). Environ Int. 2020;144:106011. https://doi.org/10.1016/J.ENVINT.2020.106011.
59. Wang Y, Puthussery JV, Yu H, Liu Y, Salana S, Verma V. Sources of cellular oxidative potential of water-soluble fine ambient particulate matter in the Midwestern United States. J Hazard Mater. 2021;2022(425):127777. https://doi.org/10.1016/j.jhazmat.2021.127777.
60. Krall JR, Mulholland JA, Russell AG, et al. Associations between source-specific fine particulate matter and emergency department visits for respiratory disease in four U.S. cities. Environ Health Perspect. 2017;125(1):97–103. https://doi.org/10.1289/EHP271.
61. Wang S, Zhou C, Wang Z, Feng K, Hubacek K. The characteristics and drivers of fine particulate matter (PM2.5) distribution in China. J Clean Prod. 2017;142:1800–9. https://doi.org/10.1016/J.JCLEPRO.2016.11.104.
62. Allen JT, Tippett MK, Sobel AH. An empirical model relating U.S. monthly hail occurrence to large-scale meteorological environment. https://doi.org/10.1002/2014MS000397.
63. Samset BH, Lund MT, Bollasina M, Myhre G, Wilcox L. Emerging Asian aerosol patterns. https://doi.org/10.1038/s41561-019-0424-5.
64. Ramachandran S, Rupakheti M, Lawrence MG. Aerosol-induced atmospheric heating rate decreases over South and East Asia as a result of changing content and composition. Published online. 2020. https://doi.org/10.1038/s41598-020-76936-z.
65. US EPA. Integrated Science Assessment (ISA) for ozone and related photochemical oxidants (Final Report). US Environ Prot Agency. Published online 2020.

66. Tai APK, Sadiq M, Pang JYS, Yung DHY, Feng Z. Impacts of surface ozone pollution on global crop yields: comparing different ozone exposure metrics and incorporating co-effects of CO2. Front Sustain Food Syst. 2021. https://doi.org/10.3389/fsufs.2021.534616.
67. Betzelberger AM, Yendrek CR, Sun J, et al. Ozone exposure response for U.S. soybean cultivars: linear reductions in photosynthetic potential, biomass, and yield 1[W][OA]. Plant Physiol Ò. 2012;160:1827–39. https://doi.org/10.1104/pp.112.205591.
68. Lou S, Liao H, Zhu B. Impacts of aerosols on surface-layer ozone concentrations in China through heterogeneous reactions and changes in photolysis rates. Atmos Environ. 2014;85:123–38. https://doi.org/10.1016/J.ATMOSENV.2013.12.004.
69. Pusede SE, Steiner AL, Cohen RC. Temperature and recent trends in the chemistry of continental surface ozone. Chem Rev. 2015. https://doi.org/10.1021/cr5006815.
70. Coates J, Mar KA, Ojha N, Butler TM. The influence of temperature on ozone production under varying NOx conditions - a modelling study. Atmos Chem Phys. 2016;16(18):11601–15. https://doi.org/10.5194/acp-16-11601-2016.
71. Wu S, Mickley LJ, Leibensperger EM, Jacob DJ, Rind D, Streets DG. Effects of 2000–2050 global change on ozone air quality in the United States. J Geophys Res Atmos. 2008. https://doi.org/10.1029/2007JD008917.
72. Paasonen P, Asmi A, Petäjä T, et al. Warming-induced increase in aerosol number concentration likely to moderate climate change. Nat Geosci. 2013. https://doi.org/10.1038/ngeo1800.
73. Romer PS, Duffey KC, Wooldridge PJ, et al. Effects of temperature-dependent NOx emissions on continental ozone production. Atmos Chem Phys. 2018;18(4):2601–14. https://doi.org/10.5194/acp-18-2601-2018.
74. Nassar R, Logan JA, Megretskaia IA, Murray LT, Zhang L, Jones DBA. Analysis of tropical tropospheric ozone, carbon monoxide, and water vapor during the 2006 El Niño using TES observations and the GEOS-Chem model. J Geophys Res Atmos. 2009. https://doi.org/10.1029/2009JD011760.
75. Jaffe DA, Wigder NL. Ozone production from wildfires: a critical review. Atmos Environ. 2012. https://doi.org/10.1016/j.atmosenv.2011.11.063.
76. Shindell DT, Pechony O, Voulgarakis A, et al. Interactive ozone and methane chemistry in GISS-E2 historical and future climate simulations. 2013. Atmos Chem Phys. https://doi.org/10.5194/acp-13-2653-2013.
77. Pozzer A, Schultz MG, Helmig D. Impact of U.S. oil and natural gas emission increases on surface ozone is most pronounced in the Central United States. Environ Sci Technol. 2020. https://doi.org/10.1021/acs.est.9b06983.
78. Sarwar G, Gantt B, Schwede D, Foley K, Mathur R, Saiz-Lopez A. Impact of enhanced ozone deposition and halogen chemistry on tropospheric ozone over the Northern Hemisphere. Environ Sci Technol. 2015. https://doi.org/10.1021/acs.est.5b01657.
79. Tuite K, Brockway N, Colosimo SF, et al. Iodine catalyzed ozone destruction at the Texas Coast and Gulf of Mexico. Geophys Res Lett. 2018. https://doi.org/10.1029/2018GL078267.
80. Liu X, Qu H, Huey LG, et al. High levels of daytime molecular chlorine and nitryl chloride at a rural site on the North China Plain. Environ Sci Technol. 2017. https://doi.org/10.1021/acs.est.7b03039.
81. Gunthe SS, Liu P, Panda U, et al. Enhanced aerosol particle growth sustained by high continental chlorine emission in India. Nat Geosci. 2021. https://doi.org/10.1038/s41561-020-00677-x.
82. Wang DS, Ruiz LH. Secondary organic aerosol from chlorine-initiated oxidation of isoprene. 2017. Atmos Chem Phys. https://doi.org/10.5194/acp-17-13491-2017.
83. Hollaway M, Wild O, Yang T, et al. Photochemical impacts of haze pollution in an urban environment. 2019. Atmos Chem Phys. https://doi.org/10.5194/acp-19-9699-2019.
84. Xiang S, Liu J, Tao W, et al. Control of both PM2.5 and O3 in Beijing-Tianjin-Hebei and the surrounding areas. Atmos Environ. 2020;224:117259. https://doi.org/10.1016/J.ATMOSENV.2020.117259.
85. Asia Pacific Clean Air Partnership (APCAP) Policy Brief 1 – PM 2.5 and Ozone Co-Control – March 2019, Clean Air Brief 2019. https://cleanairweek.org/documents/. 2019.
86. Ren J, Xie S. Diagnosing ozone-NOx-VOC sensitivity and revealing causes of ozone increases in China based on 2013–2021 satellite retrievals. Atmos Chem Phys Discuss. 2022;2022:1–22. https://doi.org/10.5194/acp-2022-347.
87. Tan Z, Lu K, Jiang M, et al. Exploring ozone pollution in Chengdu, southwestern China: a case study from radical chemistry to O3-VOC-NOx sensitivity. Sci Total Environ. 2018;636:775–86. https://doi.org/10.1016/J.SCITOTENV.2018.04.286.
88. Nelson BS, Stewart GJ, Drysdale WS, et al. In situ ozone production is highly sensitive to volatile organic compounds in Delhi, India. Atmos Chem Phys. 2021;21(17):13609–30. https://doi.org/10.5194/acp-21-13609-2021.
89. Kumar U, Prakash A, Jain VK. A photochemical modelling approach to investigate O3 sensitivity to NOx and VOCs in the urban atmosphere of Delhi. Aerosol Air Qual Res. 2008;8(2):147–59. https://doi.org/10.4209/aaqr.2007.09.0037.
90. Tripathi N, Sahu LK, Wang L, et al. Characteristics of VOC composition at urban and suburban sites of New Delhi, India in winter. J Geophys Res Atmos. 2022;127(12):e2021JD035342. https://doi.org/10.1029/2021JD035342.
91. Sharma A, Sharma SK, Mandal TK. Ozone sensitivity factor: NOX or NMHCs?: a case study over an urban site in Delhi, India. Urban Clim. 2021;39:100980. https://doi.org/10.1016/J.UCLIM.2021.100980.
92. Ansari TU, Wild O, Ryan E, Chen Y, Li J, Wang Z. Temporally resolved sectoral and regional contributions to air pollution in Beijing: informing short-term emission controls. Atmos Chem Phys. 2021;21(6):4471–85. https://doi.org/10.5194/acp-21-4471-2021.
93. Conibear L, Silver BJ, Chen Y, Knote C, Arnold SR, Spracklen DV. Statistical emulation of winter ambient fine particulate matter concentrations from emission changes in China. GeoHealth. 2021. https://doi.org/10.1029/2021GH000391.
94. Sokhi RS, Singh V, Querol X, et al. A global observational analysis to understand changes in air quality during exceptionally low anthropogenic emission conditions. Environ Int. 2021;157:106818. https://doi.org/10.1016/J.ENVINT.2021.106818.
95. Venter ZS, Aunan K, Chowdhury S, Lelieveld J. COVID-19 lockdowns cause global air pollution declines. PNAS. 2020;117(32):18984–90. https://doi.org/10.1073/pnas.2006853117/-/DCSupplemental.
96. Le T, Wang Y, Liu L, et al. Unexpected air pollution with marked emission reductions during the COVID-19 outbreak in China. Science. 2020. https://doi.org/10.1126/science.abb7431.
97. Shi X, Brasseur GP. The response in air quality to the reduction of Chinese economic activities during the COVID-19 outbreak. Geophys Res Lett. 2020. https://doi.org/10.1029/2020GL088070.
98. Li K, Jacob DJ, Liao H, et al. Ozone pollution in the North China Plain spreading into the late-winter haze season. Proc Natl Acad Sci. 2021;118(10):e2015797118. https://doi.org/10.1073/pnas.2015797118.
99. Chen LWA, Chien LC, Li Y, Lin G. Nonuniform impacts of COVID-19 lockdown on air quality over the United States. Sci

Total Environ. 2020;745:141105. https://doi.org/10.1016/J.SCITOTENV.2020.141105.
100. Nakada LYK, Urban RC. COVID-19 pandemic: impacts on the air quality during the partial lockdown in São Paulo state, Brazil. Sci Total Environ. 2020;730:139087. https://doi.org/10.1016/J.SCITOTENV.2020.139087.
101. Voigt C, Lelieveld J, Schlager H, et al. Cleaner skies during the COVID-19 lockdown. Bull Am Meteorol Soc. 2022;3:1–57. https://doi.org/10.1175/bams-d-21-0012.1.
102. Lee JD, Drysdale WS, Finch DP, Wilde SE, Palmer PI. UK surface NO2 levels dropped by 42% during the COVID-19 lockdown: Impact on surface O3. Atmos Chem Phys. 2020;20(24):15743–59. https://doi.org/10.5194/acp-20-15743-2020.
103. Sicard P, De Marco A, Agathokleous E, et al. Amplified ozone pollution in cities during the COVID-19 lockdown. Sci Total Environ. 2020;735:139542. https://doi.org/10.1016/j.scitotenv.2020.139542.
104. Tobías A, Carnerero C, Reche C, et al. Changes in air quality during the lockdown in Barcelona (Spain) one month into the SARS-CoV-2 epidemic. Sci Total Environ. 2020. https://doi.org/10.1016/j.scitotenv.2020.138540.
105. Zhang M, Katiyar A, Zhu S, et al. Impact of reduced anthropogenic emissions during COVID-19 on air quality in India. Atmos Chem Phys. 2021;21(5):4025–37. https://doi.org/10.5194/acp-21-4025-2021.
106. Singh RP, Chauhan A. Impact of lockdown on air quality in India during COVID-19 pandemic. Air Qual Atmos Heal. 2020. https://doi.org/10.1007/s11869-020-00863-1.
107. Soni M, Ojha N, Girach I. Impact of COVID-19 lockdown on surface ozone build-up at an urban site in western India based on photochemical box modelling. Curr Sci. 2021;120(2). https://doi.org/10.18520/cs/v120/i2/376-381.
108. Mondal A, Sharma SK, Mandal TK, Girach I, Ojha N. Frequency distribution of pollutant concentrations over Indian megacities impacted by the COVID-19 lockdown. Environ Sci Pollut Res. 2021. https://doi.org/10.1007/s11356-021-16874-z.
109. Girach IA, Ojha N, Babu SS. Ozone chemistry and dynamics at a tropical coastal site impacted by the COVID-19 lockdown. J Earth Syst Sci. 2021. https://doi.org/10.1007/s12040-021-01666-3.
110. Li R, Xu M, Li M, et al. Identifying the spatiotemporal variations in ozone formation regimes across China from 2005 to 2019 based on polynomial simulation and causality analysis. Atmos Chem Phys. 2021;21(20):15631–46. https://doi.org/10.5194/acp-21-15631-2021.
111. Singh V, Singh S, Biswal A, Kesarkar A P, Mor S, Ravindra K, Diurnal and temporal changes in air pollution during COVID-19 strict lockdown over different regions of India, Environmental Pollution,Volume 266, Part 3, 2020, https://doi.org/10.1016/j.envpol.2020.115368.