



Cite this: *Environ. Sci.: Atmos.*, 2024, **4**, 73

Enhanced summertime $\text{PM}_{2.5}$ -suppression of O_3 formation over the Eastern U.S. following the O_3 -sensitivity variations†

Jie Zhang, ^a Junfeng Wang, ^b Alexandra Catena, ^a Margaret J. Schwab, ^a Matthew Ninneman, ^c Dirk Felton ^c and James Schwab, ^a ^{*a}

The suppression of ozone (O_3) formation due to the presence of fine particulate matter ($\text{PM}_{2.5}$) has recently been highlighted for further O_3 pollution controls in regions that suffer high ozone concentrations. Here we derive multiple $\text{PM}_{2.5}$ -suppression factors for the Eastern United States (U.S.) major cities based on a non-linear fitting of the $\text{PM}_{2.5}$ and O_3 relationship from the multiyear surface observations. Our results show that these $\text{PM}_{2.5}$ -suppression factors are increasing with time and generally follow the transition of the O_3 -sensitive regime towards NO_x -limited chemistry. A spatial discrepancy of this suppression factor is seen currently with a higher value in the Southeastern U.S. than in the Northeastern U.S. A spatial similarity between urban regions and their downwind locations was observed for the New York City metro area. This more extensive formulation of the $\text{PM}_{2.5}$ -suppression factor will further improve the ability of models to help guide O_3 and $\text{PM}_{2.5}$ concentration pollution controls.

Received 21st March 2023
Accepted 20th November 2023

DOI: 10.1039/d3ea00040k
rsc.li/esatmospheres

Environmental significance

Ozone and fine particulate matter ($\text{PM}_{2.5}$) remain troublesome air pollution problems for a large number of areas, including metropolitan areas in the Northeastern U.S. like New York City. The paper's findings shed light on the interplay of these pollutants, namely the role that enhanced $\text{PM}_{2.5}$ can play in suppressing ozone formation. The magnitude of ozone suppression from $\text{PM}_{2.5}$ provides an additional indicator of the sensitivity of ozone formation from its VOC and NO_x precursors. The $\text{PM}_{2.5}$ suppression of ozone formation per unit of $\text{PM}_{2.5}$ mass concentration has been increasing over the period 2004–2019 in the Northeast U.S., indicating a transition from VOC-limited to NO_x -limited ozone formation sensitivity. It also provides guidance for further O_3 - $\text{PM}_{2.5}$ studies and pollution control regulations.

1. Introduction

The Eastern United States (U.S.) has been marked as a region persistently suffering from the co-occurrence of summertime ozone (O_3) and particulate matter (with a diameter under 2.5 μm , $\text{PM}_{2.5}$) pollution during the summertime.¹ However, these summertime concentrations of O_3 and $\text{PM}_{2.5}$ in the region's major cities have shown decreasing trends since the 1970s as a result of the implementation of emission control policies.^{2–6} While the region has seen reductions in both pollutants, extreme concentrations of O_3 and $\text{PM}_{2.5}$ (defined as the top 5%

of measured values in a given year) in New York City (NYC) have shown different overall reductions with a more significant reduction for $\text{PM}_{2.5}$ than O_3 . This can be attributed to the different reduction rates of their precursors, with the control policies targeting sulfur dioxide (SO_2) and primary $\text{PM}_{2.5}$, than volatile organic compounds (VOCs) and nitrogen oxides (NO_x).⁷

Co-occurrence of summertime maximum daily 8 h average O_3 (hereafter: MDA8 O_3) and the daily 24 h average $\text{PM}_{2.5}$ (hereafter: DA24 $\text{PM}_{2.5}$ for simplification) based on ground measurements in NYC has shown a direct relationship between the pollutants,⁷ with a monotonically increasing near-linear relationship for low PM concentrations. A leveling-off or even decreasing relationship for high PM concentrations was observed in megacity clusters in China.^{7–10} This flat or declining relationship has been partly attributed to the scavenging of hydroperoxy (HO₂)/nitrate radicals (NO₃) by high concentrations of $\text{PM}_{2.5}$ that inhibits the photochemical production of O_3 ,^{11–15} or reduced photolysis rates with $\text{PM}_{2.5}$ increasing.¹⁶ A number of model simulations have used a uniform reactive uptake coefficient for HO₂ on aerosols ($\gamma_{\text{HO}_2} = 0.2$)^{9,17–21} to focus

^aAtmospheric Sciences Research Center, University at Albany, State University of New York, Albany, NY 12226, USA. E-mail: jschwab@albany.edu; jzhang35@albany.edu

^bJiangsu Key Laboratory of Atmospheric Environment Monitoring and Pollution Control, School of Environmental Science and Engineering, Nanjing University of Information Science and Technology, Nanjing 210044, China

^cDivision of Air Resources, New York State Department of Environmental Conservation, Albany, NY, USA

† Electronic supplementary information (ESI) available. See DOI: <https://doi.org/10.1039/d3ea00040k>

on (1) studying emission control policies for O_3 pollution reduction with the understanding that reduced $PM_{2.5}$ concentration promotes more efficient O_3 formation^{9,17–20} and (2) possible consideration of a third ‘aerosol inhibited’ regime for O_3 formation in addition to the regular VOC-limited and NO_x -limited regimes.²¹

According to Zhang *et al.* (2022),⁷ a non-linear polynomial function can be used to depict the O_3 – $PM_{2.5}$ co-relationships for NYC based on surface measurements, with a positive linear term reflecting the O_3 /PM_{2.5} co-occurrence and a negative power function term reflecting the O_3 formation suppression by PM_{2.5} (hereafter PM_{2.5}-suppression factor). The PM_{2.5}-suppression factors were also identified and likely to change along with the O_3 -sensitivity of the chemical regime in NYC.⁷ To verify the connection between the PM_{2.5}-suppression factors and the O_3 -sensitive regime existing over a larger region, major cities of the Eastern U.S. and their downwind regions are selected in this study, with a focus on the Long Island Sound and surrounding area located just downwind of NYC.^{22–24} This study will focus on the spatial variation of the PM_{2.5}-suppression factors over several major Eastern U.S. cities and also from urban to downwind regions (NYC as an example) and the temporal variation from 2004 to 2019 to explore the relationship between the PM_{2.5}-suppression factors and the corresponding O_3 -sensitivity regimes over this regional scale.

2. Methods

2.1 Study locations and periods

Five air quality sites in NYC and downwind locations over Long Island Sound (Fig. S1† for the locations) were chosen as examples of the PM_{2.5}-suppression factor variation from urban to downwind regions. These five sites include (a) two urban sites: the IS52 site in Bronx County of NYC and the Queens College (QC) site in Queens County of NYC in New York State, and (b) three downwind sites: the Eisenhower Park (EP) site in western Long Island in New York State, the Holtsville (HOL) site in central Long Island, and the Criscuolo Park (CP) site in New Haven in Connecticut state, which was chosen because it is frequently influenced by sea breezes that transport NYC urban outflow plumes.^{22–24} The IS52, QC, EP, and HOL sites belong to the New York State Department of Environmental Conservation (NYSDEC), and the CP site belongs to the Connecticut Department of Energy and Environmental Protection. In addition to these five sites for NYC and downwind locations, another 11 major Eastern U.S. cities with the U.S. Environmental Protection Agency measurement sites for O_3 and PM_{2.5} in the urban regions were chosen to investigate the spatial/temporal variation of the PM_{2.5}-suppression factor. To better cover the study period (2004–2019) with a daily report for O_3 and PM_{2.5} of each city, we generally select the data from 2–3 sites in or near (within 10 km) the downtown regions, with the location of the representative site of each city shown in Table S1.† The study period is from 2004 to 2019 during summertime (June, July, and August), which was separated into three subperiods based on the PM_{2.5} variation (Fig. S2,† SP1: 2004–2008; SP2: 2009–2013, SP3: 2014–2019). These MDA8 O_3 and the PM_{2.5} from the above

sites can be found at <https://www.epa.gov/outdoor-air-quality-data/download-daily-data>.

2.2 Non-linear fitting of the O_3 –PM_{2.5} relationship

Eqn (1) includes (1) a positive linear term to reflect the O_3 /PM_{2.5} co-occurrence with its slope influenced by the aerosol chemical composition (this slope varies as a function of the atmospheric chemical composition, which in turn varies with emission controls), (2) a negative power function term with an exponent of 5/3 reflecting the suppression of O_3 formation by PM_{2.5}, (*i.e.*, the uptake of HO₂/NO₃ by PM_{2.5}, *etc.*), and (3) a constant with the possibility of implying the background O_3 without PM_{2.5}.⁷ The power function exponent was set to 5/3, based on the consideration that (a) the uptake coefficients of the radicals related to the aerosol surface concentration which is expected to be proportional to the 2/3 power of PM_{2.5} mass concentration, and (b) the radical concentrations were simply assumed to relate with the O_3 concentration, which is proportional to the PM_{2.5} mass concentration as mentioned in the above positive linear term.

$$O_3 = aPM_{2.5} - b(PM_{2.5})^{5/3} + c \quad (1)$$

The coefficient a is the slope of the linear term, b is the power function coefficient, and c is the constant. These three factors will be obtained through non-linear fitting. It should be noted that eqn (1) only represents a very idealized solution for the non-linear O_3 –PM_{2.5} relationship, with oversimplified terms and factors, and will cause some uncertainty for the results. More specifically, the impact of aerosol on the O_3 formation could be much more complicated with additional or different processes, including other heterogeneous reactions besides the uptake of HO₂/NO₃ by PM_{2.5}, changes to photolysis rates and direct radiative forcing, changes to meteorological conditions, *i.e.*, boundary layer height and ventilation, temperature and wind speed, *etc.*^{25,26} All of these would complicate the equation and introduce new terms to better describe the relationship between PM_{2.5} and O_3 . Further studies related to the mechanism are warranted to explore a more accurate function.

The power function coefficient is defined as the ‘PM_{2.5}-suppression factor’, which indicates the magnitude of suppression of O_3 formation by PM_{2.5} at the same PM_{2.5} level when doing spatial/temporal comparisons, and will be the focus of this study. In this study, the MDA8 O_3 and the DA24 PM_{2.5} concentrations were used for the O_3 and PM_{2.5} values. The PM_{2.5} data were initially binned following the approach in Li *et al.*⁸ and Buysse *et al.*²⁷ with increments of 5 $\mu\text{g m}^{-3}$, which was used to ensure enough statistical points in each bin. However, any site with only four bins when using the increment of 5 $\mu\text{g m}^{-3}$ was binned in increments of 4 $\mu\text{g m}^{-3}$ to ensure sufficient points for fitting. In addition, discrete bins having only one or two highest PM_{2.5} mass concentrations were left out, as they are considered quite probably to be due to the influence of other factors, such as the extreme wildfire smoke plumes. It should be noted that the binned dataset used in this study would constitute a simplified but empirically valid mechanism for the MDA8 O_3 and DA24 PM_{2.5} relationship.



2.3 O₃-sensitivity regime

The ratio of the HCHO column concentration and NO₂ column concentration from the Ozone Monitoring Instrument (OMI)^{28,29} was used as the indicator for the early afternoon O₃-sensitivity regime (considering the OMI overpass time around 01:30 pm at local time) for each site as described by Jin *et al.* (2020),²⁷ and the data can be found from <https://giovanni.gsfc.nasa.gov/giovanni>. Jin *et al.* (2020)³⁰ determined that the high ozone probability (over 70 ppb 8 hour average) peaks at an HCHO/NO₂ ratio near 3.6 with a range of [3.2–4.1] for the average of the 7 cities they studied (Los Angeles, New York, Chicago, Washington, Pittsburgh, Atlanta, Houston). The ratios below this are roughly considered as VOC-limited chemistry, and above as NO_x-limited chemistry. In this study, a spatial range of 0.5° × 0.5° was used for each city to get the area-averaged column concentration of HCHO and NO₂, which were used further to obtain the seasonal averaged HCHO/NO₂ ratio.

3. Results and discussion

3.1 Enhanced summertime PM_{2.5}-suppression on O₃ formation over the NYC metro area from urban to downwind

The MDA8 O₃ and the DA24 PM_{2.5} concentrations were used to derive their relationships for these subperiods (SP1: 2004–2008; SP2: 2009–2013, SP3: 2014–2019) of the study period (2004–2019), and these relationships were further fitted using eqn (1) as mentioned in Section 2.2. The subperiods were divided based on the variation of the DA24 PM_{2.5} mass concentration, and the

standard deviation of the annual summertime average DA24 PM_{2.5} in each period was below 1 $\mu\text{g m}^{-3}$.⁷ A detailed description of the polynomial equation for fitting the non-linear relationship between O₃ and PM_{2.5} can be found in Zhang *et al.* (2022).⁷ Over the past 16 years, the linear slope of the O₃–PM_{2.5} relationship increased with time for both NYC urban sites and its downwind ones (Fig. 1a–e), which was verified to be related to the increased mass fraction of the secondary organic aerosol (SOA) and ammonium nitrate (NH₄NO₃), caused by the more significant reductions in the emissions of SO₂ and PM_{2.5} than those of VOCs and NO_x.⁷

For the highlighted PM_{2.5}-suppression factor in this study, a clear trend was observed in the NYC urban sites (IS52 and Queens College, Fig. 1a and b) over the past 16 years, with a slight increase from SP1 to SP2 (both near 0.1) and a significantly enhanced step from SP2 to SP3 (near 0.3). The enhanced PM_{2.5}-suppression factor is consistent with the increased HCHO/NO₂ ratio (Fig. 1f), which is used to indicate the O₃-sensitivity regime through satellite observations.²⁷ An increase in the value of the HCHO/NO₂ ratio indicates a shift of the O₃-sensitivity regime towards NO_x-limited chemistry, and could be (1) a complete change in O₃ chemistry sensitivity from VOC-limited to NO_x-limited, (2) a shift towards NO_x sensitivity (VOC-limited to weakly VOC-limited/transitional), or (3) increasingly NO_x-limited O₃ chemistry. Fig. 1f shows that the NYC urban region (IS52 and QC) shifted from a strong VOC-limited regime at SP1 (HCHO/NO₂ ~ 1.5) to a weak VOC-limited regime near the lower end of the transitional regime

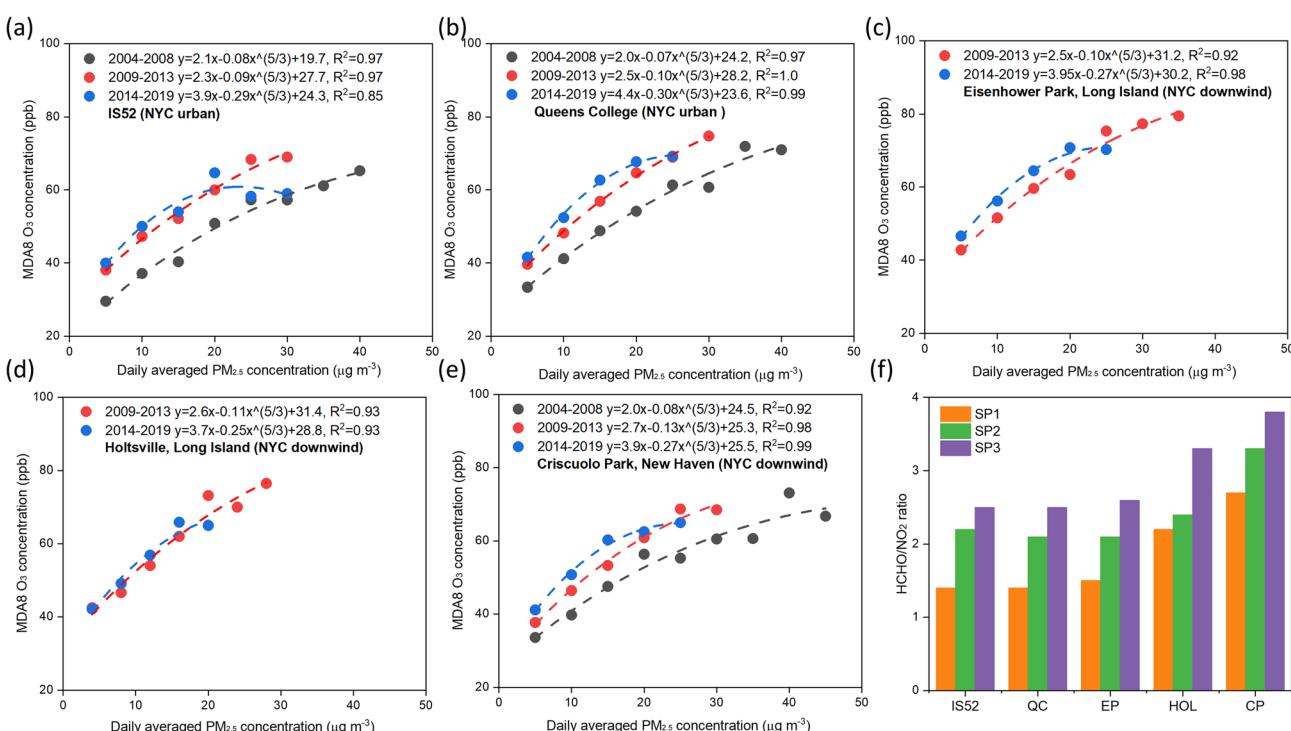


Fig. 1 (a–e) The O₃ vs. PM_{2.5} relationships over the NYC (a): the IS52 site of NYC, (b): the Queens College site of NYC and its downwind regions (c): the Eisenhower Park site of Long Island, (d): the Holtsville site of Long Island, and (e): the Criscuolo Park site of New Haven, and (f) the OMI HCHO/NO₂ ratios for the subperiods of each site (QC: Queens College; EP: Eisenhower Park; HOL: Holtsville; CP: Criscuolo Park). (SP1: 2004–2008; SP2: 2009–2013, SP3: 2014–2019).



at SP3 ($\text{HCHO}/\text{NO}_2 \sim 2.7$). However, the variation of the HCHO/NO_2 ratio comparing SP2 to SP1 (SP2 *vs.* SP1: 2.2 *vs.* 1.5, 46% increasing compared to SP1) was larger than the variation of the $\text{PM}_{2.5}$ -suppression factor (both near 0.1, Fig. 1a and b), and this generally matches the current model simulation result about the $\text{O}_x\text{-NO}_x$ relationship considering the $\text{PM}_{2.5}$ effect over Chinese urban regions from Li *et al.* (2022),³¹ which indicated that the $\text{PM}_{2.5}$ -suppression effect was weaker at higher NO_x concentrations in the VOC-limited regime, but strengthened as the O_3 -sensitivity approached the transitional regime. The weak $\text{PM}_{2.5}$ -suppression effect in the VOC-limited regime could be due to the competition for the consumption of HO_x by NO_x rather than $\text{PM}_{2.5}$, making the NO_x concentration the dominant factor for O_3 concentration sensitivity under these conditions.³¹

For the NYC downwind sites (Eisenhower Park site, Fig. 1c; Holtsville site, Fig. 1d, and Criscuolo Park site, Fig. 1e), their $\text{PM}_{2.5}$ -suppression factors for each subperiod were similar to the NYC urban sites (IS51 site, Fig. 1a and Queens College site,

Fig. 1b). This can be attributed to urban plume transport, which has been discussed in some detail from recent and current studies based on the 2018 Long Island Sound Tropospheric Ozone Study (LISTOS),^{22–24} and the formed O_3 and $\text{PM}_{2.5}$ and some unreacted precursors can be carried to downwind regions. Meanwhile, based on the fact that the averaging time period for MDA8 O_3 (8 hours) and DA24 $\text{PM}_{2.5}$ (24 hours) is generally much longer than the time scale of the photochemical reactions, it is reasonable to believe that the urban plume transport could result in the similar $\text{PM}_{2.5}$ -suppression factors for both urban and downwind sites. However, except for the Eisenhower Park site which had an HCHO/NO_2 ratio for each subperiod similar to the Queens College site as their proximity to each other (<20 km), the HCHO/NO_2 ratios of each subperiod for the Holtsville site (around 70 km from Queens College site) and the Criscuolo Park site (around 100 km from the Queens College site) were much higher than the values of the NYC urban sites (Fig. 1f). Based on satellite observations, this indicates that Holtsville

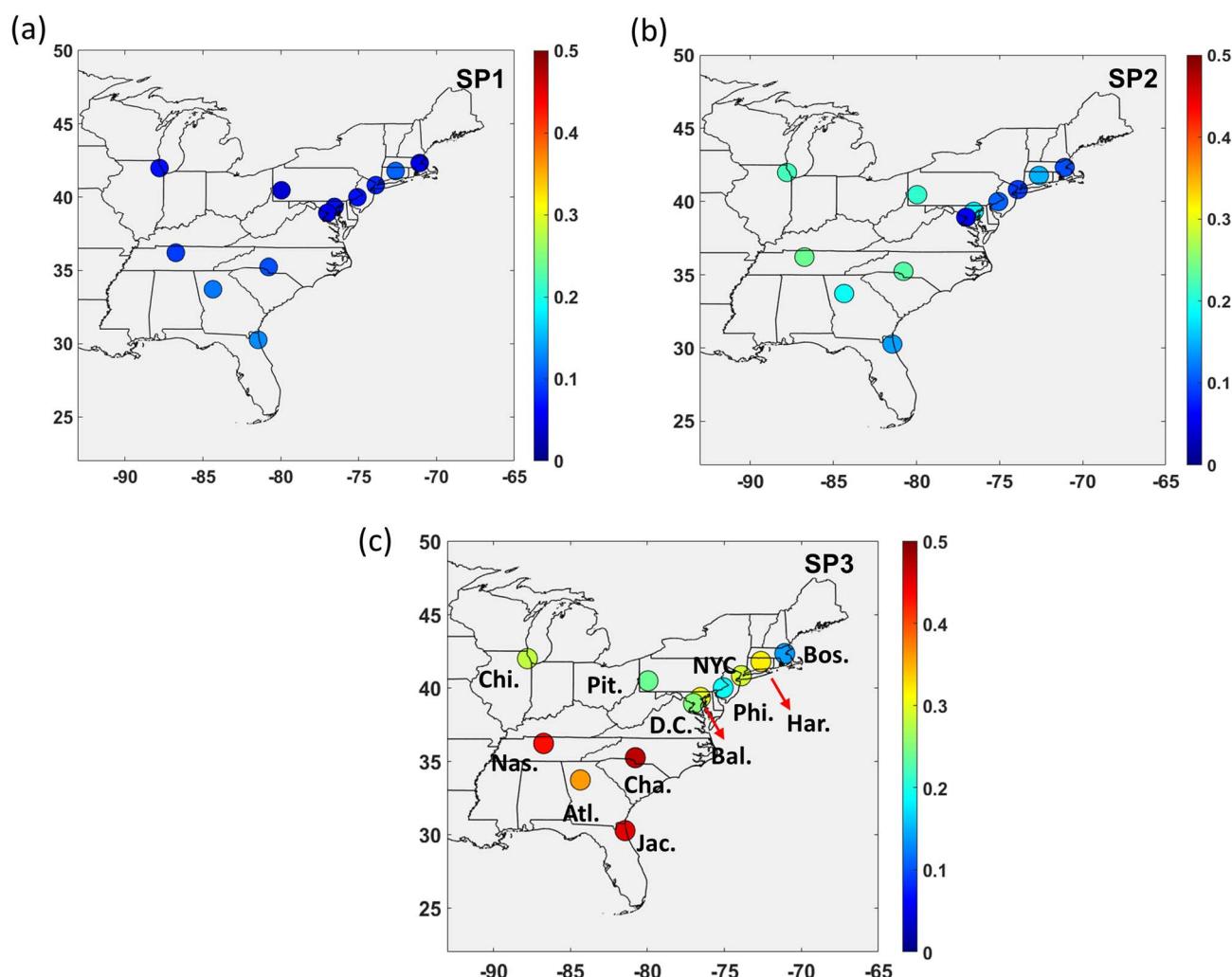


Fig. 2 The $\text{PM}_{2.5}$ -suppression factor distribution map for the twelve major cities in the Eastern U.S. for (a) Subperiod 1 (SP1:2004–2008), (b) Subperiod 2 (SP2: 2009–2013), and (c) Subperiod 3 (SP3: 2014–2019). These twelve major cities include Chicago (Chi., Illinois), Pittsburgh (Pit., Pennsylvania), Boston (Bos., Massachusetts), Hartford (Har., Connecticut), NYC (New York), Philadelphia (Phi., Pennsylvania), Baltimore (Bal., Maryland), Washington, D.C. (District of Columbia), Charlotte (Cha., North Carolina), Atlanta (Atl., Georgia), Jacksonville (Jac., Florida), and Nashville (Nas., Tennessee).



and Criscuolo Park were closer to the NO_x -limited regime than NYC urban sites. These differences between the $\text{PM}_{2.5}$ -suppression factors and the HCHO/NO_2 ratios could come from the discrepancy between the hourly/daily averaged ground measurements and the early afternoon much shorter time period of satellite data capture.

3.2 Enhanced summertime $\text{PM}_{2.5}$ -suppression on O_3 formation over major cities in the Eastern U.S.

The analysis of the increased $\text{PM}_{2.5}$ -suppression effect was also expanded to other major cities of the Eastern U.S., as shown in Fig. 2. More detailed information about the O_3 vs. $\text{PM}_{2.5}$ relationships of each city and their fitting results are shown in Fig. S3.† The $\text{PM}_{2.5}$ -suppression factors of these major cities were below/near 0.1 in SP1 (Fig. 2a and 3) and over 0.2 in SP3 (Fig. 2c and 3), and generally increased following the O_3 -sensitivity regime moving toward a stronger NO_x -limited chemistry, being indicated by their increased HCHO/NO_2 ratios from SP1 to SP3 (Fig. 3). Meanwhile, the $\text{PM}_{2.5}$ -suppression factors showed a clear spatial discrepancy throughout the Eastern U.S. comparing north and south, especially in SP3 (2014–2019). The $\text{PM}_{2.5}$ -suppression factors of the southern cities (*i.e.*, Nashville, Atlanta, Charlotte, and Jacksonville) in SP3 were near or over 0.4, which were clearly higher than the ones of the northern cities (Chicago, Pittsburgh, Boston, NYC, Philadelphia, Baltimore, and Washington, D.C.) with a major range of [0.2–0.3]. The $\text{PM}_{2.5}$ -suppression factors of these southern cities, to some extent, matched their stronger NO_x -limited chemistry, as shown by their relatively higher HCHO/NO_2 ratios. Under a stronger NO_x -limited chemistry, the O_3 concentration is more sensitive to the NO_x variation, which will enhance the effect of the NO_3 uptake by $\text{PM}_{2.5}$ on O_3 formation and match a higher $\text{PM}_{2.5}$ -suppression factor.

Based on the fact that (1) the $\text{PM}_{2.5}$ -suppression factor increases as the O_3 -sensitivity regime shifts toward stronger

NO_x -limited chemistry and (2) the relatively small variation range of the $\text{PM}_{2.5}$ -suppression factors for the cities with similar urban conditions (*i.e.*, those northeast urban cities with a range of [0.2–0.3] in SP3), it is reasonable to suppose the summertime $\text{PM}_{2.5}$ -suppression factor derived from the polynomial fits can be used to provide information for identifying the O_3 -sensitivity regimes based on the ground measurement, in addition to the satellite measurements using the HCHO/NO_2 ratios. However, it is hard to build a highly correlated relationship between the $\text{PM}_{2.5}$ -suppression factor and the HCHO/NO_2 ratios (as shown in Fig. S4†), considering the facts that (1) $\text{PM}_{2.5}$ -suppression factor changed little in the VOC-limited regime, (2) local atmospheric chemistry varied, (3) the HCHO/NO_2 ratios are for early afternoon photochemistry while the $\text{PM}_{2.5}$ -suppression factor for a daily average, *etc.* More studies are warranted for exploring the usage of the $\text{PM}_{2.5}$ -suppression factor for O_3 -sensitivity regimes and its relationship with the satellite-measured HCHO/NO_2 ratios in a world range. More especially, the future datasets from the TEMPO satellite (<https://tempo.si.edu>) with hourly fluctuations in HCHO/NO_2 values will largely promote the understanding of the hourly variation of the O_3 -sensitivity regimes and the daily averaged O_3 -sensitivity regimes, which could build a better relationship to the ground measurements/derived suppression factors from this study.

3.3 Atmospheric implications of the increased summertime $\text{PM}_{2.5}$ -suppression factor

The relationship of the $\text{PM}_{2.5}$ -suppression factor with the O_3 -sensitivity regime also indicates the necessity of considering the effect of the HO_2 uptake coefficient (γ_{HO_2}) on aerosol surfaces, which is most commonly implemented as a constant value, *i.e.*, $\gamma_{\text{HO}_2} = 0.2$ for a number of previous studies.^{15–21} The increased $\text{PM}_{2.5}$ -suppression factor following the O_3 -sensitivity regime from VOC-limited to NO_x -limited can also partly explain the

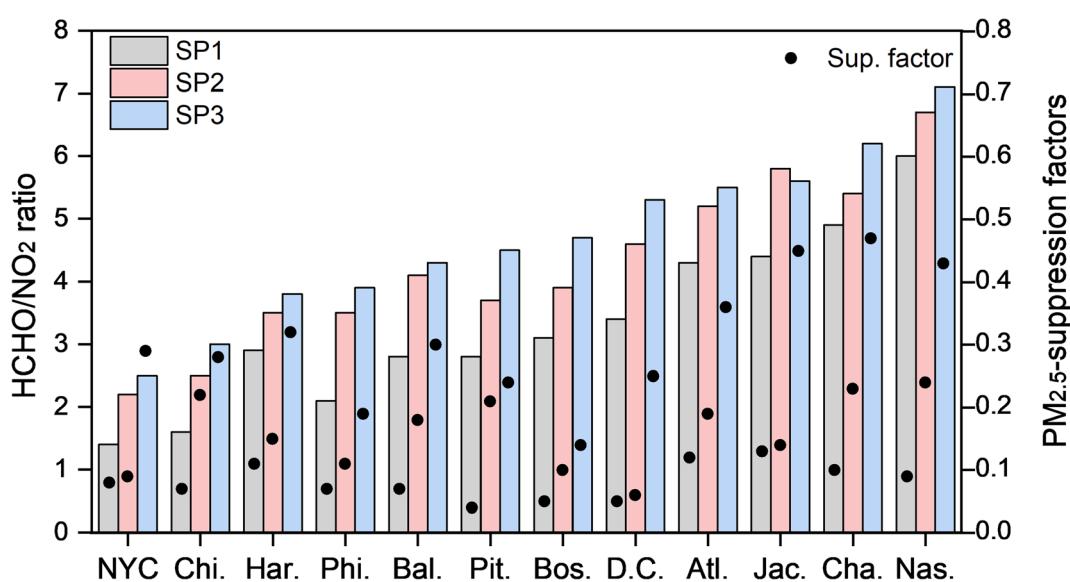


Fig. 3 The OMI HCHO/NO_2 ratios and $\text{PM}_{2.5}$ -suppression factors for the subperiods of each city mentioned in Fig. 2.



discrepancy between the ambient measurement of HO_2 uptake from a rural site in North China Plain (Wangdu County, Hebei province, China) by Tan *et al.* (2020)³² and the model result for Chinese urban areas from Li *et al.* (2019).⁸ Given that Wangdu County was located at a VOC-limited regime in 2014,^{32,33} it is reasonable to infer that its $\text{PM}_{2.5}$ -suppression factor was relatively low, considering (1) the rural Wangdu measurement site in 2014 was highly influenced by polluted regions upwind (Baoding City with a distance of about 35 km and Shijiazhuang City with a distance of about 85 km), making it similar to the conditions of the NYC downwind sites, and (2) a derived $\text{PM}_{2.5}$ -suppression factor of approximately 0.02 was determined for North China Plain major cities (including Shijiazhuang City and Baoding City, Zhang *et al.*, 2022).⁷ This quite low $\text{PM}_{2.5}$ -suppression factor implies little influence of HO_x uptake by aerosol and matches the derived low HO_2 uptake coefficient during daytime (~ 0.08) from Tan *et al.* (2020)³² and Song *et al.* (2022).³³ Following the increase in $\text{PM}_{2.5}$ -suppression factor from 0.02 to 0.06 from 2014–2016 to 2017–2019 for North China Plain major cities,⁷ the γ_{HO_2} would be expected to increase to be nearer to the value ($\gamma_{\text{HO}_2} = 0.2$) used in the model.^{15–21} However, the uptake coefficients γ_{HO_2} vary significantly depending on both aerosol size and composition, and ambient conditions such as the humidity. Its variation related to the change in $\text{PM}_{2.5}$ -suppression factor under different O_3 -sensitivity regimes, as well as the underlying mechanism needed to be verified and explored through more field measurements, especially in the highly air polluted regions.

4. Conclusion

The $\text{PM}_{2.5}$ -suppression factors on the surface ozone production of Eastern US major cities were derived based on non-linear fitting of the $\text{PM}_{2.5}$ and O_3 relationship. These factors derived from urban regions showed increasing trends over 16 years of continuous ground measurements and increased from a value below 0.1 during 2004–2008 to over 0.2 during 2014–2019, and generally followed the transition of the O_3 -sensitivity regime from VOC-limited toward NO_x -limited chemistry.

A spatial similarity for the $\text{PM}_{2.5}$ -suppression factors between urban regions and their downwind locations was shown for the New York City metro area. However, the spatial distribution of the $\text{PM}_{2.5}$ -suppression factors of different urban regions showed generally larger values over cities with higher HCHO/NO_2 values – which implies a more NO_x -limited regime. The temporal and spatial variation of these $\text{PM}_{2.5}$ -suppression factors was consistent with the variation in the O_3 -sensitivity regime and provides a feasible way of identifying the O_3 -sensitivity regimes through this factor with the possibility of better representing the near ground O_3 -sensitivity chemistry than the current satellite-measured HCHO/NO_2 column concentration ratios. The variation of $\text{PM}_{2.5}$ -suppression factors raises the possibility of a changeable HO_2/NO_3 uptake coefficient on aerosol surfaces for the model simulation of O_3 pollution, and more studies/measurements are required to verify this possibility. The results from this study will provide useful guidance for further O_3 - $\text{PM}_{2.5}$ studies considering a range of $\text{PM}_{2.5}$ -

suppression factors, which in turn will constrain/evaluate model simulations for O_3 and $\text{PM}_{2.5}$ concentrations after considering the varied $\text{PM}_{2.5}$ -suppression factors and/or the varied HO_2/NO_3 uptake coefficients. These also benefit the ability of models to develop accurate O_3 and $\text{PM}_{2.5}$ concentration pollution control policies.

Conflicts of interest

There are no conflicts of interest to declare.

Acknowledgements

This work has been supported through the New York State Energy Research and Development Authority (NYSERDA) Contract 48971. We acknowledge the free use of ozone and $\text{PM}_{2.5}$ data from the US EPA, the OMI HCHO , and NO_2 data from the US NASA. Special thanks to Dr Lukas Valin from the US EPA for providing valuable comments for this study.

References

- 1 J. L. Schnell and M. J. Prather, Co-occurrence of extremes in surface ozone, particulate matter, and temperature over eastern North America, *Proc. Natl. Acad. Sci. U.S.A.*, 2017, **114**(11), 2854–2859.
- 2 S. A. Strode, J. M. Rodriguez, J. A. Logan, O. R. Cooper, J. C. Witte, L. N. Lamsal, *et al.*, Trends and variability in surface ozone over the United States, *J. Geophys. Res. Atmos.*, 2015, **120**(17), 9020–9042.
- 3 T. J. Sullivan, C. T. Driscoll, C. M. Beier, D. Burtraw, I. J. Fernandez, J. N. Galloway, *et al.*, Air pollution success stories in the United States: The value of long-term observations, *Environ. Sci. Policy*, 2018, **84**, 69–73.
- 4 EPA, *Our Nation's Air: Status and Trends through 2021*, 2021. <https://gispub.epa.gov/air/trendsreport/2021/>.
- 5 C. L. Blanchard, S. L. Shaw, E. S. Edgerton and J. J. Schwab, Emission influences on air pollutant concentrations in New York State: I. ozone, *Atmos. Environ. X*, 2019, **3**, 100033.
- 6 C. L. Blanchard, S. L. Shaw, E. S. Edgerton and J. J. Schwab, Emission influences on air pollutant concentrations in New York state: II. $\text{PM}_{2.5}$ organic and elemental carbon constituents, *Atmos. Environ. X*, 2019, **3**, 100039.
- 7 J. Zhang, J. Wang, Y. Sun, J. Li, M. Ninneman, J. Ye, *et al.*, Insights from ozone and particulate matter pollution control in New York City applied to Beijing, *NPJ Clim. Atmos. Sci.*, 2022, **5**(85), 1–7.
- 8 K. Li, D. J. Jacob, H. Liao, J. Zhu, V. Shah, L. Shen, *et al.*, A two-pollutant strategy for improving ozone and particulate air quality in China, *Nat. Geosci.*, 2019, **12**(11), 906–910.
- 9 X. Chen, Z. Jiang, Y. Shen, R. Li, Y. Fu, J. Liu, *et al.*, Chinese regulations are working—why is surface ozone over industrialized areas still high? Applying lessons from Northeast US air quality evolution, *Geophys. Res. Lett.*, 2021, **48**(14), e2021GL092816.



10 J. Zhu, L. Chen, H. Liao and R. Dang, Correlations between $PM_{2.5}$ and ozone over China and associated underlying reasons, *Atmosphere*, 2019, **10**(7), 352.

11 D. J. Jacob, Heterogeneous chemistry and tropospheric ozone, *Atmos. Environ.*, 2000, **34**(12–14), 2131–2159.

12 Y. Kanaya, P. Pochanart, Y. Liu, J. Li, H. Tanimoto, S. Kato, *et al.*, Rates and regimes of photochemical ozone production over Central East China in June 2006: a box model analysis using comprehensive measurements of ozone precursors, *Atmos. Chem. Phys.*, 2009, **9**(20), 7711–7723.

13 L. K. Whalley, K. L. Furneaux, A. Goddard, J. D. Lee, A. Mahajan, H. Oetjen, *et al.*, The chemistry of OH and HO₂ radicals in the boundary layer over the tropical Atlantic Ocean, *Atmos. Chem. Phys.*, 2010, **10**(4), 1555–1576.

14 Y. Sakamoto, Y. Sadanaga, J. Li, K. Matsuoka, M. Takemura, T. Fujii, *et al.*, Relative and absolute sensitivity analysis on ozone production in Tsukuba, a city in Japan, *Environ. Sci. Technol.*, 2019, **53**(23), 13629–13635.

15 S. Stadtler, D. Simpson, S. Schröder, D. Taraborrelli, A. Bott and M. Schultz, Ozone impacts of gas-aerosol uptake in global chemistry transport models, *Atmos. Chem. Phys.*, 2018, **18**(5), 3147–3171.

16 R. R. Dickerson, S. Kondragunta, G. Stenchikov, K. L. Civerolo, B. G. Doddridge and B. N. Holben, The impact of aerosols on solar ultraviolet radiation and photochemical smog, *Science*, 1997, **278**(5339), 827–830.

17 K. Li, D. J. Jacob, H. Liao, L. Shen, Q. Zhang and K. H. Bates, Anthropogenic drivers of 2013–2017 trends in summer surface ozone in China, *Proc. Natl. Acad. Sci. U.S.A.*, 2019, **116**(2), 422–427.

18 F. Taketani, Y. Kanaya, P. Pochanart, Y. Liu, J. Li, K. Okuzawa, *et al.*, Measurement of overall uptake coefficients for HO₂ radicals by aerosol particles sampled from ambient air at Mts. Tai and Mang (China), *Atmos. Chem. Phys.*, 2012, **12**(24), 11907–11916.

19 L. K. Xue, T. Wang, J. Gao, A. J. Ding, X. H. Zhou, D. R. Blake, *et al.*, Ground-level ozone in four Chinese cities: precursors, regional transport and heterogeneous processes, *Atmos. Chem. Phys.*, 2014, **14**(23), 13175–13188.

20 W. Wang, D. D. Parrish, X. Li, M. Shao, Y. Liu, Z. Mo, *et al.*, Exploring the drivers of the increased ozone production in Beijing in summertime during 2005–2016, *Atmos. Chem. Phys.*, 2020, **20**, 15617–15633.

21 P. D. Ivatt, M. J. Evans and A. C. Lewis, Suppression of surface ozone by an aerosol-inhibited photochemical ozone regime, *Nat. Geosci.*, 2022, **15**(7), 536–540.

22 J. Zhang, M. Ninneman, E. Joseph, M. J. Schwab, B. Shrestha and J. J. Schwab, Mobile laboratory measurements of high surface ozone levels and spatial heterogeneity during LISTOS 2018: Evidence for sea breeze influence, *J. Geophys. Res. Atmos.*, 2020, **125**(11), e2019JD031961.

23 M. H. Couillard, M. J. Schwab, J. J. Schwab, C. H. Lu, E. Joseph, B. Stutsrim, *et al.*, Vertical Profiles of Ozone Concentrations in the Lower Troposphere Downwind of New York City During LISTOS 2018–2019, *J. Geophys. Res. Atmos.*, 2021, **126**(23), e2021JD035108.

24 J. Zhang, A. Catena, B. Shrestha, J. Freedman, E. McCabe, M. J. Schwab, *et al.*, Unraveling the interaction of urban emission plumes and marine breezes involved in the formation of summertime coastal high ozone on Long Island, *Environ. Sci. Atmos.*, 2022, **2**(6), 1438–1449.

25 J. Xing, R. Mathur, J. Pleim, C. Hogrefe, C. M. Gan, D. C. Wong, *et al.*, Air pollution and climate response to aerosol direct radiative effects: a modeling study of decadal trends across the northern hemisphere, *J. Geophys. Res.*, 2015, **120**, 12221–12236.

26 J. Xing, J. Wang, R. Mathur, S. Wang, G. Sarwar and J. Pleim, *et al.*, Impacts of aerosol direct effects on tropospheric ozone through changes in atmospheric dynamics and photolysis rates, *Atmos. Chem. Phys.*, 2017, **17**, 9869–9883.

27 C. E. Buysse, A. Kaulfus, U. Nair and D. A. Jaffe, Relationships between particulate matter, ozone, and nitrogen oxides during Urban smoke events in the Western US, *Environ. Sci. Technol.*, 2019, **53**, 12519–12528.

28 K. F. Boersma, H. Eskes, A. Richter, I. D. Smedt, A. Lorente, S. Beirle, *et al.*, QA4ECV NO₂ Tropospheric and Stratospheric Column Data from OMI (2017), 2017, <https://doi.org/10.21944/qa4ecv-no2-omi-v1.1>.

29 I. De Smedt, N. Theys, H. Yu, T. Danckaert, C. Lerot, S. Compernolle, *et al.*, Algorithm theoretical baseline for formaldehyde retrievals from S5P TROPOMI and from the QA4ECV project, *Atmos. Meas. Tech.*, 2018, **11**, 2395–2426.

30 X. Jin, A. Fiore, K. F. Boersma, I. D. Smedt and L. Valin, Inferring changes in summertime surface Ozone–NO_x–VOC chemistry over US urban areas from two decades of satellite and ground-based observations, *Environ. Sci. Technol.*, 2020, **54**(11), 6518–6529.

31 C. Li, Q. Zhu, X. Jin and R. C. Cohen, Elucidating Contributions of Anthropogenic Volatile Organic Compounds and Particulate Matter to Ozone Trends over China, *Environ. Sci. Technol.*, 2022, **56**(18), 12906–12916.

32 Z. Tan, A. Hofzumahaus, K. Lu, S. S. Brown, F. Holland, L. G. Huey, *et al.*, No evidence for a significant impact of heterogeneous chemistry on radical concentrations in the North China Plain in summer 2014, *Environ. Sci. Technol.*, 2020, **54**(10), 5973–5979.

33 H. Song, K. Lu, H. Dong, Z. Tan, S. Chen, L. Zeng, *et al.*, Reduced aerosol uptake of hydroperoxy radical may increase the sensitivity of ozone production to volatile organic compounds, *Environ. Sci. Technol. Lett.*, 2021, **9**(1), 22–29.

