High-resolution large-eddy simulation to understand ozone formation and atmospheric oxidation capacity in Houston, Texas

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Abstract

Highly reactive volatile organic compounds (HRVOCs) from mobile and petrochemical sources are important players in
15 atmospheric photochemistry that contribute to the formation of ozone (O_3). In a typical elevated O_3 episode, we applied a high-resolution large eddy simulation (LES), coupled with the Weather Research and Forecasting model with chemistry (WRF-LES-Chem) to understand the mechanism of high O_3 production over the Houston area. Our modeling was constrained and evaluated using field measurements from the NASA Tracking Aerosol Convection Interactions ExpeRiment – Air Quality (TRACER-AQ) project, Texas Commission on Environmental Quality (TCEQ), and vertical column density observations from
20 Pandora spectrometers. The modeling results show enhanced performance in the LES domain, compared to the mesoscale models in simulating key chemicals. O_3 sensitivity in the Houston urban area demonstrates a nearly homogenous early morning VOC-limited regime and transits to a noontime NO_X-limited regime. As the day progresses into the afternoon, the atmospheric oxidative capacity (AOC) increases with major contribution from hydroxyl (OH) radical (90 %). High concentrations of alkenes also increased O_3 (8-10 %) contribution to AOC in the late afternoon. The OH reactivity (K_{OH}) is dominated by
25 isoprene (35.76 %), carbon monoxide (CO; 12.98 %), formaldehyde (HCHO; 12.21 %), and alkanes with C > 3 (6.29 %), thus accelerating the production of hydroperoxyl (HO_2) and peroxy (RO_2) radicals. The concentrations of short-lived VOCs such as HCHO and acetaldehyde from the oxidation of HRVOCs, increased in the afternoon, which elevated O_3 production rates under a NO_X-limited regime. The oxidation of isoprene also accelerated the production of HCHO and contributed to the production of HO_2 radicals, thus leading to a high O_3 production rate. This study suggests the possible impacts of NO_X-O_3-
30 VOC sensitivity on O_3 production rates in polluted urban areas with high emission of HRVOCs, and also provides insights on radical chemistry that drives the photochemical processes of O_3 formation. Ultimately, the study underlines the need to control

anthropogenic emissions such as alkenes and HCHO and also highlights the role of naturally emitted isoprene species in elevated urban O_3 levels.

1. Introduction

35   The Houston metropolitan area is home to approximately 7.3 million people and represents a heavily-polluted urban atmosphere due to the presence of complex industrial emissions. Like many polluted urban cities in the United States, Houston is identified as an ozone (O_3) nonattainment area by the U.S. Environmental Protection Agency (US EPA) due to its summertime exceedances of the National Ambient Air Quality Standards (NAAQS) (Shahriar et al., 2015; Botlaguduru et al., 2018), which cause severe human health concerns (Blanchard et al., 2008; Rammah et al., 2020). In 2015, the EPA lowered

40   the 8 h O_3 NAAQS to 70 ppb with projected national attainment by 2025; however, this does not appear realistic over Houston, as evident from the observed O_3 concentrations from the TRacking Aerosol Convection ExpeRiment – Air Quality (TRACER-AQ) 2021 summertime campaign. Generally, O_3 exceedances are highly dependent on the source of primary emissions, chemical reaction rates of primary and secondary oxidants, and meteorological processes (Spiridonov et al., 2019). However, the sensitivity of O_3 to precursors' regimes may also undermine or enhance its production in the atmosphere (Zhou et al., 2014;

45   Ye et al., 2016). Additionally, boundary layer dynamics that affect nonlinear chemistry-turbulence interactions are not well-resolved in mesoscale models with coarse resolutions, yet these processes are essential to understand chemical behaviors in polluted environments (Wang et al., 2023). Thus, this study is aimed to understand the complex pathways of O_3 production and changes in O_3 formation regime in the Houston urban atmosphere using a high-resolution modeling approach.

The O_3 problem in Houston, unlike other large metropolitan cities in the U.S., is a combined product of (i) highly reactive

50   volatile organic compounds (HRVOCs) and nitrogen oxides ($NO_X = NO_2 + NO$) emissions from automobile and industrial sources along the Houston Ship Channel (Souri et al., 2016b; Botlaguduru and Kommalapati, 2020), (ii) variability in the along-shore sea breeze and the planetary boundary layer (PBL) depth (Banta et al., 2011), (iii) the midday reversals of flow patterns or clustering of weak easterly and northeasterly winds (Darby, 2005; Souri et al., 2016a), (iv) potential transport of pollutants from regional and long-range sources (Morris et al., 2006; Berlin et al., 2013), and (v) complex photochemistry

55   under the hot and humid Houston environment (Perring et al., 2013). Because of the heterogeneous automobile emission sources and the presence of multiple oil and natural gas industries in Houston, anthropogenic emissions play an important role in O_3 formation. Primarily, the photochemical processes of ethylene, propylene and 1,3-butadiene from the alkene group are identified as the major anthropogenic VOC contributors to enhanced O_3 formation over Houston (Nam et al., 2006; Kim et al., 2011; Czader and Rappenglück, 2015; Sadeghi et al., 2022). Additionally, the photochemical reactivities of isoprene and

60   monoterpenes from biogenic sources are also major contributors to O_3 formation (Leong et al., 2017; Ma et al., 2022).

The self-cleaning response of the atmosphere to emissions, defined as atmospheric oxidative capacity (AOC), determines the rate of removal of primary pollutants (e.g., CO, NO_X, and VOCs) by oxidizing chemical reactions with O_3, OH, and nitrate radicals (NO_3) (Geyer et al., 2003; Chen et al., 2020). AOC is an important driving force in understanding atmospheric

chemistry, as it determines the removal rate of primary pollutants and the production rate of secondary pollutants (Mao et al., 2010). The total sum of the oxidation rates of these pollutant species is thus used to estimate AOC (Li et al., 2021). Several studies have been carried out to quantify atmospheric oxidation processes and have shown that AOC is dominated by pollutant oxidation from OH radicals, especially in heavily polluted environments (Xue et al., 2016; Li et al., 2018; Zhu et al., 2020). In such environments, the average daytime AOC may range from 6.7×10^7 molecules cm^{-3} s^{-1} to 3.2×10^8 molecules cm^{-3} s^{-1} (Liu et al., 2022). In contrast, NO_3 is insignificant during daytime photochemistry but principally contributes to nighttime AOC (Brown et al., 2011; Chen et al., 2019). However, experimental results from the TexasAQ-2000 campaign suggested that NO_3 contributes significantly to the daytime oxidation of monoterpenes and phenol in Houston (Geyer et al., 2003). Still, there is a very limited understanding of the daytime AOC over the Houston region, as compared to some other major urban polluted regions (Tan et al., 2019; Liu et al., 2022).

O_3 formation is driven by directly emitted precursors, NO_X and VOCs, yet the O_3-NO_X-VOCs photochemistry is nonlinear, and two regimes with different O_3-NO_x-VOC sensitivity have been identified (Sillman et al., 2003). Understanding the photochemical responses of O_3 to reductions in emissions of NO_X and VOCs is crucial to developing an effective O_3 control strategy. In a VOC-limited regime, for example, an incremental increase in VOCs promotes the chemical production of peroxy (RO_2) and hydroperoxy (HO_2) radicals, which increases NO_X cycling and ultimately increases ambient O_3 (Ye et al., 2016). By contrast, in a NO_X-limited regime, increased emissions of NO_X enhance the photolysis of NO_2, thus producing free reactive oxygen atoms, and ultimately reacting with O_2 to produce O_3, resulting in increased O_3 concentrations (Jin and Holloway, 2015). Here, O_3 is produced when the HO_2 and RO_2 radicals convert NO to NO_2 without the consumption of O_3. Previous air quality campaigns over Houston have focused extensively on understanding O_3 photochemistry to NO_X- and VOC-limited regimes. For instance, the TexAQS-2006 campaign (conducted from September to October 2006) showed that the rapid production of O_3 was associated with a VOC-limited regime and a slow production was characterized by a NO_X-limited regime (Zhou et al., 2014). Evidence from the SHARP (April – May 2009) and DISCOVER-AQ (September – October 2013) campaigns also show that O_3 production was more VOC-limited in the morning hours and NOx-limited in the afternoon hours (Ren et al., 2013; Mazzuca et al., 2016). Despite evidence from the TexAQS-2006 campaign that reducing NO_X emissions from power plants led to reduced O_3 concentrations (Cowling et al., 2007; Parrish et al., 2009), high O_3 concentrations up to 120 ppb were observed during the TRACER-AQ campaign. Nevertheless, this sensitivity regime is poorly understood over the Houston region and may be exacerbated by the complex interactions in the region. The exact dependence of O_3 on NO_X- and VOC-limited regimes varies greatly on assumptions and different conditions, hence, threshold criteria for transition regimes are likely to change depending on changes in emission rates and patterns (Zhang et al., 2023), which may create a shift in O_3 regimes. Thus, we need an accurate modeling interpretation of chemical and physical processes to more accurately determine O_3-NO_X-VOCs sensitivity.

Since O_3-NO_X-VOCs sensitivity is mostly determined from photochemical air quality models (Sillman et al., 2003; Song et al., 2010; Li et al., 2011; Goldberg et al., 2016; Du et al., 2022), accurate modeling representations of chemical and physical processes are crucial to better determine O_3 formation regime, thus more effectively supporting ozone mitigation strategies in

Houston. However, previous model-based studies are subject to large uncertainties from simulated O_3 concentration and its precursors. The diurnal evolution of the PBL can significantly influence the mixing, transport, and chemistry of pollutants (Miao et al., 2019; Xiang et al., 2019), and thus must be accurately resolved in modeling. Based on previous studies, turbulence results in reductions of chemical reaction rates in the boundary layer, as a result of negatively correlated concentrations and vertical velocities (Li et al., 2016). Stockwell (1995) showed turbulence causes a non-uniform mixing of gas-phase trace species, thus limiting the accuracy of chemical models. This is particularly true if the model resolution is too coarse to resolve heterogeneous urban emissions and wind flows (Chen et al., 2009; Uebel and Bott, 2018). Also, these chemical models cannot adequately resolve the concentration and transport of O_3 pollutants in convective, unstable atmospheres due to a misrepresentation of boundary layer evolution (Li et al., 2016; Cuchiara et al., 2014). In a recent study, Wang et al. (2023) showed an improved modeling-observation agreement of the spatial and vertical distributions of key chemical species (e.g., O_3, NO and NO_2) from large eddy simulation (LES) compared with mesoscale simulations, indicating LES is an effective tool to resolve turbulence effects on chemical reactions. WRF-LES-Chem is advantageous due to its potential to improve the physical and chemical processes of pollutants at different scales.

In this work, we perform high-resolution simulations using WRF-LES-Chem over an O_3 exceedance period in September 2021 during the TRACER-AQ experiment, aiming to provide a comprehensive understanding of photochemical and physical processes involved in O_3 formation over Houston. Recent modeling studies have investigated ozone episodes during the TRACER-AQ campaign (Soleimanian et al., 2023; Li et al., 2023; Liu et al., 2023). This work expands upon those efforts by applying higher resolution LES simulations on the particular case day of September 8, 2021. Model results are comprehensively validated utilizing long-term ground-based observations from the TCEQ monitoring stations, measurements from the TRACER-AQ campaign, and observations from ground-based Pandora spectrometers to ensure modeling accuracy. This study also assesses spatial and temporal variability of the O_3 formation regime and investigates AOC, which can provide useful information to environmental managers to better control O_3 pollution over Houston.

2. Methodology

2.1 Model description and set-up

2.1.1 Dynamic settings

We use the WRF-Chem model (Grell et al., 2005; Fast et al., 2006) version 4.4 with the integrated rate reactions capability that allows for the investigation of specific chemical pathways. Since LES is better suited to represent the PBL evolution and the transport of large eddies, we use a WRF-LES-Chem approach to simulate the concentration and transport of O_3 pollutants in convective, unstable atmospheres evolution (Li et al., 2016; Wang et al., 2023). Our modeling consists of four domains (D01 – D04) with horizontal grid resolutions of 8.1 km, 2.7 km, 900 m, and 300 m from outermost to innermost domains on a Lambert conformal projection (Fig. 1). The first three domains (D01 – D03) represent mesoscale setups while the innermost

domain (D04) is the LES domain. While the ideal LES resolution may range from 100 m and below, we select a 300 m
130   resolution due to the computational feasibility of our study that aims to better account for both boundary layer dynamics and
detailed atmospheric chemistry, as used in other literature (e.g., Wang et al., 2023). At this resolution, we aim to reach a
compromise between model computational efficiency and model resolution, since LES at finer scale resolutions are otherwise
too computationally expensive, especially given the size of our innermost domain that aims to focus on most of the Houston
area. The LES domain extends over the Houston urban area, including all TRACER-AQ measurement stations and Houston
135   Ship Channel, and includes 427 x 415 grid cells horizontally. A two-way nesting approach (the inner domain provides feedback
to the outer domain) is set up between the domains. We perform two simulations where a two-way nesting between domains
D01 and D02 is used to drive another two-way nested simulation between domains D03 and D04. The simulations are
performed at a time step integration of 24 seconds and 1.8 seconds, respectively. The vertical layers comprise of 61 levels
across all domains with the lowest layer within ~20 m near the surface, and include 25 layers located within the PBL to resolve
140   boundary layer processes (~1.5 km). The vertical grid spacing gradually increases up to the top of the model at 50 hPa.

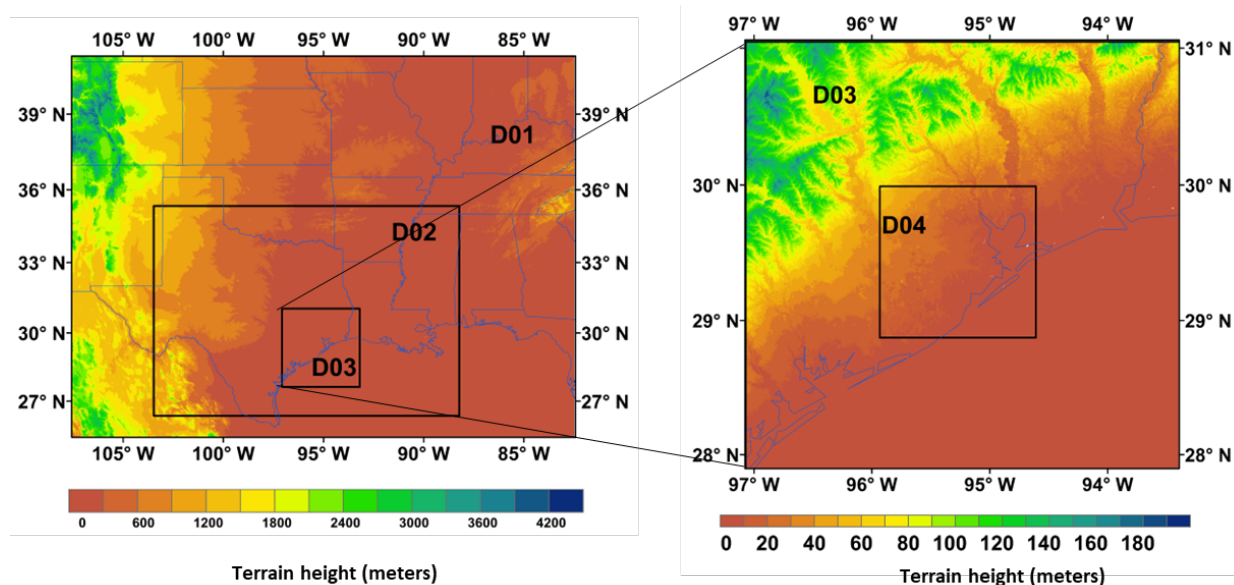



Figure 1: The four-layer domain setup with the background contour showing average topography.

To better examine chemical processes in a complex, dynamic urban atmosphere like Houston, a more accurate meteorological
145   product is needed to drive the simulations. We therefore examine model sensitivity to meteorological initial and boundary (IC
& BC) conditions from the High-Resolution Rapid Refresh (HRRR; Dowell et al., 2022) and the Final Operational Global
Analysis (NCEP-FNL; NCEP, 2000) data, with these two meteorological products widely used in previous studies (Jiménez
et al., 2022; Pinto et al., 2021; Gholami et al., 2021) (Fig. S1). NCEP-FNL has a spatial resolution of 1° x 1° and is provided
at a 6 h interval. HRRR is provided at a horizontal grid spacing of 3 km and available every 1 h up to 48 h forecasts. We
150   perform a two-way nested simulation using domains D01 and D02 without chemistry to test the performance of both FNL and

HRRR as IC & BC. The simulation extends from September 6, 2021 (00:00 UTC; September 5, 19:00 CDT) to September 9, 2021 (00:00 UTC, September 8, 19:00 CDT). Our model results, driven by the two datasets show similar spatial distributions of the surface heat budget (Fig. S1). We also show a wind rose diagram which gives a succinct view of how wind speed and direction are typically distributed at a particular location. Here, a prevailing north-westerly wind based on the TCEQ in-situ

155   observations is observed. The model results driven by the NCEP-FNL more closely matches the TCEQ observed north-westerly wind than HRRR.. Wind fields are key parameters to determine the transport and mixing of pollutants. Hence, although HRRR has a much higher spatial resolution, we select the NCEP-FNL meteorological product to provide a more realistic dynamic field in our simulations.

Key physical parameterizations in our modeling include: the Rapid Radiative Transfer Model for GCMs (RRTMG) scheme

160   for longwave and shortwave radiation (Iacono et al., 2008), Morrison double moment scheme for cloud microphysics (Morrison et al., 2009), the Noah land surface model (Chen and Dudhia, 2001), and the revised MM5 Monin-Obukhov for the surface layer scheme (Jiménez et al., 2012). The Grell 3D scheme (Grell and Dévényi, 2002) is used for cumulus parameterization to resolve sub-grid cloud processes for coarser resolution (D01 and D02), while it is turned off for D03 and the LES (D04) domain. We use the Yonsei University PBL scheme (YSU; Hong et al., 2006) to parameterize the sub-grid

165   scale turbulent fluxes within the PBL for the mesoscale domains. YSU represents a non-local and first-order closure scheme to estimate turbulent diffusion, where vertical diffusion is expressed as a function of a non-local gradient adjustment term. This PBL option is turned off for the LES domain to resolve large eddies directly, however, finer-scale eddies are resolved by the three-dimensional 1.5 order turbulent kinetic energy (TKE) closure (Deardorff, 1970).

Simulations for the mesoscale domains (D01 – D02) cover the period from September 6, 2021 (00:00 UTC) to September 9,

170   2021 (00:00 UTC), which corresponds to Houston's local time from September 5, 2021 (19:00 Central Daylight Time, CDT) to September 8, 2021 (19:00 CDT), with the first 20 hours as the spin-up time. The output was used to drive parallel simulations for D03 and the LES domain and cover the period from 11:00 to 21:00 UTC (06:00 – 16:00 CDT) on September 8, 2021.

2.1.2 Chemistry settings

Our study adopts the recent MOZART (Model of Ozone and Related Chemical Tracers) tropospheric chemistry scheme (T1;

175   Emmons et al., 2020) to represent the gas-phase chemistry. MOZART-T1 is a more comprehensive chemistry scheme compared with previous versions. It includes 151 gas-phase chemical species, 65 photolysis reactions and 287 kinetic reactions. Compared to previous versions of the MOZART scheme, specific modifications are made to the oxidation of isoprene and terpenes in MOZART-T1, as well as the speciation of aromatic and organic nitrate to improve the representation of O_3 and secondary organic aerosol (SOA) precursors (Emmons et al., 2020). The Tropospheric Ultraviolet-Visible (TUV; Tie et al.,

180   2003) transfer scheme is used for the estimation of photolysis rates. We also use output from the Community Earth System Model - Whole Atmosphere Community Climate Model (CESM2-WACCM; Liang et al., 2022; Gettelman et al., 2019) as the IC and BC of chemical species in our simulations. Initial conditions and boundary conditions for each nested domain are generated from the parent domains, which are produced by the 8.1 km simulation. The Model of Emissions of Gases and

Aerosols from Nature (MEGAN; Guenther et al., 2006) and the 2017 US EPA National Emissions Inventory (NEI; Reff et al.,

185   2020) are used as biogenic and anthropogenic emissions, respectively, while the recent release of the Fire Inventory from NCAR (FINNv2.5; Wiedinmyer and Emmons, 2022) is used to provide fire emissions and biomass burning. NEI emissions data consists of point and non-point sources at a 12-km resolution and are downscaled to respective model resolutions for inner domains through WRF nesting. Emissions from MEGAN and FINN are available at a base resolution of 1 km, and all emission inventories are mapped to the same resolution for each of the domains. Our simulation results with the 2017 NEI show

190   overestimated NO_X concentrations, as found in previous urban studies (Choi and Souri, 2015; Souri et al., 2016a; Silvern et al., 2019; Herrmann and Gutheil, 2022). To more accurately represent emissions during the high ozone episode in Houston, we compare simulated concentrations of NO and NO_2 with their observations from TRACER-AQ and develop scaling factors of 0.3 and 0.2, respectively, to adjust NEI emissions of NO and NO_2. The total NO_2 emission over Houston accounts for 11.32 % of the total emitted NO_X based on the NEI over the study period. On average, the diurnal pattern of the rescaled NO_X

195   emission is characterized by high early morning emissions at ~0.65 mol km^{-2} h^{-1} and then peaks in the late afternoon at ~0.74 mol km^{-2} h^{-1}. The results we show in this study are based on simulations with the scaled NO_X emissions. While existing studies show large uncertainties in previous versions of FINN which may result from the coarse resolution of the active fire product (Faulstich et al., 2022; Liu et al., 2020), the improved spatial resolution of the VIRS product in the FINNv2.5 provides more sensitive detection of fires at local and regional scales to minimize the estimated emission uncertainties (Wiedinmyer et al.,

200   2023).

2.2 Observational data

2.2.1 TRACER-AQ airborne observations

The NASA TRACER-AQ experiment aimed to provide a robust understanding of how pollutants vary spatially, vertically and temporally over the Houston region. A wide range of instruments were operated during TRACER-AQ, which generated large

205   measurement datasets from airborne, ship-based, mobile and stationary ground platforms (Knapp and Boyer, 2022; Griggs et al., 2024). Continuous atmospheric profile measurements from Sondes and Lidar provide unique and valuable data to understand the variability of pollutant profile and column concentrations (Gronoff et al., 2019). We use measured vertical distributions of temperature, relative humidity, wind speed, potential temperature, and ozone from field ozonesondes, which were launched from Galveston Bay and La Porte stations (see Fig. 2) on 8 September 2021 at 12.47 CDT and 13:07 CDT,

210   respectively. Since the balloon drifts during its ascent in response to background winds, we use the nearest model grids along the sounding trajectory for accurate comparison.

2.2.2 TCEQ and Pandora ground station measurements

In-situ measurements from 18 Texas Commission on Environmental Quality (TCEQ) stations are used to evaluate our simulated surface meteorological fields (temperature, wind speed, vertical velocity and water vapor) and pollutant (O_3, NO_2)

215 concentrations. Data from 67 stations in Houston are available in the TCEQ Texas Air Monitoring Information System (TAMIS) database at every hour. Data coverage from TCEQ stations is widely used in previous studies for VOCs, hydrocarbon and meteorology monitoring (Schade and Roest, 2018; Vizuete et al., 2022).

We also use level 2 (L2) data products retrieved from the two Pandora spectrometers in La Porte (29.67° N; 95.06° W) and the University of Houston (UH; 29.72° N; 95.34° W), including column concentrations of O_3 as well as surface concentrations of

220 HCHO and NO_2 (Fig. 2). Pandora spectrometers directly use the differential optical absorption spectroscopy (DOAS) method to obtain slant column density from measured solar irradiances, and translated to estimate the tropospheric column of pollutant variables (Platt and Stutz, 2008; Herman et al., 2009, 2018; Thompson et al., 2019). Since light detected by Pandora spectrometer systems can transverse through different paths depending on the observation geometry, it allows the detection of trace gas absorption at various altitudes. Previous studies found good agreements between Pandora measurements and

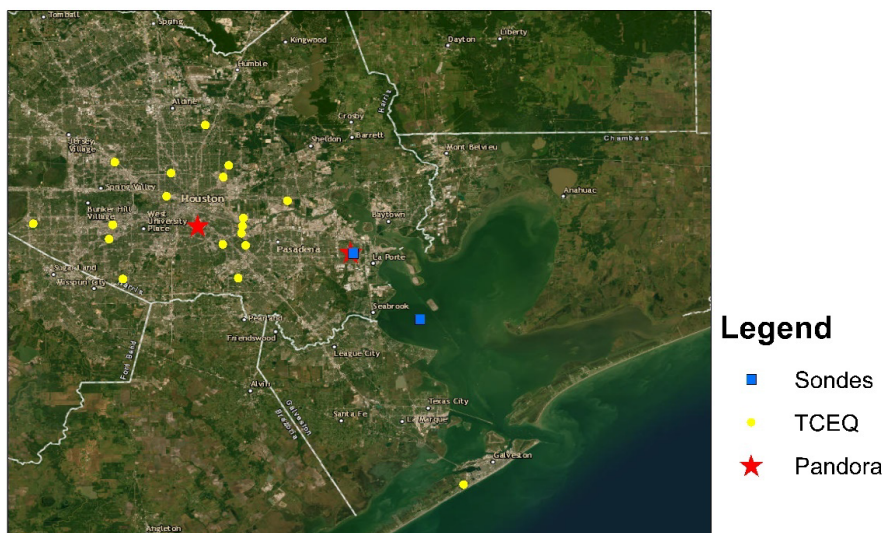
225 calibrated in-situ measurements as well as between Pandora and satellite retrieved data products (Baek et al., 2017; Martins et al., 2016; Thompson et al., 2019; Spinei et al., 2018). The sky-scanning properties of the Pandora spectrometer are critically sensitive to trace gases absorptions in the lower atmosphere and can be used to measure the tropospheric column of NO_2 and HCHO (~4 km top of atmosphere; TOA), while measurements at large zenith direction provide information on O_3 stratospheric gas absorption properties.

230

Figure 2: Locations of the TCEQ monitoring stations, TRACER-AQ sondes and Pandora spectrometers in the Houston region (Credits for the background image: Esri, HERE, Garmin, © OpenStreetMap contributors, and the GIS user community).

2.3 O_3 production rate and sensitivity regime

In daytime photochemical reactivity, the O_3 production rate (PO_3) is primarily dependent on NO_2 production from the reactivity

235 of NO with RO_2 and HO_2 radicals. In contrast, the loss of O_3 depends on the photolysis of O_3, the formation of nitrate acid from OH and NO_2, and the ozonolysis of alkenes. Overall, we estimate net PO_3 using Eq. 1 as defined by Mazzuca et al. (2016).

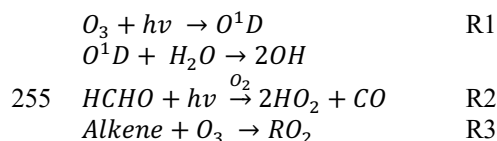
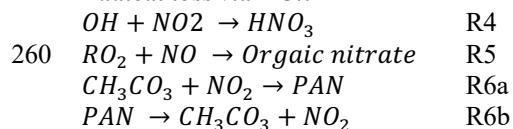
$$\textbf{Net } P(\textbf{O}_3) = k_{HO_2+NO}[HO_2][NO] + \sum k_{RO_{2i}+NO}[RO_{2i}][NO]$$
$$- k_{OH+NO_2}[OH][NO_2][M] - P(RONO_2) - k_{HO_2+O_3}[HO_2][O_3]$$
$$- k_{OH+O_3}[OH][O_3] - k_{O(1D)+H_2O}[O(1D)][H_2O] - L(O_3 + alkenes) \qquad (1)$$

Here, the k terms represent the reaction rate coefficients; RO_{2i} is the individual organic peroxy radicals. Several indicators are used to determine the response of O_3 production to NO_X and VOC emissions, thus to determine NO_X-limited and VOC-limited regimes for O_3 formation. More predominantly used methods include the ratio of the rate of radical loss via NO_X reaction (L_N) to the total primary radical production (Q) (Mazzuca et al., 2016; Zhou et al., 2014; Yu et al., 2020; Zhang et al., 2023) and HCHO to NO_2 ratio (FNR; Duncan et al., 2010; Jin and Holloway, 2015; Schroeder et al., 2017; Goldberg et al., 2022; Chen et al., 2023). As proposed by Kleinman (2005), L_N/Q can be calculated by dividing the rate of radical loss via NO_X reaction (L_N = R4 + R5 + R6a – R6b) by the total of primary radical productions (Q = 2*R1 + 2*R2 + R3). The photodissociation of O_3 and HCHO represent the primary pathways for the production of HO_X (HO_2 + OH) radicals (R1-R3), while radical termination through NO_X is summarized in R4-R6. A reactivity-weighted threshold of 0.5 indicates the transition between a VOC-limited regime and a NOx-limited regime, with L_N/Q < 0.5 representing a NO_X-limited regime, while L_N/Q > 0.5 indicating a VOC-limited regime (Kleinman et al., 2001; Kleinman, 2005).

Radical Initiation:

$O_3 + hv \rightarrow O^1D$	R1
$O^1D + H_2O \rightarrow 2OH$	
$HCHO + hv \xrightarrow{O_2} 2HO_2 + CO$	R2
$Alkene + O_3 \rightarrow RO_2$	R3

Radical loss via NOx:

$OH + NO2 \rightarrow HNO_3$	R4
$RO_2 + NO \rightarrow Orgaic\ nitrate$	R5
$CH_3CO_3 + NO_2 \rightarrow PAN$	R6a
$PAN \rightarrow CH_3CO_3 + NO_2$	R6b

Research on FNR as an indicator for PO_3 uses HCHO as an estimate of the oxidation of short-lived VOC emissions (Duncan et al., 2010). In their study, HCHO and NO_2 are retrieved from satellite instruments; hence this ratio is introduced as a non-local or regional proxy for the chemical regime (Duncan et al., 2010). Previous studies have adopted varying FNR threshold values to indicate NO_X or VOC-limited regimes in regions with distinct emission characteristics (Wang et al., 2021; Chen et al., 2023). In a heavily polluted environment from HRVOC emissions, the assumption of HCHO as a proxy for short-lived VOCs may not hold, thus affecting the FNR threshold (Schroeder et al., 2017; Souri et al., 2020). For example, when a large influx of HRVOC emissions is introduced in a heavily polluted atmosphere, radical species and NO_X partitioning reach new steady state concentrations faster while HCHO may be slower to respond, which temporarily reduces the ability of HCHO to act as a proxy until it reaches a new steady state (Schroeder et al., 2017). Tonnesen and Dennis (2000) used a threshold of 1.0 to estimate the transition of the two regimes over the New York area. Duncan et al. (2010) assumed a VOC-limited regime when FNR < 1.0 and a NO_X-limited when FNR > 2.0, which are also adopted in other studies (Tang et al., 2012; Jin and

275   Holloway, 2015; Chen et al., 2023). There is still a high uncertainty relating to the transition threshold between the two regimes
       for FNR, and peak values of FNR may reach up to ~6 in southeastern US cities (Li et al., 2022). Hence, we apply both the
       FNR and L_N/Q methods to examine O_3 formation regimes and use the L_N/Q to investigate the FNR transition threshold in the
       Houston area. Calculations are conducted using model results.

2.4. Model evaluation metrics

280   To quantitatively evaluate the reliability of the model, we calculate the mean absolute error (MAE) and root mean square error
       (RMSE) for the comparison of modeled results with TCEQ observations. All comparisons are made on time-paired data for
       air temperature, wind speed, O_3 and NO_2 across selected stations. MAE and RMSE are calculated using the following
       equations:

$$MAE = \frac{\sum_{i=1}^{n} |y_i - x_i|}{n} \qquad\qquad (2)$$

285
$$RMSE = \sqrt{\frac{\sum_{1=1}^{n} (y_i - x_i)^2}{n}} \qquad\qquad (3)$$

       where y_i and x_i are time-paired model and TCEQ observation values.

3. Results

3.1 Spatial distributions

       Since local production of O_3 is also influenced by temperature and water vapor (Kleinman, 2005; Duncan et al., 2010), it is
290   important to capture the accurate representation of the meteorology. Thus, we first investigate horizontal distributions of
       simulated key meteorological fields, including temperature, horizontal and vertical wind speed, and water vapor mixing ratio
       near the surface across in model domains D02 and D04 on September 8, 2021. As we apply an online two-way coupling that
       allows feedback between the intermediate domain D03 and the LES domain D04, results from the mesoscale D03 are modified
       by feedback from the LES thus these two domains show similar spatial characteristics. D03 results are therefore not shown
295   nor further discussed. Two time slices, including 09:00 CDT (Fig. S2) and 15:00 CDT (Fig. 3), are selected to represent distinct
       convective hours. Here, meteorological conditions reveal that warm, moist air prevails in the broad Houston region during the
       morning, while hot, drier air with prevailing northerly winds dominate the inland Houston during the afternoon. From 09:00
       CDT to 15:00 CDT, increasing solar radiation drives a more dynamic boundary layer, with temperature, wind speed and water
       vapor mixing ratio all demonstrating more heterogeneities in the afternoon. We therefore show maps of the meteorological
300   fields at 15:00 CDT here but keep the 09:00 CDT fields in the Supplementary Materials as a reference. At both times, model
       generally overestimates (underestimates) temperature (horizontal wind speed), although the results from the LES domain show
       better agreements with observations from the TCEQ stations than the results from the mesoscale D02 simulation, especially at
       15:00 CDT under a more developed PBL condition. The post-sunrise (9:00 CDT) surface temperatures are relatively low and

exhibit homogeneous distributions both inland and over the sea. As the day progresses into the afternoon, surface temperature
305   rises over land due to increased solar radiation, with higher temperatures dominating the Central Houston region due to the
impact of the urban heat island (Marsha et al., 2018). Vertical wind speed also indicates enhanced daytime convective updrafts
at the surface in the LES domain compared to the mesoscale domains. In addition, both the mesoscale and LES results show
warmer biases compared to observations, though the LES domain D04 has lower RMSE of 2.39 °C and 2.74 °C for temperature
at the Deer Park and Clinton stations compared to 2.57 °C and 3.53 °C from the mesoscale domain D02 (Table S1). Similarly,
310   compared with the observed average wind speed of 4.67 m s^{-1}, MAE values show a lower model bias for the LES at 2.42 m s^{-1} than 2.91 m s^{-1} for the mesoscale domain at the Clinton station. While turbulence may be better resolved at finer LES
resolutions (Wang et al., 2023), our LES results with a horizontal spacing of 300 m demonstrate reasonable capacity to resolve
PBL dynamics as in Lebo and Morrison (2015) and Grant et al. (2022).

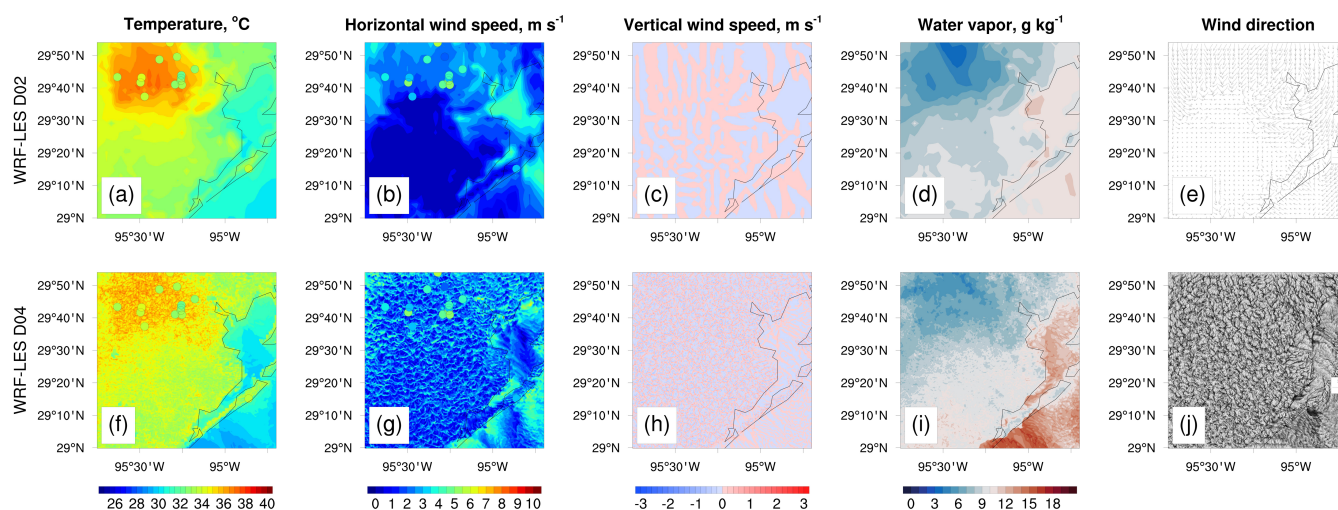



Figure 3: Contour maps are horizontal distributions of simulated temperature (first column), horizontal wind speed (second column), vertical wind speed (third column), water vapor (fourth column), and wind direction (fifth column) near the surface at 15:00 CDT for D02 (first row) and D04 (second row) simulations. Circled dots represent measurements from TCEQ stations.

320   The near-surface concentrations of modeled O_3, NO_2, HCHO and isoprene during the afternoon (15:00 CDT) are shown in
Fig. 4. Overall, the model shows slight overestimation of simulated O_3, NO_2, and HCHO. However, the model captures well
the diurnal variations of these chemical species when compared with observations. In contrast to the LES domain D04, the
mesoscale domain D02 shows minimal spatial variation due to its coarse resolution. Like O_3, the LES domain D04 shows
improved model resolving capacity for chemical species as they show the best agreement with observations in the morning
325   and afternoon hours. Peak O_3 concentrations occur around 15:00 CDT. While the LES domain (D04) slightly overestimates

observations of O_3 over Galveston Island by up to 10 ppb, our model results in D04 still demonstrate good agreement with observations from the majority of the TCEQ stations, particularly in the Central Houston region with a dense observation network. High NO_2 concentrations are both observed and simulated in the morning hours (Fig. S3), with a maximum surface concentration of 42 ppbv identified in Central Houston, resulting from road traffic emissions during rush hour (Souri et al.,

330 2016b). In the late afternoon, the photolysis of NO_2 is enhanced, thus resulting in reduced NO_2 concentrations. Overall, the model captures NO_2 dominant zones within the Central Houston and the Houston Ship Channel areas, as their presence in the atmosphere is largely dependent on anthropogenic emissions. The statistical comparison of both the mesoscale and LES domains with TCEQ observations (Table S1) show an overestimation of O_3 mixing ratio, though LES simulates a lower bias (RMSE = 5.49 ppbv, MAE = 4.32 ppbv) compared to the mesoscale domain D02 (RMSE = 16.01 ppbv, MAE = 15.05 ppbv)

335 at the Deer Park station. Like O_3, the LES results exhibit lower biases at both stations for NO_2 when compared with the TCEQ observations. Although the LES performs better, the results in this domain still show biases, which are likely induced by the uncertainties and coarser resolutions of the emissions inventories used in this study (Herrmann and Gutheil, 2022). Also, the temporal variability of the WACCM boundary conditions used in our simulations may introduce transport errors as identified in other regional modeling studies (Tang et al., 2023). In contrast to NO_2, HCHO follows an opposite diurnal pattern with

340 lower concentration in the morning hours and higher concentration in the afternoon due to photochemical reactions of primary and oxygenated VOCs (Zhang et al., 2022). The temporal distributions of NO_2 and HCHO are different as afternoon HCHO is a product of emitted HCHO and secondary HCHO produced from other precursors, thus higher in the afternoon when atmospheric oxidation is high (Nowlan et al., 2018). Isoprene, a major precursor to daytime O_3 production, shows higher concentrations in the late afternoon (Fig. 4), as the biogenic emissions are dependent on temperature and light (Rinne et al.,

345 2002; Guenther et al., 2006; Zeng et al., 2023). In addition, surface and column concentrations of NO_2, HCHO and O_3 from Pandora spectrometers are used to evaluate the model's performance as shown in the Supplementary Materials (Fig. S4). We also see the LES domain D04 show improved column concentration of HCHO, NO_2 and O_3, which is consistent with the diurnal variations of the pollutants from the Pandora spectrometers.

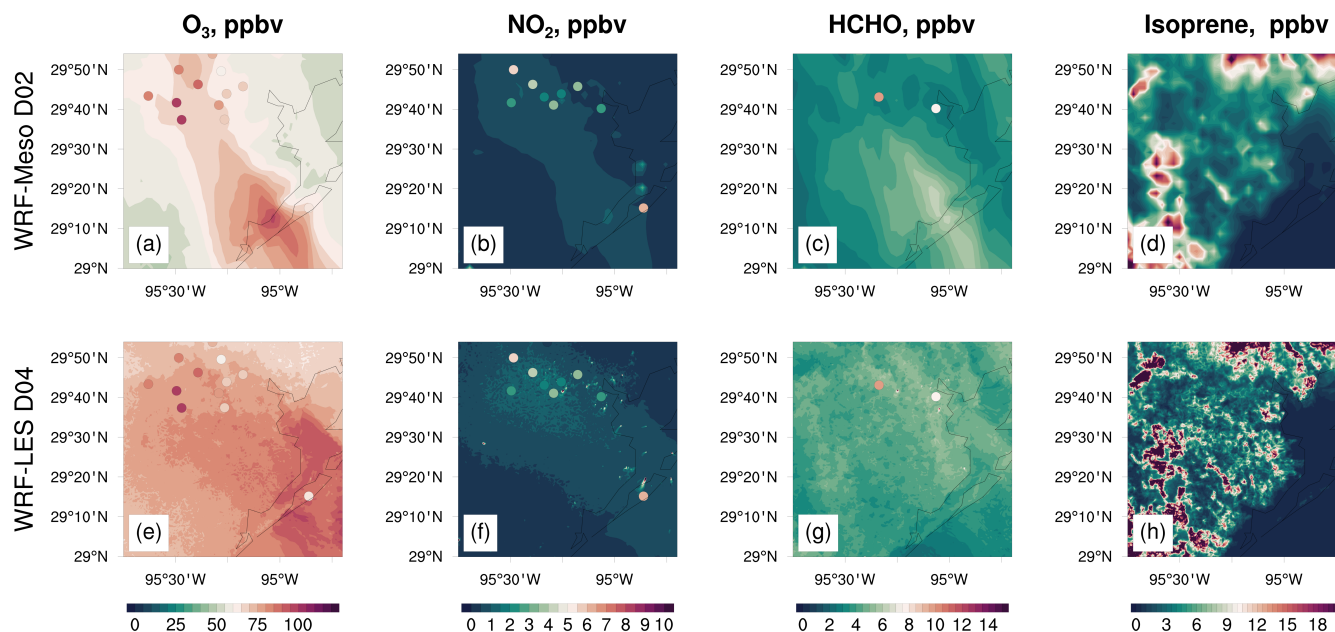



Figure 4: Contour maps are horizontal distributions of simulated O₃ (first column), NO₂ (second column), HCHO (third column) and isoprene (fourth column) near the surface at 15:00 CDT for D02 (first row) and D04 (second row) simulations. Dots represent TCEQ observations of O₃ and NO₂ and Pandora observations of NO₂ and HCHO. Isoprene and other key VOCs are validated separately.

To understand O_3 formation, an accurate model interpretation of its VOC precursors is needed. We, therefore, validate our modeled VOCs by comparing key VOCs that are measured by the Proton Transfer Reaction – Mass Spectrometry (PTR-MS) at San Jacinto Battleground State Historical site (29.753 °N, 95.091 °W), a TRACER-AQ site (Fig. 5). The comparison results indicate the model adequately reproduces observations, with results from the LES domain D04 better matching observed VOCs, especially for acetaldehyde, isoprene, HCHO, methyl ethyl ketone (MEK), toluene and xylenes. Although acetone is largely underestimated in the mesoscale D02 simulations, the LES results better reproduce observations. We also acknowledge that the difference in measurement time (30 s) and model average time (1 h), as well as representing a station with one model grid cell may account for further disparities between the model and station measurement. This intercomparison considers daytime VOC concentrations, while the San Jacinto site has previously reported anomalous, high isoprene and MEK concentrations overnight that may likely arise from anthropogenic activity instead of biogenic emission and oxidation (Shrestha et al., 2023). The daytime intercomparison here likely reflects the local biogenic contributions to isoprene.

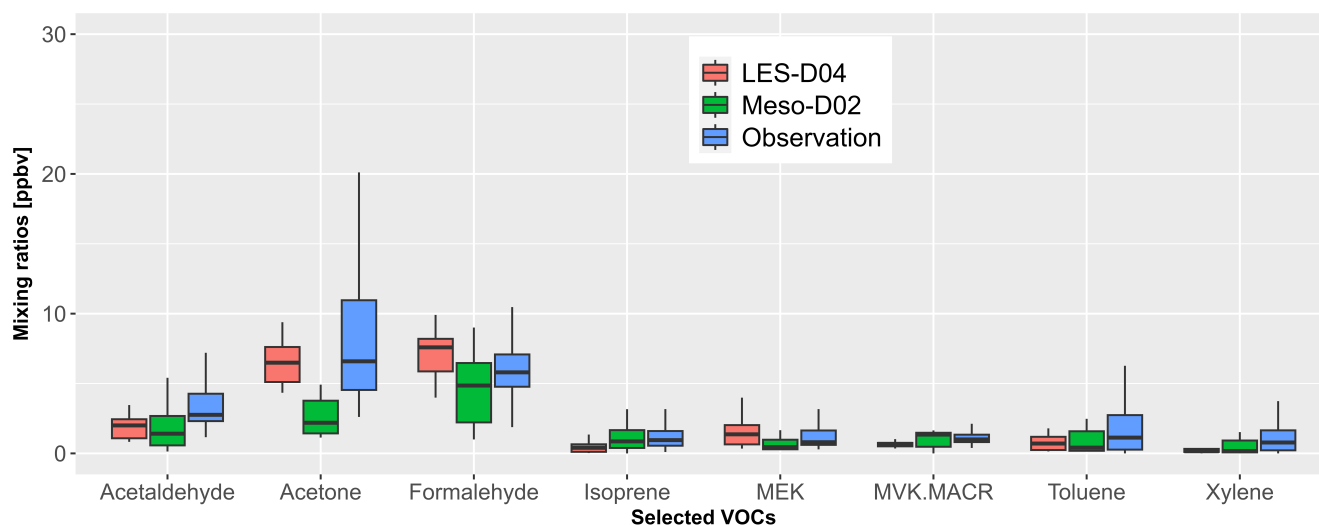



Figure 5: Comparison of daily concentration averages (over 8:00 – 16:00 CDT) of selected VOCs from our simulations in D02 and D04 and measurements at the San Jacinto Battleground State Historical site.

3.2 Vertical Profiles

We next examine simulated vertical profiles of key meteorological and chemical fields to study the role of boundary layer mixing in modifying surface levels of air pollutants. Simulated vertical profiles of O_3 and key meteorological variables are compared with sonde measurements from TRACER-AQ over two launch sites, La Porte and Galveston Bay, as shown in Fig. 6. The trajectories of sondes can drift gradually from the launch sites to higher altitudes; therefore, simulation results aloft are selected according to sonde positions to minimize comparison bias of the vertical profiles. For potential temperature, the LES domain D04 provides a better match with observations than the mesoscale domain D02. Additionally, the LES domain D04 simulates a PBL height (PBLH) of approximately 1.3 km at the Galveston Bay launch site, comparable with sonde measurements of about 1.1 km (Fig. 6d). At the Galveston Bay site, the LES overestimates wind speed by ~3 m s^{-1} near the surface and ~2 m s^{-1} at 1.3 km altitude when compared to sonde observations. Similarly, from the surface to ~0.5 km altitude, wind speed is also overestimated by the model by a maximum value of ~1.5 m s^{-1}, a 30 % difference based on the observations at the La Porte site. For relative humidity, the model simulates contrasting patterns below and above ~1.3 km altitude and underestimates relative humidity by 30 % below this transition altitude in the Galveston Bay region. Compared with the Galveston site, modeled vertical profiles of relative humidity depict smaller biases compared to observations at the La Porte site, with less bias from the LES domain than the mesoscale domain. The larger model-observation discrepancy at the Galveston station may be related to the proximity of the station to the open water surface of the Gulf of Mexico, as the rapid changes in air dynamics over the land-sea interface typically could not be accurately resolved through modeling (Ward et al., 2020). In addition, land surface parameterization in models in accurately modifying surface sensible and latent heat fluxes may vary over land and near water surfaces, thus further affecting simulated moisture metrics (Kantha Rao and Rakesh, 2019). Similarly, the NCEP-FNL meteorology at a 1° x 1° coarse resolution used to drive our modeling may also introduce simulated

390   moisture bias at upper altitudes, which could primarily be due to inaccurate surface soil moisture that cannot be well represented at this coarse resolution (Ochsner et al., 2013).

The vertical structures of temperature show reasonable performance compared with observations at both sites, with the LES domain slightly performing better (+0.5 °C bias) than the mesoscale domain (+0.7 °C bias). Simulated O_3 profiles also show biases ~~comparing~~ with observed profiles. The LES domain D04 overestimates O_3 mixing ratio while the mesoscale domain

395   D02 underestimates it over both sites. Discrepancies between model and observations may be attributed to the time lag between sonde measurements and simulation outputs. Minute-to-minute changes in wind speed and directions affect the distribution and dispersion of air masses (Li et al., 2019), thus the time differences may affect the model-observation comparison. In addition, the accuracy of reanalysis meteorological datasets, which are used to drive model simulations, in representing vertical profiles of key meteorological factors and their diurnal evolution may also contribute to the comparison discrepancy. We

400   further investigate the temporal evolution of the vertical cross-section at the University of Houston and La Porte in the Supplementary Materials (Fig. S5) and show the LES domain captures the peak low-level O_3 concentration (96 ppb) between 18:00 UTC (13:00 CDT) and 21:00 UTC (16:00 CDT) when compared to ground-based Lidar measurements from TRACER-AQ (90 - 125 ppb).

In addition, we compare the evolution of O_3 and NO_2 vertical profiles from the mesoscale domain D02 and LES domain D04

405   to show cross-domain differences in resolving variations of pollutants vertically (Fig. 7). As the day progresses, simulated O_3 mixing ratio near the surface increases, reaching up to 80 ppb in the LES domain in the late afternoon. In the early morning, O_3 increases while NO_2 decreases with height, indicating high O_3 accumulation aloft from the nocturnal stable layer and high NO_X emissions from surface anthropogenic activities during morning rush hours. A more well-mixed atmosphere during the daytime then reflects higher dispersion of pollutants from the surface to aloft. The inter-domain comparison shows contrasting

410   results in O_3 profile both below and above the PBL. O_3 in the LES domain D04 decreases with height within the PBL and gradually increases above 2 km, which is generally in line with lidar observations. More inter-domain comparison of diurnal evolution of vertical profiles of key meteorological and chemical factors are shown in Fig. S6. Following the comparative evaluation of the meteorology and chemistry across domains and between modeling and observations, the LES domain D04 provides a more realistic atmospheric condition.

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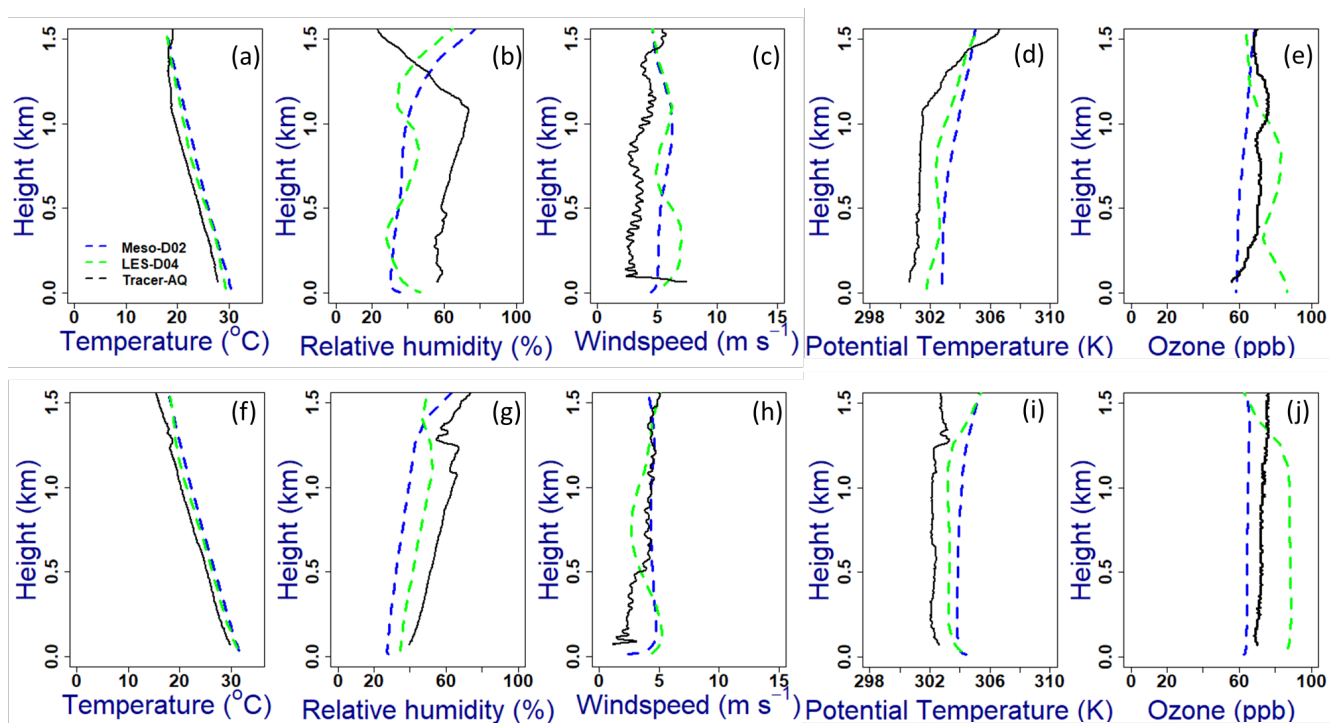
Figure 6: Comparison of vertical profiles of temperature, relative humidity, wind speed, potential temperature, and O₃ mixing ratio from the TRACER-AQ ozonesondes over Galveston Bay (12:47 CDT; a-e) and La Porte (13:07 CDT; f-j) with model results (13:00 CDT). The black lines represent observations; the blue dashed lines are for the D02 results with a resolution of 2700 m; the green dashed lines are for the LES D04 results with a resolution of 300 m.
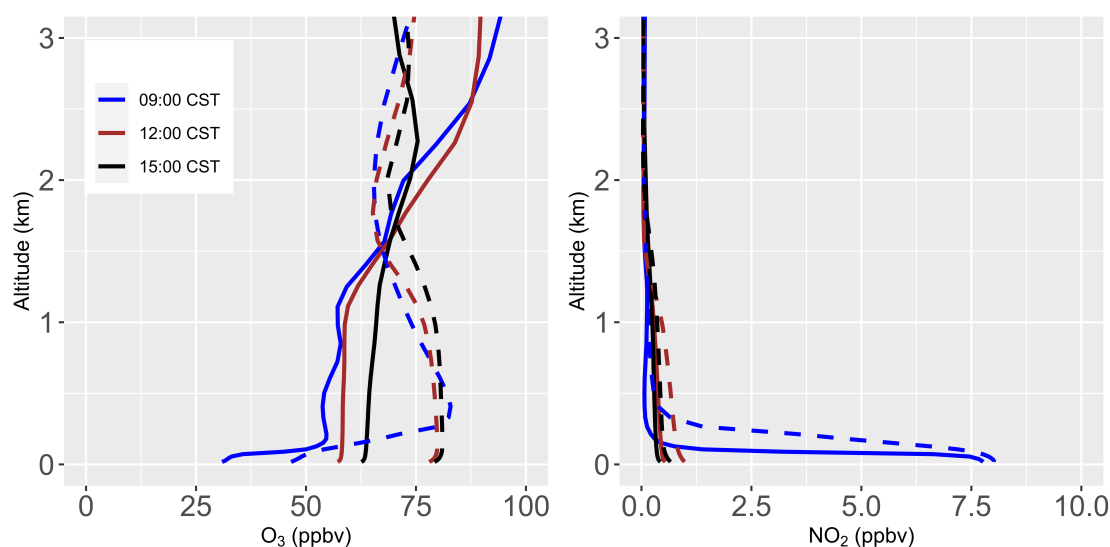
Figure 7: Domain-averaged vertical profiles of O₂ (left panel) and NO₂ (right panel) at 09:00 (blue), 12:00 (brown) and 15:00 CDT (black). Domain D02 results are shown as solid lines while the LES results over the D04 domain are represented as dashed lines.

3.3 O₃ formation regime

Next, we investigate the variability of O_3 production under different formation regimes. High O_3 production rate (PO_3) occurs in the Central Houston region and extends towards the Houston Ship Channel during the daytime (Fig S7). Previous studies have found that O_3 formation is driven by a non-linear photochemical reaction under NO_X-limited and VOC-limited regimes, and determined by different sensitivity indicators (Sillman et al., 2003; Ye et al., 2016). However, O_3 sensitivity indicators, including L_N/Q and FNR, may vary spatially and at different times due to the spatial and temporal heterogeneity of total radical loss via NO_X and radical production as well as the mixing ratio of HCHO and NO_2 (Jin and Holloway, 2015). Figure 8 shows a spatial distribution of surface photochemical regimes associated with O_3 production from the simulated LES domain. Here, L_N/Q indicates O_3 production is mainly a VOC-limited regime ($L_N/Q > 0.5$) in the morning, and then transitions into a NO_X-limited regime ($L_N/Q < 0.5$) in the late afternoon. However, regions around Baytown, Texas City and Deer Park exhibit a VOC-limited or transitional regime also in the late afternoon. The FNR of the whole Houston area in the morning is less than 1.2, indicating that a VOC-limited regime is prevalent during this hour and a noontime transition into a NO_X-limited regime is also observed. It is worth noting that the transitional regime between a VOC-sensitive and NO_X-sensitive regime as defined by the FNR indicator varies with regions as described by Duncan et al. (2010). Previous findings by Schroeder et al. (2017) define the FNR transition regime over Houston between $0.7 < FNR < 2.0$, based on the relative rate of radical termination from radical-radical interactions to radical-NO_X interactions. The ratio of FNR changes in response to the changes in VOCs and NO_X, and variations in NO_2 may have greater impacts on the ratio than variations in HCHO. Thus, it is difficult to define an explicit cut-off for the FNR transition regime based on fluctuations in NO_2. Ambient changes in HCHO and NO_2 respond immediately to changes in emissions sources and FNR will change in accordance, depending on which is predominant. Thus, using the ratio of radical production through HCHO and loss through NO_X provides a more useful determinant for estimating FNR transition ratio of O_3 sensitivity. Hence, using model estimated values of the transition regime from the L_N/Q indicator, we calculated the threshold for FNR transition regime, and find a range of $0.6 < FNR < 1.8$ lies in the transition regime zone. Based on the similarities in the radical indicator approach to define the FNR transition regime, our associated range of FNR values is fairly consistent with the range described by Schroeder et al. (2017).

The discrepancy in our work compared to other studies relating to the modeled $P(O_3)$ when using either L_N/Q or FNR may be attributed to differences in model chemistry and spatial resolution. Previous studies from Ren et al. (2013) and Mazzuca et al. (2016) used a simplified alkene chemistry in the Carbon Bond mechanism, version 5 (CB05) and overpredicted O_3 production in the morning in locations where alkenes dominate VOC reactivity. The alkene chemistry from the MOZART-T1 chemical mechanism used in our work considers relative alkene contributions from biogenic sources (Emmons et al., 2020), which is not often considered in other chemical mechanisms.
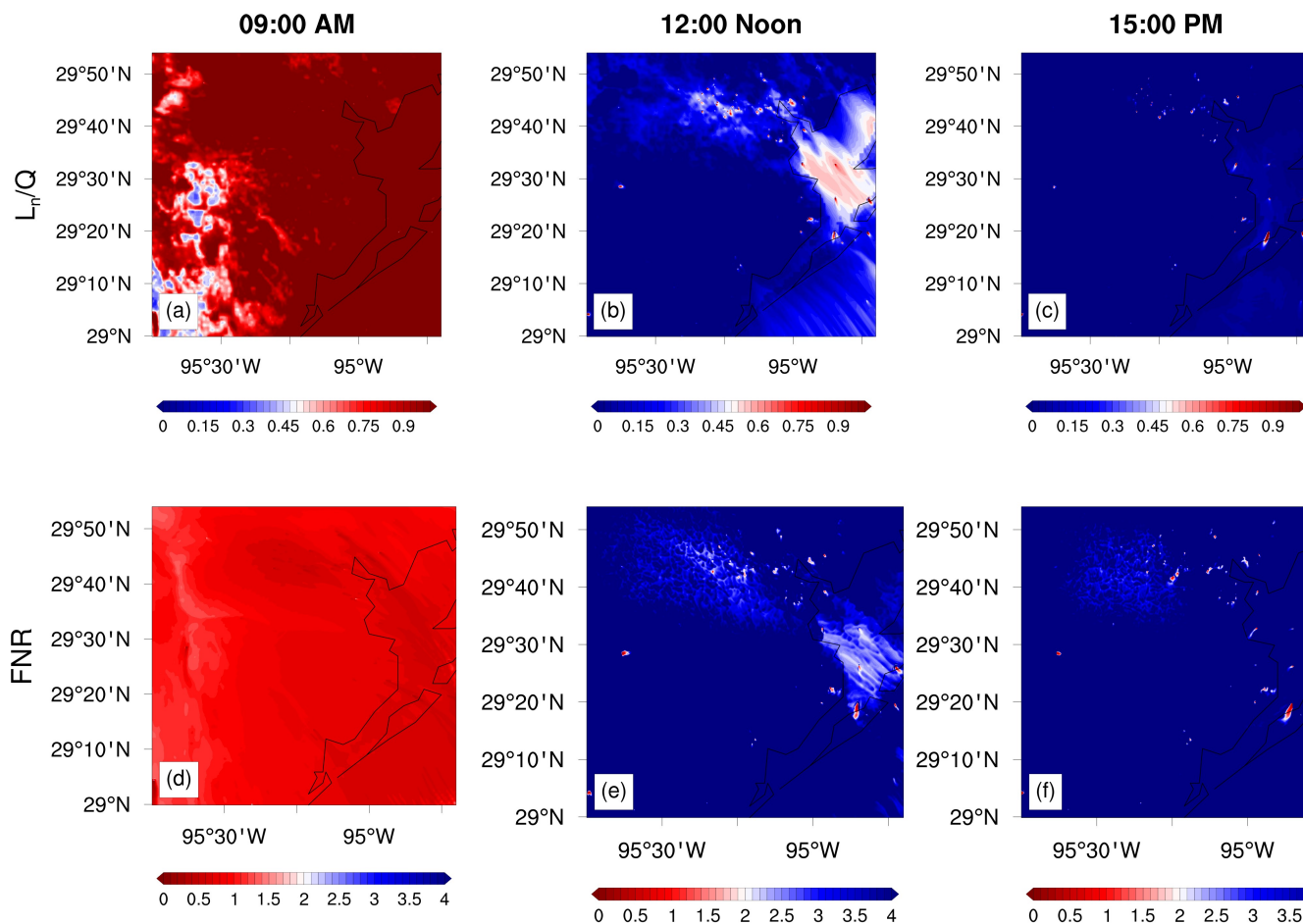
Figure 8: Spatial distributions of the O₃ formation regime determined by L_N/Q (upper panel) and FNR (lower panel) at 09:00 CDT, 12:00 CDT and 15:00 CDT. Calculated based on results from the LES domain over Houston.

The non-linear process of NO_2 and HCHO in relation to O_3 formation and sensitivity regimes is further discussed in Fig. 9 from the model simulations. In Fig. 9, PO_3 describes the rate of ozone production under different sensitivity regimes at specific diurnal hours while O_3 as a function of NO_2 and HCHO depicts the overall daytime concentration of O_3 under these regimes. In the morning, when NO_2 is high but HCHO is low, PO_3 is higher with increasing HCHO, indicating a VOC-limited regime. Though we note that the production rate of HO_X (OH + HO_2) radicals is lower than their loss rate during this period under high NO_2 concentrations, thus leading to a lower production of O_3 in the morning than in the afternoon. In the Supplementary Materials (Fig. S8), the maximum net PO_3 in the morning is estimated at 10 ppb h^{-1} over the whole of Houston. In contrast, during the afternoon when NO_2 is low but HCHO is high, PO_3 increases with increasing NO_2 levels and indicates a NO_X-limited regime. During this period, the rate of HO_X radical production is higher following the oxidation of VOCs by OH, thus

enhancing PO$_3$ in the afternoon. An increase in HO$_2$ reaction rate under a NO$_X$-limited regime leads to the enhancement of O$_3$

production (Li et al., 2023). The dependence of O$_3$ concentrations on NO$_X$ levels during daytime chemistry shows that the O$_3$

470   mixing ratio is a maximum when NO$_X$ levels are low (< 10 ppbv; Fig. S8). PO$_3$ reaches 50 ppb h^{-1} in the late afternoon, which

is higher than the average afternoon O$_3$ production of 20 - 30 ppb h^{-1} observed during the DISCOVER-AQ campaign (Mazzuca

et al., 2016). This further shows the high O$_3$ production during the TRACER-AQ campaign in 2021. The distinction, however,

between these photochemical regimes is not simple and may be influenced by meteorology or unique environmental

characteristics (De Foy et al., 2020; Li et al., 2023).

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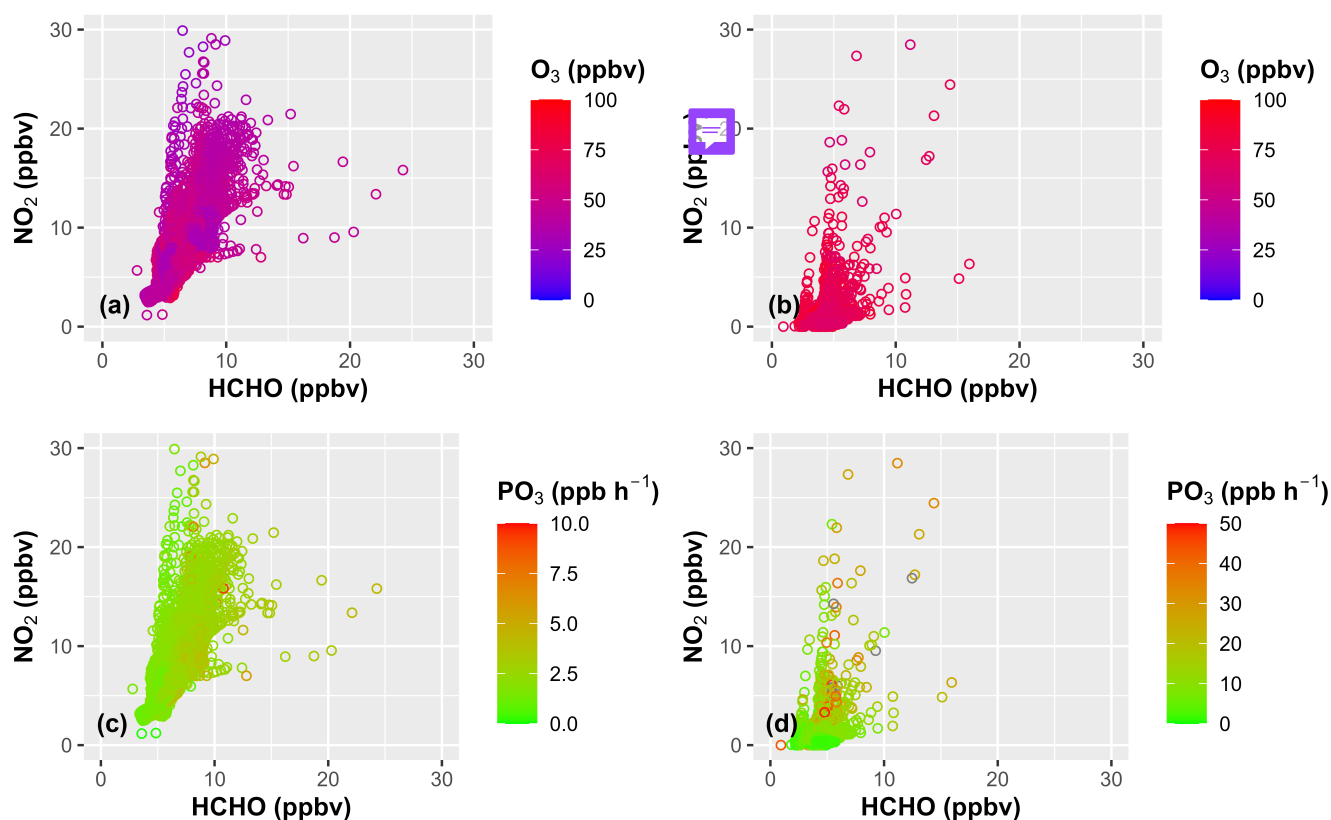
Figure 9: O$_3$ (upper panel) and PO$_3$ (lower panel) as a function of NO$_2$ and HCHO for 09:00 CDT (a and c) and 15:00 CDT (b and d) in Houston from the LES domain. NOTE: we use different color scales for morning and afternoon periods for distinct understanding of ozone formation at different times.

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Since the observed NO$_X$-O$_3$-VOC chemistry in Fig. 8 shows some spatial heterogeneity of the O$_3$ sensitivity regimes, we

further examine the photochemical production of O$_3$ in relation to different environments over Houston. Figure 10 shows the

diurnal variability of O$_3$ and NO$_X$ and the O$_3$ sensitivity regime indicators over four regions including Central Houston, La

Porte, Brazoria-Galveston region (BGR) and the whole of Houston. According to Hossan et al. (2021), the contributions from

485 point sources in La Porte, Texas City near Galveston and downtown Houston contribute most to O_3 formation as point sources alongside other mobile sources. The stations also represent different pollution characteristics to understand how unique emission sources affect the overall photochemical sensitivity to PO_3. For example, heavy power plants contribute the most NO_X in a VOC-limited regime and transportation in a NO_X-limited regime (Jin and Holloway, 2015). According to Nowlan et al. (2018), the highest NO_X emissions in Houston are from the metropolitan area from mobile sources. Among other HRVOCs,

490 light olefins (including butene, ethane and propane) emissions dominate La Porte and Houston Ship Channel from petrochemical plants (Murphy and Allen, 2005). We show that NO_X levels are highest (NO_2 = 26 ppbv; NO = 6 ppbv) during the early morning over La Porte, followed by Central Houston (NO_2 = 22 ppbv; NO = 6 ppbv) and lowest at the Brazoria-Galveston region. The high level of NO_X over La Porte is mainly due to emissions from petrochemical plants and marine vessels at the HSC (Kim et al., 2011) and other mobile sources. On the other hand, NO_X emissions over Central Houston are

495 from mobile sources (Nowlan et al., 2018; Kim et al., 2011), while the NO_X levels from BGR can be attributed to petroleum refining sources around Texas City (Souri et al., 2018). Notably, NO_X levels are generally low from the late morning hours into the afternoon and underline the diurnal variability of NO_X emission sources as well as the fast photodissociation of NO_X. Although the NEI anthropogenic emissions inventory reasonably captures these diurnal patterns, improvements are further needed given its overestimation of NO_X in Houston.

500 The diurnal variation of L_N/Q and FNR indicates that O_3 formation is mainly a VOC-limited regime in the early morning over the Central Houston, La Porte and Whole Houston (domain average), and then transitions into a NO_X-limited regime later in the day (Fig. 10). During the morning hours, the lower O_3 mixing ratio in Central Houston and La Porte is mainly associated with a VOC-limited regime due to higher NO_X levels; however, they increase gradually towards noontime as NO_X decreases with increasing VOCs. This is consistent with previous studies that have shown that O_3 production is mainly suppressed by

505 high NO_X concentrations in a VOC-limited regime (Li et al., 2011). On average, O_3 formation is mainly NO_X-limited in the afternoon (between 11:30 and 16:00 CDT) across all the regions, and the high O_3 (Fig. 10) is dependent on the daytime abundance of RO_2 and HO_2 radicals through the photolysis of O_3, HCHO and other secondary VOCs (Ma et al., 2022). In the Brazoria-Galveston region, the high O_3 in the early morning hours is associated with O_3 production being mainly NO_X-limited between 6:30 and 7:30 CDT (L_N/Q < 0.5) before transitioning to a VOC-limited regime in the remaining hours of the morning

510 (8:00 and 11:30 CDT). During this NO_X-limited period, high sources of NO_2 rapidly increases O_3 formation (Duncan et al., 2010; Schroeder et al., 2017). In addition, we show how O_3 formation regime may vary spatially with height in the Supplementary Information (Fig. S9). We find that the production of O_3 is under a VOC-limited regime at the surface in the morning and transitions into a NO_X-limited regime in the upper atmosphere (> 200 m AGL) due to the rapid photochemical consumption nature of NO_X near the surface.

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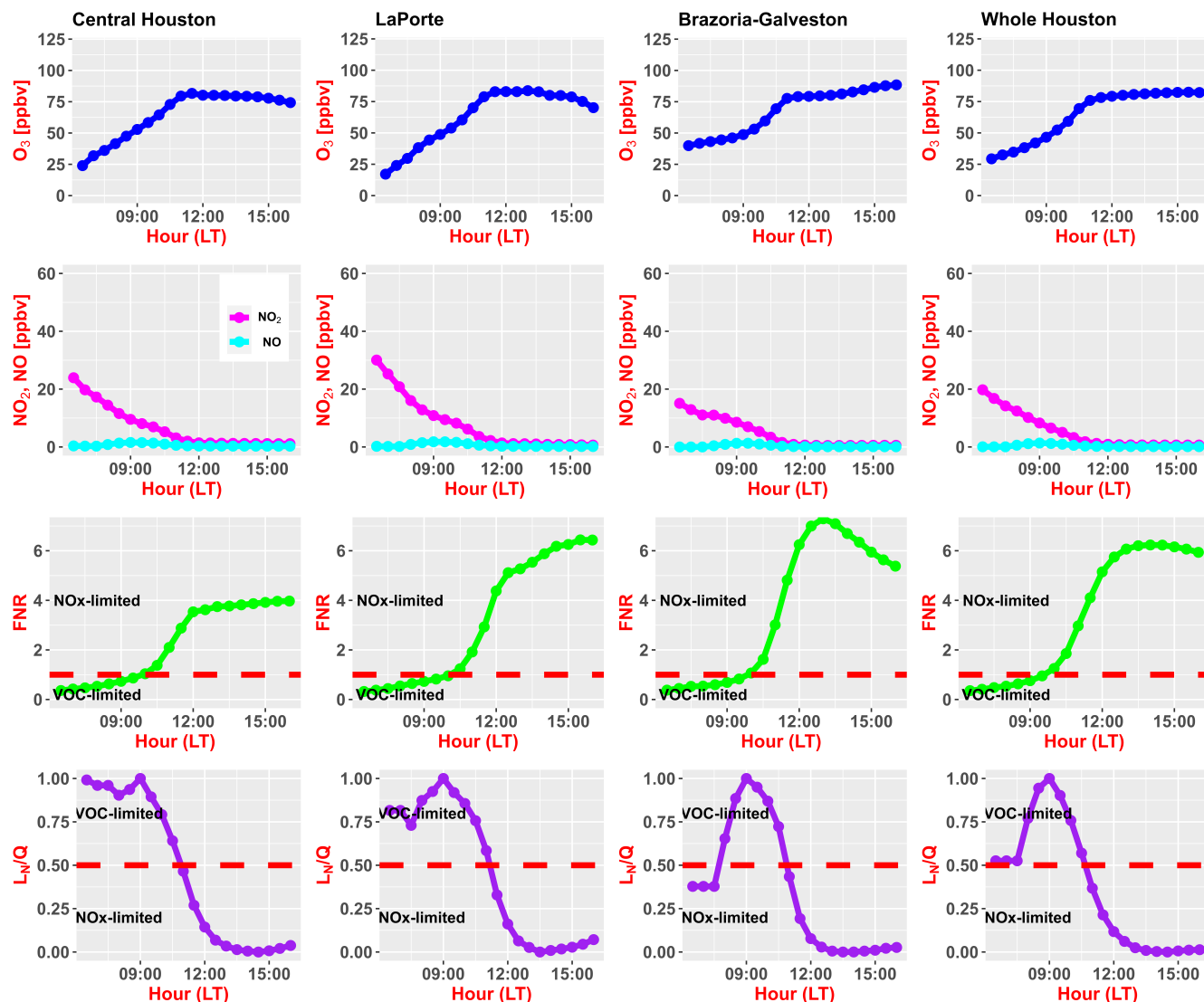
Figure 10: Diurnal variations of O$_3$ (first row), NO and NO$_2$ (second row), FNR (third row) and L$_N$/Q (fourth row) in the near-surface layer averaged over Central Houston (first column), La Porte (second column), Brazoria-Galveston (third column) and the whole LES domain (fourth column). Results are from the LES domain (D04).

3.4 OH reactivity

We further quantify the total OH reactivity (K$_{OH}$) to characterize the roles of VOCs in the formation of O$_3$. K$_{OH}$ is defined as the sum of concentrations of OH reactants multiplied by their reaction rate coefficients (Yang et al., 2016; Fuchs et al., 2017; Ma et al., 2022), as shown in Eq. (4)

$$K_{OH} = \sum_i^n K_{VOC_i} \times [VOC_i] \qquad (4)$$

525 where [VOC$_i$] is the VOC concentrations and K$_{VOCi}$ is the corresponding reaction rate coefficients. In this study, K$_{OH}$ is calculated from 29 major modeled VOC species which are grouped into three groups, including non-methane VOCs (NMVOCs), OVOCs and biogenic VOCs (BVOCs). The NMVOCs consist of alkanes (ethane, propane and lumped alkanes with C > 3), alkenes (ethene, propene/propylene, lumped alkenes with C > 3), alkynes (acetylene), aromatics (benzene, toluene, xylene) and glyoxal, while the BVOCs include isoprene and monoterpenes (limonene, α-pinene, β-pinene, β-caryophyllene

530 and other sesquiterpenes). The OVOCs species are mainly secondary VOC species and include ketones (methyl vinyl ketone, methyl ethyl ketone, acetone), aldehydes (formaldehyde, acetaldehyde, methacrolein, benzaldehyde, glycolaldehyde, lumped aldehyde), acids (formic acid, acetic acid), and alcohols (methanol, ethanol and phenol). Table 1 provides a summary of the total OH, O$_3$ and NO$_3$ reactivity with individual VOC species. The OH reactivity analysis shows isoprene is the highest contributor to total K$_{OH}$ at noon (12:00 CDT) at 9.86 s^{-1} (35.76 %), followed by CO (12.98 %), HCHO (12.21 %), Bigalk

535 (lumped alkanes; 6.29 %), and NO$_2$ (4.46 %). The contribution of the daytime reactivity of isoprene to the total K$_{OH}$ is substantial and very high when compared to other urban studies by Fuchs et al. (2017, 20 %) and Ma et al. (2022, 5 %). The contribution of HCHO (12.21 %) and acetaldehyde (4.88 %) as the most dominant OVOC species in reactivity with OH is consistent with the established literature from other locations (Whalley et al., 2016; Ma et al., 2022). Also, the high value of HCHO reactivity is related to the high values from the oxidation of isoprene, as HCHO in the Houston-Brazoria region is

540 predominantly from VOC oxidation (Parrish et al., 2012; Zhou et al., 2014). For most species, the modeled K$_{OH}$ reduces at 15:00 CDT compared to the noontime, while isoprene was higher at 15:00 CDT (9.98 s^{-1}; 46.61 %). In addition, we consider the modeled reactivity of O$_3$ and NO$_3$ with VOCs and show that the total O$_3$ reactivity is dominated by the BVOCs. For example, the ozonolysis of β-caryophyllene increases from 0.39 s^{-1} at noon to 0.84 s^{-1} (27.07 %) in the late afternoon, due to the increased concentration of O$_3$ at 15:00 CDT.

545 The overall breakdown of the VOC groups and their OH reactivity over the focused regions averaged from 6:30 to 16:00 CDT is summarized in Fig. 11. The daytime total VOC concentration is dominated by OVOCs (73 %), while BVOCs contribute the least (~ 3 %). In comparison to other regions, the maximum VOC concentration is observed in La Porte. On the other hand, the total K$_{OH}$ is dominated by BVOCs (58 %) in the Central Houston region. From individual VOC species, NMVOCs species are mostly dominant with alkanes contributing to about 50–55 % of the total mixing ratio but are less reactive with OH radicals,

550 while alkenes are more reactive with OH radicals and contribute 60-65 % of the total K$_{OH}$ from NMVOCs across all zones. For BVOCs, isoprene is highly dominant with monoterpenes contributing less than 2 % in the total BVOC mixing ratio and reactivity with OH and mixing ratio. Aldehydes are the most reactive OVOCs (70-73 %), and alcohols dominate the total OVOC concentration (~50 %).

Figure 12 shows the mixing ratio and OH reactivity of individually modeled compounds as a stacked area plot over the whole

555 Houston region. In the daytime, isoprene as the main biogenic species accounted for more than half of the total reactivity, while VOCs from anthropogenic activities and in oxygenated forms including alkanes, alkenes, aldehydes, alcohols and aromatics account for the other 50 % of the total OH reactivity. For NMVOCs and OVOCs, the total OH reactivity peaks at about 11:00 (CDT), while the OH reactivity of isoprene increases continuously during the day and peaks at 16:00 CDT. For

the mixing ratio, the NMVOCs peak in the early morning hours and gradually reduce towards late afternoon. Similar stacked
560 area plots for Central Houston, La Porte and the BGR are provided in the Supplementary Materials (Fig. S10-S12). The results
are consistent with the diurnal patterns observed over the whole Houston area, but a higher magnitude for total OH reactivity
of NMVOCs and OVOCs is observed in La Porte, while isoprene contributes the highest to the total K_{OH} in Central Houston.
The analysis of OH reactivity with VOC is important to estimate the potential of O_3 formation from VOC oxidation. The
oxidation of VOC by OH produces RO_2 and HO_2 radicals, which is identified as the major contributor to O_3 formation in
565 Houston (Zhou et al., 2014). While we show that isoprene contributes the highest to the total K_{OH} due to its high reactivity
with OH, it is unknown how much this affects O_3 formation. A recent study from Li et al. (2023) shows that isoprene reactivity
did not increase during high O_3 events during the NASA TRACER-AQ campaign, and shows similar OH reactivity on clean
O_3 days. OVOCs, similarly, show a high contribution to the total OH reactivity, particularly from HCHO and acetaldehyde,
however, they are considered the most important VOC species oxidized by OH that increases O_3 production (Zhou et al., 2014;
570 Li et al., 2023). According to previous studies in Houston, HCHO represents the highest contributor to HO_2 production (Parrish
et al., 2012; Ren et al., 2013; Wong et al., 2013). From the Supplementary Material (Table S2), we also show the photolysis
of HCHO, methyl glyoxal and glyoxal dominates the production of HO_X radicals over the whole Houston region. HCHO loss
rate through photolysis peaks (with a maximum of 0.14 pptv s^{-1}) at 12:00 CDT, similar to other loss rates across cities in
Europe at 0.28 pptv s^{-1} (Nussbaumer et al., 2021). Industrial emissions are identified as the main primary source of HCHO in
575 the Houston area (Johansson et al., 2014) with an average emission over the Houston region estimated at 8631.02 mol km^{-2} hr^{-1}
over the simulation period based on the NEI inventory. In contrast, a comprehensive chemical analysis of airborne and ground
measurements of HCHO and emissions data concluded that HCHO is mainly from VOC oxidation in the Houston area (Parrish
et al., 2012). As HCHO is one of the major oxidation products of isoprene by OH (Sprengnether et al., 2002), we conclude
that high isoprene reactivity with OH enhances HCHO formation; the photolysis of HCHO forms HO_2 which then increases
580 the potential of O_3 formation.

Table 1: Reactivity of OH, O_3, and NO_3 with key chemical species (s^{-1}) averaged over the LES domain.

HRVOC Groups	Species	OH reactivity		O_3 reactivity		NO_3 reactivity	
		12:00 Noon	15:00 CDT	12:00 Noon	15:00 CDT	12:00 Noon	15:00 CDT
NMVOCs	Methane	0.29165	0.215078	-*	-	-	-
	Ethane	0.003451	0.002297	-	-	-	-
	Propane	0.008885	0.004246	-	-	-	-
	Bigalk (C_5H_{12})	0.481717	0.630683	-	-	-	-
	Ethene	0.202339	0.095153	0.016405	0.014473	-	-
	Propene	0.087006	0.047582	0.013262	0.015315	0.000033	0.000018
	Bigene (C_4H_8)	0.053194	0.042853	-	-	0.000394	0.000296
	Acetylene/Propylene	0.000193	0.000122	-	-	-	-
	Benzene	0.001054	0.000575	-	-	-	-
	Toluene	0.032341	0.011311	-	-	-	-
	Xylene	0.025272	0.009502	-	-	-	-
OVOCs	Formaldehyde	0.934925	0.486296	-	-	0.000076	0.000034

	Acetaldehyde	0.374008	0.165279	-	-	0.000066	0.000026
	Methacrolein	0.127278	0.098421	0.002455	0.003566	-	-
	Benzaldehyde	0.002014	0.000886	-	-	-	-
	Glycolaldehyde	0.086827	0.076907	-	-	-	-
	Methyl ethyl ketone	0.026124	0.012443	-	-	-	-
	Methyl vinyl ketone	0.058917	0.0486	0.006997	0.009998	-	-
	Acetone	0.020977	0.013498	-	-	-	-
	Acetic acid	0.011161	0.007475	-	-	-	-
	Formic acid	0.000758	0.000822	-	-	-	-
	Methanol	0.254674	0.166337	-	-	-	-
	Phenol	0.000364	0.000221	-	-	-	-
	Ethanol	0.154746	0.065338	-	-	-	-
	2-methyl-3-buten-2-ol (MBO)	-	-	0.000112	0.000197	-	-
	Glyoxal	0.036711	0.023485	-	-	-	-
	Methylglyoxal	0.079254	0.044156	-	-	0.000019	0.000009
BVOCs	Isoprene	2.738534	2.772071	0.271831	0.5330888	0.028773	0.02584423
	Limonene	0.014099	0.013134	0.011429	0.01969507	0.001292	0.00109182
	α-pinene	0.007128	0.00584	0.007377	0.01115669	0.001146	0.00082383
	β-pinene	0.006993	0.005868	0.001247	0.00197046	0.000317	0.00023717
	β-caryophyllene and other sesquiterpenes ($C_{15}H_{24}$)	0.003188	0.003879	0.109703	0.23419166	0.00041	0.00044159
Other trace species	CO	0.994169	0.644294	-	-	-	-
	SO_2	0.062321	0.026949	-	-	-	-
	O_3	0.124186	0.094638	-	-	-	-
	NO_2	0.341709	0.101966	-	-	0.044032	0.00924
	NO	-	-	-	-	0.152788	0.041117

Bigalk represents lumped alkanes with C > 3, and BIGENE for lumped alkenes with C > 3.

- also represents chemical reactivity not included in the model chemical mechanism*)
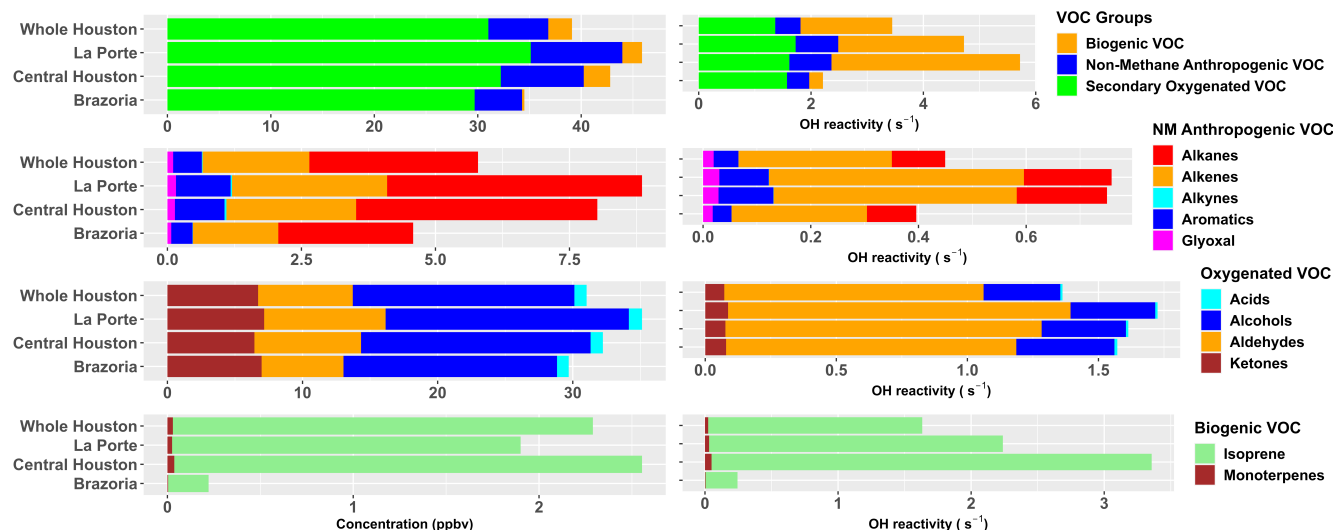
Figure 11: The breakdown of grouped VOC mixing ratio (left panel) and OH reactivity (right panel) averaged over Central Houston, La Porte, Brazoria-Galveston and the whole LES domain.
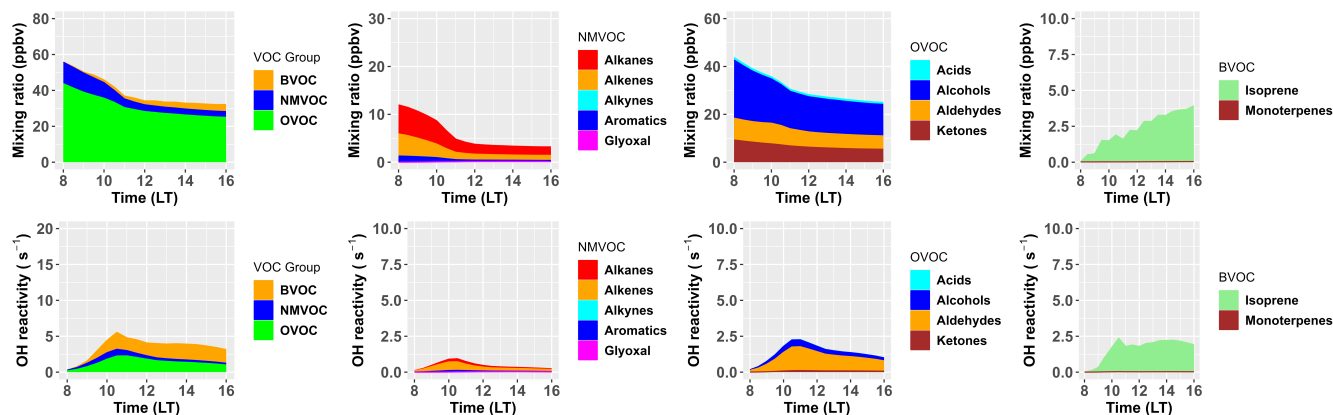
Figure 12: Diurnal variations of VOC mixing ratio (upper panel) and OH reactivity (lower panel) averaged over the LES domain for (a, e) the all-VOC group, (b, f) NMVOC, (c, g) OVOC and (d, h) BVOC.

3.5 AOC analysis

The oxidative capacity of the atmosphere is important to understand the abundance of RO_X radicals including HO_2, OH and RO_2 radicals, which drive the production, transformation and recycling of O_3 (Liu et al., 2022). AOC is calculated as the sum of oxidation rates of various pollutants by the major oxidants (OH, NO_3 and O_3) (Xue et al., 2016; Chen et al., 2020; Liu et al., 2022). In this study, AOC is calculated based on the modeled VOCs and their rate coefficients, as shown in Eq. (5-7), and does not account for the reactivity of OH, NO_3 and O_3 with other VOC oxidation products that are not included in the gas phase mechanism used within our WRF-LES-Chem modeling. Thus, we acknowledge that possible discrepancies may exist between

the calculated values of these reactivities and their actual measurements, as shown in previous studies (Mao et al., 2010; Ren et al., 2013).

$$Total\ OH\ reactivity = \begin{array}{l} \sum k_{OH+NMVOC_i}[NMVOC_i] + \\ k_{OH+CH_4}[CH_4] + k_{OH+CO}[CO] + \\ k_{OH+NO}[NO] + k_{OH+NO_2}[NO_2] + \\ k_{OH+SO_2}[SO_2] + k_{OH+O_3}[O_3] + \cdots \end{array} \tag{5}$$

$$Total\ O_3\ reactivity = \begin{array}{l} \sum k_{O_3+NMVOC_i}[NMVOC_i] + \\ k_{O_3+CH_4}[CH_4] + k_{O_3+NO}[NO] + \\ k_{O_3+NO_2}[NO_2] + \cdots \end{array} \tag{6}$$

$$Total\ NO_3\ reactivity = \begin{array}{l} \sum k_{NO_3+NMVOC_i}[NMVOC_i] + \\ k_{NO_3+CH_4}[CH_4] + k_{NO_3+NO}[NO] + \\ k_{NO_3+NO_2}[NO_2] + k_{NO_3+SO_2}[SO_2] + \cdots \end{array} \tag{7}$$

Figure 13 shows the time series of the model-calculated AOC over Houston. The estimated AOC daytime averages (08:00 – 16:00 CDT) is 1.11 x 10^8 molecules cm^{-3} s^{-1}, 0.88 x 10^8 molecules cm^{-3} s^{-1}, 0.52 x 10^8 molecules cm^{-3} s^{-1} and 0.69 x 10^8 molecules cm^{-3} s^{-1} respectively for Central Houston, La Porte, Brazoria-Galveston and Whole Houston. As shown in Fig. 13, the daily maximum AOC is observed in Central Houston and La Porte at 1.5 x 10^8 molecules cm^{-3} s^{-1}, and much higher than those in the less polluted region in Brazoria-Galveston at 0.8 x 10^8 molecules cm^{-3} s^{-1}. The daily maximum AOC is similar to other areas with high pollution from oil and gas in China (1.8 x 10^8 molecules cm^{-3} s^{-1}; Chen et al., 2020) or other urban areas in Hong Kong (1.3 x 10^8 molecules cm^{-3} s^{-1}; Xue et al., 2016) or Xiamen (1.3 x 10^8 molecules cm^{-3} s^{-1}; Liu et al., 2022). The differences in AOC across these regions may be attributed to relatively higher solar radiation or pollutant concentrations among different sites (Zhu et al., 2020; Liu et al., 2022). As expected, OH is the major daytime oxidant, accounting for more than 90 % of the total AOC. In the late afternoon, the contribution of O_3 to AOC increases up to 8-10 % in Central Houston and La Porte compared to the morning hours when they contribute less than 5 %. In addition, the higher contribution of O_3 to AOC is attributed to the high concentration of alkenes in the region (Xue et al., 2016), and alkene ozonolysis increases as the day progresses with the abundance of O_3. AOC contributed by NO_3 is negligible as NO_3 can be quickly photodissociated during daytime thus having a low daytime concentration. Figure 14 elucidates the daytime variability of primary anthropogenic VOCs, OVOCs and trace species and their reactivity with OH, O_3 and NO_3. In Brazoria-Galveston, the oxidation of alkanes, CO and methane (CH_4) by OH increases slightly as the day progresses into late afternoon. The simulated OVOCs are the dominant contributor to OH reactivity and contribute 30-40 % of the total AOC across all regions. This is similarly evident over the whole of Houston. Further information from the Supplementary material shows that the total OH reactivity for OVOCs is much higher (1.36 ± 0.59 s^{-1}) (mean ± standard deviation) on average over the whole domain compared to other groups (see Table S3), while CO (0.72 ± 0.29 s^{-1}), CH_4 (0.2 ± 0.09 s^{-1}), alkanes (0.3 ± 0.12 s^{-1}), alkenes (0.28 ± 0.16 s^{-1}), aromatics (0.05 ± 0.03 s^{-1}), and SO_2 (0.04 ± 0.02 s^{-1}) also make up the averaged total OH reactivity, thus indicating the strong influence of anthropogenic emissions in Houston. The comparison across different zones shows that the oxidation of alkanes (0.39 ± 0.19

625 s^{-1}), aromatics (0.1 ± 0.07 s^{-1}), and CO (0.93 ± 0.47 s^{-1}) is highest over Central Houston, while the oxidation of alkenes (0.47 ± 0.30 s^{-1}) and SO$_2$ (0.06 ± 0.04 s^{-1}) is mostly dominant over La Porte. In contrast, the OH reactivity via CH$_4$ oxidation (0.27 ± 0.13 s^{-1}) is the highest over the BGR region. CH$_4$ represents an important factor that affects AOC, with OH + CH$_4$ being an important sink for OH in the atmosphere (Morgenstern et al., 2013). Overall, this underlines the role of spatial characteristics and unique local anthropogenic emission sources, which may contribute to the overall photochemical processes.

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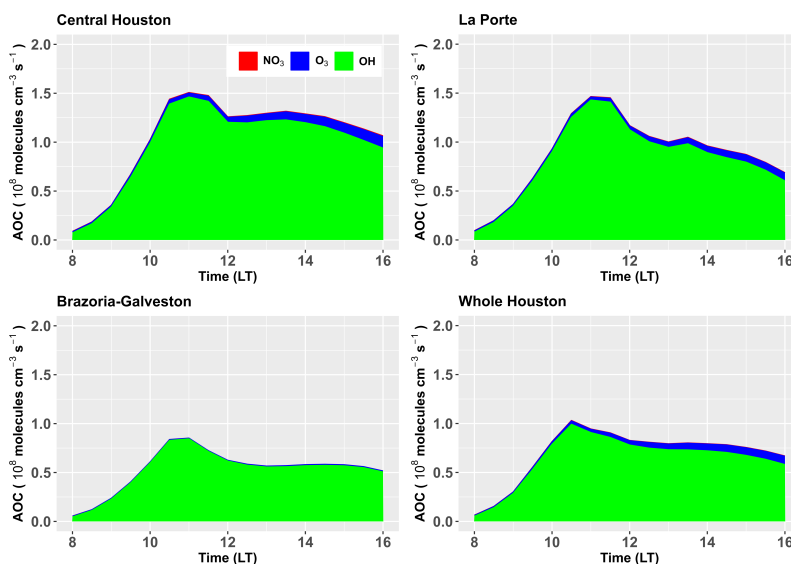
Figure 13: Diurnal variations of AOC averaged over the Central Houston, La Porte, Brazoria-Galveston and the whole LES domain. Results are from the LES -domain.
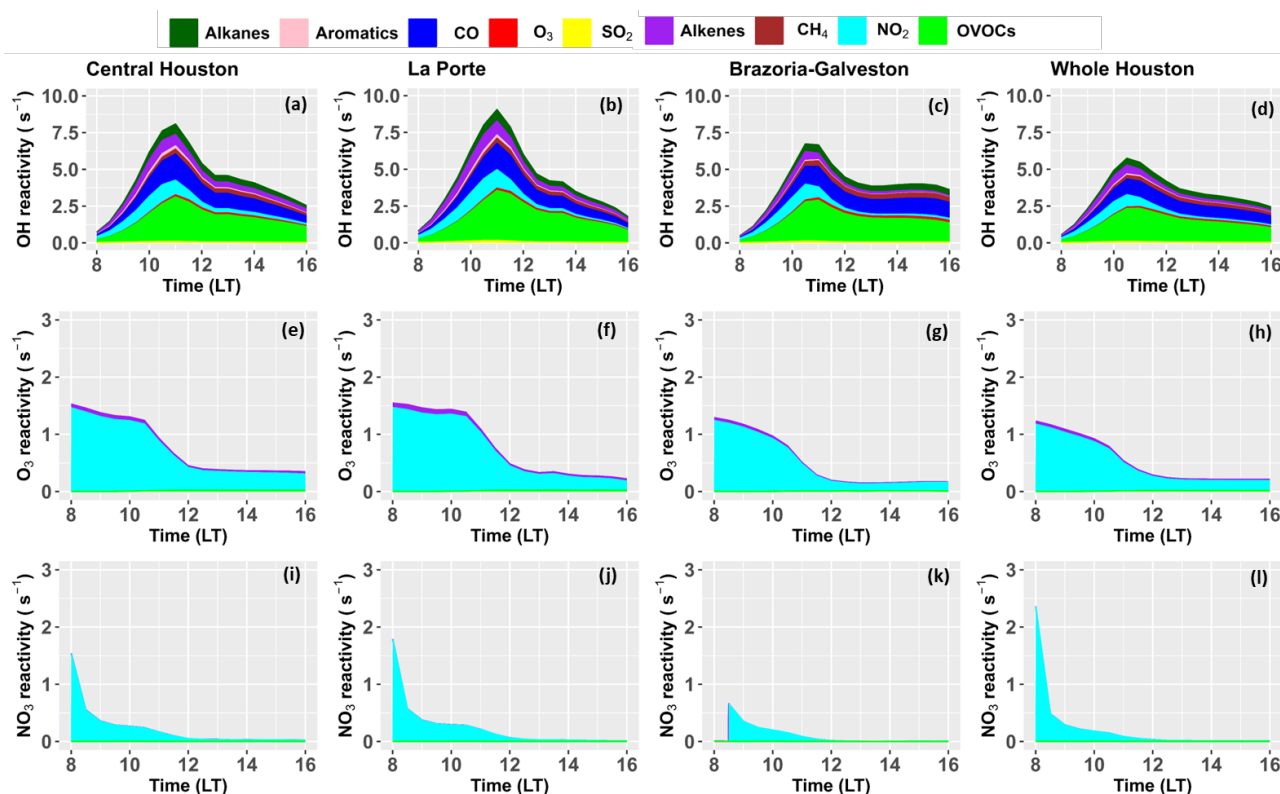
Figure 14: Diurnal variations of OH reactivity (a-d), O_3 reactivity (e-h), and NO_3 reactivity (i-l) of selected major anthropogenic chemicals averaged over the Central Houston (first panel), La Porte (second panel), Brazoria-Galveston (third panel) and the whole LES domain (fourth panel). Results are from the LES domain.

4. Conclusion

In this study, we investigate the detailed atmospheric photochemistry during a high O_3 episode during TRACER-AQ 2021 in Houston. During this episode, observed surface O_3 significantly exceeded national standards by 20-50 ppb, particularly in the Central areas of Houston. We use a high-resolution WRF-LES-Chem model to simulate atmospheric O_3 as well as the detailed atmospheric photochemistry associated with these O_3 exceedances. By resolving turbulence and eddies at a scale relevant for pollutant chemistry, the LES simulation 300 m spatial resolution adequately captures the surface concentration of O_3, NO_X, and HCHO, and provides an improved representation of temperature, wind speed and water vapor compared to the mesoscale WRF-Chem simulations.

Two indicators for the O_3 sensitivity regime (L_N/Q and FNR) are compared and show that the photochemical properties of O_3 formation in Houston is largely a VOC-limited regime in the morning hours and transitions into a NO_X-limited regime in the afternoon. Additionally, the transition threshold for radical loss-initiation ratio, L_N/Q, is used to define the FNR transition regime, which is observed at 0.6 < FNR < 1.8. Some local regions, including Baytown, Texas City and Deer Park show no diurnal variability in O_3 formation sensitivity regimes, due to unique environmental and emission sources. On September 8,

2021, the rapid radical production and recycling of HO_X radicals and their reactivity with increased VOC emissions and reduced NO_2 emissions under a NO_X-limited regime accelerated fast and high O_3 production. The total radical production rate of HO_X radicals increased PO_3 in the afternoon hours, reaching 50 ppb h^{-1}, and higher than the previously recorded average in the DISCOVER-AQ campaign.

655 Increased HRVOC emission is the main cause of radical production. The HO_2 radical is mainly formed from the oxidation of HCHO and represents the highest source of HO_2 in Houston. The oxidation of other OVOCs including acetaldehyde also enhances the production of HO_2 radicals, which contributes to increased O_3 production. Our analysis linked the high oxidation of isoprene to enhance the production of HCHO, thus increasing the potential of O_3 formation when oxidized by OH. Our results further underline the role of BVOCs and how they may account for enhanced O_3 production, which is consistent with

660 some recent studies (Leong et al., 2017; Ma et al., 2022). Similarly, RO_2 sources are alkenes oxidation, as compared to the oxidation of alkanes and aromatics.

Overall, the role of anthropogenic emissions cannot be underestimated. Primary emissions of CO and non-methane hydrocarbons (alkenes and other alkanes) affect the overall AOC levels, though the contribution of each compound to AOC varies with location and time. OVOCs also contribute about 30-40 % to the total AOC. Our study thus concludes the important

665 role of anthropogenic emissions in the production of RO_2 and HO_2 radicals, and how they may influence the enhancement or inhibition of O_3 formation, depending on the NOx-O_3-VOC sensitivity regimes. This suggests the importance of further reductions in highly reactive anthropogenic VOCs to achieve further reductions in O_3 levels.

Code and data availability

670 The source code of WRF-Chem is publicly available on GitHub at https://github.com/wrf-model/WRF. Observational data from TCEQ and Pandora can be downloaded from the TAMIS web interface (https://www17.tceq.texas.gov/tamis/index.cfm; last access: 30 June 2023) and the Pandonia Global Network (https://www.pandonia-global-network.org/; last access: 3 July 2023) respectively. The TRACER-AQ data is archived on the NASA webpage (https://www-air.larc.nasa.gov/cgi-bin/ArcView/traceraq.2021). The Lidar data from the TRACER-AQ campaign is also available on the NASA Tropospheric

675 Ozone Lidar Network (https://tolnet.larc.nasa.gov).

Authors contributions

AF performed the simulations, carried out the analyses and wrote the manuscript. YL conceived and supervised the study and revised the paper. JD provided guidance in modeling setup and contributed to the editing of the paper. RS, SU, PW, JS, GG, JF, ME and TG provided measurement datasets and contributed to the editing of the paper.

680 **Competing interests**

The authors have declared no competing interests.

Disclaimer

No Disclaimer.

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