



## Gas–particle partitioning of semivolatile organic compounds when wildfire smoke comes to town

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**Abstract.** Wildfires have become an increasingly important source of organic gases and particulate matter in the western USA. A large fraction of organic particulate matter emitted in wildfires is semivolatile, and the oxidation of organic gases in smoke can form lower-volatility products that then condense on smoke particulates. In this research, we measured the gas- and particle-phase concentrations of semivolatile organic compounds (SVOCs) during the 2017 northern California wildfires in a downwind urban area, using semivolatile thermal desorption aerosol gas chromatography (SV-TAG), and measured SVOCs in a rural site affected by biomass burning using cTAG (comprehensive thermal desorption aerosol gas chromatography mass spectrometry) in Idaho in 2018. Commonly used biomass burning markers such as levoglucosan, mannosan, and nitrocatechols were found to stay predominantly in the particle phase, even when the ambient organic aerosol (OA) was relatively low. The phase partitioning of SVOCs is observed to be dependent on their saturation vapor pressure, while the equilibrium absorption model underpredicts the particle-phase fraction of most of the compounds measured. Wildfire organic aerosol enhanced the condensation of polar compounds into the particle phase but not some nonpolar compounds, such as polycyclic aromatic hydrocarbons.

## 1 Introduction

In the western USA, wildfires have become an increasingly important source of organic gases and particulate matter in the atmosphere (Dennison et al., 2014; McClure and Jaffe, 2018; Iglesias et al., 2022). Biomass burning (BB) in wildfires emits thousands of organic compounds with vastly different properties (such as volatility and reactivity; Jen et al., 2019; Hatch et al., 2015; Liang et al., 2022a). Many of these compounds have known direct health impacts, and the oxidation of organic compounds can also produce secondary organic aerosols (SOAs), which are also hazardous (Kim et al., 2018; Tuet et al., 2017, 2019; Wong et al., 2019).

Among the wildfire emissions, the semivolatile and intermediate-volatility organic compounds (SVOCs and IVOCs), with effective saturation concentrations ( $C^*$ ) between  $10^{-1}$  and  $10^6 \mu\text{g m}^{-3}$  are of particular interest because of their partitioning between the gas and particle phases, which can either decrease or increase the mass concentration of organic aerosol (OA, which only refers to the particles hereafter; Hodshire et al., 2019). May et al. (2013) found that the majority of primary organic aerosol (POA) emitted from BB is semivolatile. In a source apportionment analysis, an ideal tracer for a source-specific factor should be a stable compound with a relatively low volatility compared with the bulk BBOA (biomass burning organic aerosol; Donahue et al., 2012). However, levoglucosan, the most commonly used BB marker, was also found to evaporate and photochemically decay in the atmosphere (Hennigan et al., 2010; Xie et al., 2014). Through in situ measurement, Palm et al. (2020) found that up to a third of the BB POA evaporated as a wildfire smoke plume diluted by a factor of 5–10, and the majority of SOA formed was from the oxidation of evaporated BB POA. However, the molecular identities of these evaporated compounds are not known. The BB SOA compounds formed by atmospheric oxidation of primary emissions may also be semivolatile. For instance, Palm et al. (2020) also found that the nitrophenolic compounds, which were predicted to be in the particle-phase only from thermal desorption analysis (Finewax et al., 2018), may also exist in the gas phase. These knowledge gaps motivated our study of the gas–particle-partitioning behaviors of BB POA and SOA compounds in the atmosphere.

When wildfire smoke enters the urban area, the gases and particles in the smoke plumes interact with urban air pollutants. In absorptive partitioning theory that is assumed in most atmospheric models, the fraction of organic compounds in the particle phase increases with the mass of absorbing aerosol (Pankow, 1994; Donahue et al., 2006). Therefore, the massive amount of BBOA (which can exceed  $100 \mu\text{g m}^{-3}$  of fine particulate matter) entering the urban atmosphere is expected to absorb SVOCs into the condensed phase to form SOA. The BBOA may also affect the downwind gas–particle partitioning of compounds such as polycyclic aromatic hydrocarbons (PAHs) and phthalates in the urban area,

thereby altering the mechanisms and magnitude of their deposition in the respiratory tract (Pankow, 2001). However, whether preexisting organic aerosol can alter the partitioning behavior of SVOCs also depends on the aerosol composition (Asa-Awuku et al., 2009; Ye et al., 2016, 2018; Liu et al., 2021). Differences in the composition between urban aerosol and BBOA may make them (or some components of them) not miscible. Mahrt et al. (2022a) recently reported that two types of organic aerosol with O/C difference under 0.265 can probably mix with each other, while two phases will remain if the difference in O/C is above this threshold. BBOA can have O/C ratios above 0.6, while cooking OA and hydrocarbon-like OA, which accounts for nearly half of the urban OA in Oakland, CA (a city adjacent to Berkeley), have O/C ratios between 0.01 and 0.11 (Zhou et al., 2017; Shah et al., 2018). It is possible that the polar BBOA compounds can increase the activity coefficients of the relatively nonpolar organic compounds and expel them into the gas phase. In addition, BBOA was found to be semisolid, even under high relative humidity ( $\text{RH} \sim 75\%$ ), which can limit the gas–particle partitioning of organic compounds (Bateman et al., 2017). Measurements are therefore needed to find out whether wildfire aerosol can shift the gas–particle partitioning of SVOCs.

In this study, we measured the hourly concentrations of organic compounds using the semivolatile thermal desorption aerosol gas chromatography (SV-TAG) in Berkeley, California, during the October 2017 northern California wildfires. Concentrations of organic compounds were also measured in McCall, Idaho, in August 2018, as a part of the Fire Influence on Regional to Global Environments and Air Quality (FIREX-AQ) study, using the comprehensive thermal desorption aerosol gas chromatography (cTAG). The TAG instruments can achieve molecular speciation at an hourly time resolution, which enables us to better explore the impact of environmental factors on the gas–particle partitioning of individual compounds. Our main aims here are to examine the gas–particle partitioning behaviors of organic compounds, especially BBOA marker compounds when BB smoke is diluted and enters the urban atmosphere, and to evaluate the impact of the massive amount of wildfire aerosol on the partitioning behaviors of SVOCs (including IVOCs, hereafter).

## 2 Materials and methods

### 2.1 Field sites, measurement periods, and the wildfires

The 2017 northern California wildfires started on 8 October in Napa and Sonoma counties and lasted for more than 8 weeks. The fuels and perimeters of these wildfires can be found in our previous publications (Liang et al., 2021, 2022b). These wildfires caused multiple air quality measurement stations in the San Francisco Bay Area to record  $\text{PM}_{2.5}$  (particulate matter with diameter less than  $2.5 \mu\text{m}$ ) exceeding  $100 \mu\text{g m}^{-3}$  (Liang et al., 2021). These air masses contained fire emissions from mainly oak woodland land-

scapes, combined with suburban housing developments, and agricultural areas. Our SV-TAG measurements were conducted on the University of California (UC) Berkeley campus (37°52′24.5″ N, 122°15′48.2″ W), located in the urban San Francisco Bay Area, 55–65 km downwind of the major wildfires. SV-TAG data are reported for 10 to 19 October.

Data were also collected by the cTAG in rural Idaho between 12 and 27 August 2018. The McCall, Idaho, field site (44°52′18.5″ N, 116°06′55.7″ W) was a rural location approximately 4 km south of the town of McCall (population < 3500), which is located 180 km north of Boise and surrounded by national forest. This site experienced smoke from the following four fires: Rattlesnake Creek (~ 3300 ha; ~ 45 km northwest of McCall), Mesa (~ 14 000 ha; 38 km southwest of McCall), Rabbit Foot (~ 15 000 ha; ~ 145 km east of McCall), and a fire 60 km southwest of McCall and fires from central Oregon. More detailed descriptions of these fires can be found in Lindsay et al. (2022). Backward trajectory analysis by Lindsay et al. (2022) suggests that the smoke plumes traveled at least 3–5 h and up to 18–30 h prior to arriving at this site.

## 2.2 Speciated measurements by SV-TAG and cTAG

During the 2017 California wildfires, individual compounds were measured by the SV-TAG with in situ derivatization at hourly time resolution (Zhao et al., 2013b; Isaacman et al., 2014). Ambient air was pulled to the SV-TAG at a flow rate > 1000 L min<sup>-1</sup> through a 2 m × 6 in. (15.2 cm) outer diameter duct to reduce condensation of gas-phase SVOCs in the tubing. The SV-TAG subsampled 20 L min<sup>-1</sup> of air from the center of the airstream through passivated tubing and a 1.0 μm cutoff (PM<sub>1</sub>) cyclone. Then, 10 L min<sup>-1</sup> of air was delivered to each of the two identical coated metal mesh filter cells, with one of the air samples passing through a carbon denuder that removes gases. This approach enables us to determine the particle-phase fraction ( $F_p$ ) of each measured compound.

The same strategy was used for cTAG sampling in McCall, Idaho. The cTAG subsampled 10 L min<sup>-1</sup> of air from the middle of a 25 cm diameter duct through a PM<sub>1</sub> cyclone. The details of the cTAG can be found in Wernis et al. (2021). The cTAG measured volatile organic compounds (VOCs) and SVOCs concurrently, and this research focuses on the SVOCs. The main difference between the cTAG and the SV-TAG is that the cTAG alternated between gas plus particle and particle-only sampling by either having the sampled air passing through the denuder (same as the denuder for the SV-TAG) or bypassing the denuder. Since gas plus particle and particle-only organics were not collected simultaneously, we calculated the  $F_p$  value (2 h time resolution) of each compound using the interpolation method as follows:

$$F_{P,n} = \frac{C_{P,n}}{(C_{G+P,n-1} + C_{G+P,n+1})/2}, \quad (1)$$

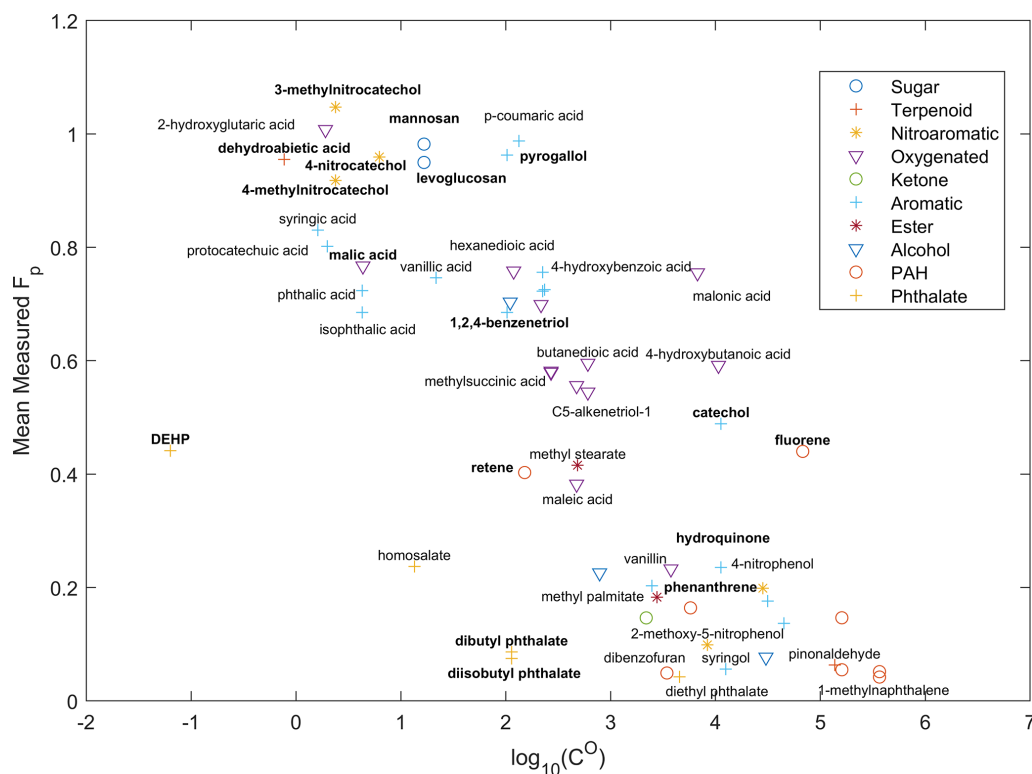
where  $C_{P,n}$  is the particle-phase concentration (proportional to the internal-standard-normalized signal) for sample  $n$ , and the denominator is the sum of gas plus particle concentrations in the previous hour and the subsequent hour (Zhao et al., 2013a). More materials and methods about SV-TAG and cTAG can be found in the Supplement.

## 2.3 Uncertainty in the measurements

The uncertainty in the SV-TAG measurements has been studied in Isaacman et al. (2014). Uncertainty in quantification of compounds was estimated to be 20 %–25 %. However, because  $F_p$  is the ratio of the signals from the two cells, the uncertainty of  $F_p$  only depends on how well samples from the two cells can be intercompared. We collected 11 bypass samples simultaneously on two cells as a direct comparison during this campaign to equalize the signal of the two cells. This equalization helps to reduce the cell-to-cell variabilities caused by differences in the collection efficiency, derivatization efficiency, etc. The uncertainty of  $F_p$  is estimated to be 15 %–25 % when this correction is applied, so a value of  $F_p$  greater than 1 is possible (Isaacman et al., 2014). For a particle-phase-dominant compound, we expect its  $F_p$  value to follow the normal distribution, with a mean value close to 1 (Isaacman et al., 2014). Another source of uncertainty comes from carryover from previous sample, which is found to be 2.6 % for a total ion chromatogram and 2.4 % for a median single ion chromatogram and independent of volatility (Kreisberg et al., 2014). We evaluated the bias caused by carryover in the Supplement. This level of carryover can create a noteworthy positive bias to the  $F_p$  value of compounds almost entirely in the gas phase ( $F_p$  less than 0.05). Those compounds are not considered in the activity coefficient analysis. The measurement uncertainty in the campaign average  $F_p$  value can be estimated from the error of the individual  $F_p$  value by the central limit theorem, which yields 1.4 %–2.4 % of uncertainty to the campaign average  $F_p$  value (in Fig. 1).

## 2.4 Supporting measurements and models

During the 2017 northern California wildfires, PM<sub>2.5</sub> filter samples ( $N = 74$ ) with 3 to 4 h time resolution were collected concurrently with SV-TAG measurements. The chemical composition of the filter samples were determined using two-dimensional gas chromatography coupled with high-resolution time-of-flight mass spectrometry (GC×GC HR-ToF-MS; Liang et al., 2021). Compounds identified from SV-TAG measurements were validated by matching with authentic standards and comparison with GC×GC measurements. More details can be found in the Supplement. In addition, the particle mass concentration was measured by a scanning mobility particle sizer (SMPS; TSI 1080 electrostatic classifier coupled with a TSI 3788 condensation particle counter; range = 10–600 nm, sampling time = 300 s; aerosol flow rate = 0.6 L min<sup>-1</sup>; sheath flow rate = 6 L min<sup>-1</sup>), as-



**Figure 1.** Campaign average  $F_p$  value during the October 2017 northern California wildfires measured by the SV-TAG, classified by functional groups, and plotted against their logarithmic saturation concentrations  $C^O$  (in  $\mu\text{g m}^{-3}$ ) at 298 K. Only a subset of the measured compounds are labeled. Compounds discussed in the text are in bold. Alkanes and mono-carboxylic acids are shown separately in Fig. S3.

suming a density of  $\text{PM}_1 = 1.1 \mu\text{g m}^{-3}$ , according to Pokhrel et al. (2021), and averaged to the same sampling time base as SV-TAG measurements. The mass of  $\text{PM}_1$  was also measured by a Grimm 11 A OPC (GRIMM Aerosol Technik) for a shorter period. The linear fit of the SMPS measurement against the GRIMM measurement gives a slope of 0.98 and normalized root mean square error of 26 %, which suggests that the density assumed is reasonable. The mass of OA was not measured in Berkeley. In the biomass burning emission, OA was found to contribute 77 %–99 % to total aerosol mass (Lim et al., 2019). We therefore assume that 90 % of  $\text{PM}_1$  is organic in the following analysis. This assumption introduces some uncertainty but did not affect the conclusion. More discussion about the influence of this assumption is in Sect. 3.

In the FIREX-AQ 2018 campaign in McCall, the bulk properties of non-refractory  $\text{PM}_{2.5}$  were determined by an aerosol chemical speciation monitor (ACSM), using thermal vaporization and electron impact ionization mass spectrometry (Ng et al., 2011). The ACSM data were also averaged to the cTAG measurement sampling times. ACSM data and meteorological data in McCall can be accessed from Yacovitch et al. (2022).

Observed  $F_p$  values were compared with the values predicted based on the organic aerosol mass and vapor pressure of each compound by an absorption equilibrium partitioning

model which assumes instantaneous equilibrium between the gas and particle phases (Pankow, 1994). With this assumption, the  $F_p$  value of compound  $i$  can be expressed as

$$F_{p,i} = \left(1 + \frac{\gamma_i C_i^O}{C_{\text{OA}}}\right)^{-1} = \left(1 + \frac{C_i^*}{C_{\text{OA}}}\right)^{-1}, \quad (2)$$

where  $C_{\text{OA}}$  is the concentration of organic aerosol,  $\gamma_i$  is the unitless activity coefficient of compound  $i$ ,  $C_i^O$  is the saturation mass concentration of pure compound  $i$ , and  $C_i^*$  is the effective saturation concentration of  $i$  (all concentrations are in  $\mu\text{g m}^{-3}$ ). The activity coefficient captures the non-ideal interactions between the compound with the aerosol mixture. A  $\gamma$  smaller than 1 means compound  $i$  is more likely to condense into the aerosol mixture than into pure compound  $i$ . A larger  $\gamma$  means that the compound is more likely to stay in the gas phase over this aerosol mixture, and very large  $\gamma$  ( $> 10$ ) may indicate phase separation (Liu et al., 2021; Donahue et al., 2011). The subcooled liquid vapor pressure of each compound, which is used to calculate  $C_i^O$ , was calculated by the SIMPOL and EVAPORATION models (Pankow and Asher, 2008; Compernelle et al., 2011). We used the MPBPWIN component (modified Grain method) in the Estimation Program Interface (EPI) Suite of the U.S. Environmental Protection Agency (US EPA, 2012) to estimate the vapor pressures of nitroaromatic compounds because the SIM-



POL model substantially overestimates the vapor pressures of these compounds in general (Bannan et al., 2017; Wania et al., 2017). There is evidence that vapor pressures of multifunctional compounds may not be very accurate when by calculated SIMPOL and other group contribution methods (Barley and McFiggans, 2010; Dang et al., 2019; O'Meara et al., 2014). We used these methods to estimate  $C^O$ , mainly because they are commonly used to describe the gas–particle partitioning of SVOCs. To better understand the influence of BBOA on the gas–particle partitioning of SVOCs, we modeled the activity coefficients of compounds above aerosol mixtures using the AIOMFAC model (Zuend et al., 2011). More details of these models are provided in the Supplement.

### 3 Results and discussion

#### 3.1 Average partitioning behavior of BBOA marker compounds

After quality assurance and quality control (Sect. S1 in the Supplement), we selected 89 compounds from the 2017 study and 30 compounds from the 2018 study for detailed analysis. The campaign average  $F_p$  values for each compound measured during the 2017 northern California fires are shown in Figs. 1 and S3 (alkanes and aliphatic acids), with the distribution of  $F_p$  values shown in Fig. S4. The overall trend is as expected; the gas–particle partitioning behavior is related to the volatility of compounds. We focus our discussion on commonly used biomass burning markers. In contrast to Xie et al. (2014), primary BB tracers levoglucosan and mannosan (Elias et al., 2001; Simoneit, 2002) were found to be almost entirely in the particle phase. The same result is observed in the McCall data (Fig. S6). The discrepancy between our results and Xie et al. (2014) is likely due to the much higher mass of BBOA and levoglucosan and the lower temperature in our field campaigns. In Xie et al. (2014), gas-phase levoglucosan is comparable to particle-phase levoglucosan only in samples with less than  $50 \text{ ng m}^{-3}$  of levoglucosan at high ambient temperatures. These primary BB compounds have much higher  $F_p$  values compared with other compounds with similar saturation vapor pressures. The high particle-phase fraction of them makes them less likely to react with atmospheric oxidants and decay and therefore makes them very good BB marker compounds for source apportionment studies (Donahue et al., 2012). Surprisingly, dehydroabietic acid, although having a very low-saturation vapor pressure over pure compound, has a higher fraction in the gas phase compared with levoglucosan and mannosan in FIREX-AQ 2018 samples (Fig. S5). Nevertheless, it was predominantly in the particle phase in the Napa study. Retene ( $\text{C}_{18}\text{H}_{18}$ ) is one of the most abundant PAHs emitted from BB (Jen et al., 2019; Liang et al., 2022a). In fresh BB emissions, where the level of OA is several hundreds of micrograms per cubic meter, retene is expected to be mainly in particle phase. In both campaigns, the average  $F_p$  value of retene was around

0.4–0.5, which suggests a substantial amount of retene was in the gas phase in the aged smoke plumes. This agrees with our results in the FIREX-AQ 2019 study, in which we found that the ratio of particle-phase PAHs to organic carbon decreased with increasing acetonitrile/furan ratio (with correlation coefficient  $r = -0.60$ ), which is an indicator of plume age (Liang et al., 2022a). The particle-phase PAHs, dominated by the diterpenoid-related ones such as retene, on average accounted for 7.6 % of quantified OA in the FIREX-AQ 2019 filter samples but only accounted for less than 0.3 % in the Berkeley filter samples collected simultaneously with the SV-TAG measurements (Liang et al., 2021, 2022a). Evaporation, in addition to the difference in biomass fuels, may help to explain this observed difference between fresh and aged BBOA. Given its low O/C ratio and high molecular weight, the evaporated retene (and other diterpenoid-related PAHs) may react with oxidants and form SOA with a high yield (Gentner et al., 2012; Jimenez et al., 2009). Figure 1 also shows that isomers with very similar structures and therefore similar estimated saturation concentrations may have very different gas–particle partitioning behaviors. Pyrogallol (1,2,3-benzenetriol) has much higher fractions in the particle phase than 1,2,4-benzenetriol. It is either because the actual vapor pressures of these isomers are very different or because pyrogallol has specific interactions with the aerosol phase that their isomers do not have. Such interactions can reduce the activity coefficients of these SVOCs. A large variation in the vapor pressures among isomers has been reported before (Dang et al., 2019). Catechol also shows a much higher  $F_p$  value than hydroquinone. However, catechol could come from the decomposition of particle-phase-dominant compounds in GC. For example, protocatechuic acid (without catalyst; at  $200^\circ\text{C}$ ) is shown to produce catechol when heated in water (in CAS SciFinder, <https://scifinder.cas.org/>, last access: 9 February 2023). Further studies on the thermal stability of aromatic compounds in thermal desorption systems are needed to elucidate the partitioning behavior of such compounds.

Secondary BBOA markers, 4-nitrocatechol, 3-methylnitrocatechol, and 4-methylnitrocatechol, were predominantly in the particle phase. The sum of their concentrations reached  $1.4 \mu\text{g m}^{-3}$  in the particle phase in the strongest wildfire plumes. The high  $F_p$  value and high concentration make them good BB SOA markers. Equilibrium absorption modeling using the vapor pressure estimated by the EPI Suite (US EPA, 2012) also predicted that these compounds stay mostly in the particle phase, while modeling using vapor pressure from the SIMPOL model predicted half of these compounds (by mass) would stay in the gas phase. The high  $F_p$  values of these compounds agree with the findings by Finewax et al. (2018) and Fredrickson et al. (2022), in which they found the group contribution methods overestimate the saturation vapor pressure of these compounds. Fredrickson et al. also reported that  $\text{NO}_3$ -derived nitrocatechols are less volatile than the OH-derived

nitrocatechols ( $C^* = 2.4$  vs.  $12 \mu\text{g m}^{-3}$ ) at 295 K, potentially due to the difference in the composition of the bulk SOA. However, there is no obvious diurnal trend in the  $F_p$  values of nitrocatechols in our study.

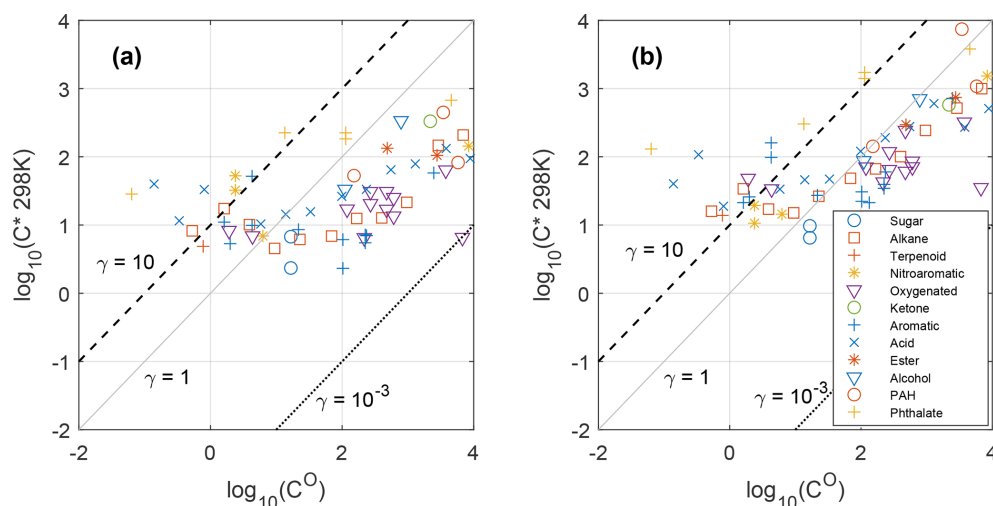
In many studies, such as Isaacman-VanWertz et al. (2016) and Nie et al. (2022), when predicting the gas–particle partitioning of organic compounds using Eq. (2),  $C^*$  is assumed to be the same as  $C^0$  ( $\gamma = 1$ ). To test how well the equilibrium absorptive partitioning model with the  $\gamma = 1$  assumption explains the partitioning behavior of individual compounds, we calculated the median  $C^*$  of each compound from measured  $F_p$  values using Eq. (2), converted them to 298 K values (see details about the conversion in the Supplement), and plotted them against the  $C^0$  over pure compound estimated by group contribution models (Fig. 2). Only compounds with  $C^0$  between  $10^{-2}$  and  $10^4 \mu\text{g m}^{-3}$  were considered here because the partitioning method may not work for very volatile or nonvolatile compounds (Stark et al., 2017). It is worth noting that the results in Fig. 2 are not very sensitive to the assumed fraction of OA in  $\text{PM}_{10}$  because  $C^*$  is proportional to  $C_{\text{OA}}$ . A 10 % change in  $C_{\text{OA}}$  will therefore only change  $\log_{10}(C^*)$  by  $\sim 0.04$ . As has been shown in our previous work, in October 2017, the level of organic carbon (OC) increased by  $\sim 6$  times during the wildfires, and the temporal variation in the levoglucosan correlated well with OC and thus BBOA (Liang et al., 2021). We therefore classified the data points into five concentration bins, based on the levels of particle-phase levoglucosan concentrations, which is a proxy for the BBOA influence (Table 1). The range of  $\gamma$  inferred from this study is similar to what Cappa et al. (2008) found for dicarboxylic acids in multicomponent mixtures. The  $\gamma$  inferred for adipic acid ( $\sim 0.3$ ) is also within the range of  $\gamma$  inferred for adipic acid in adipic acid–ambient extract mixtures (Saleh and Khlystov, 2009). Under both strong BBOA (high to extreme BBOA in Table 1) and weak BBOA (low to light BBOA in Table 1) scenarios, most of the data points fall between the lines of  $\gamma = 1$  and  $\gamma = 10^{-2}$ . The result suggests these compounds have higher  $F_p$  values than what is predicted by the models (if  $\gamma = 1$  is assumed). A few exceptions include diethylhexyl phthalate (DEHP), dibutyl phthalate, diisobutyl phthalate, and homosalate, which were found to be more volatile than expected. The low  $F_p$  value of DEHP is likely related to DEHP in coarse particles (Ma et al., 2014), which is not measured in this study. However, this reason cannot fully explain the order of magnitudes deviation of measured  $C^*$  from prediction. Many oxygenated compounds with at least two  $-\text{OH}$  groups were found to be BB SOA in these fire plumes in our previous study (Liang et al., 2021). In contrast to the GoAmazon wet season study in which they found that the  $F_p$  values of these compounds do not correlate with the vapor pressure (Isaacman-VanWertz et al., 2016), we saw that  $F_p$  and  $C^*$  values of these compounds qualitatively follow the equilibrium absorptive partitioning models (Figs. 1 and 2), though the dependence on  $C^0$  is not as strong as alkanes and carboxylic acids (Fig. S3). This is likely re-

lated to high concentration of organic aerosol (mainly from biomass burning; average estimated OA =  $20.1 \mu\text{g m}^{-3}$ ), the dominance of organic aerosol in biomass burning emissions (Lim et al., 2019; Fine et al., 2004), and relatively low average RH (47 %) in this study compared with the GoAmazon study (wet season average OA =  $1.2 \mu\text{g m}^{-3}$  with various sources; RH = 90 %; De Sá et al., 2018; Isaacman-VanWertz et al., 2016). SOA formation catalyzed by inorganic compounds such as sulfate and water, which was observed in the GoAmazon study (Yee et al., 2020), was probably less important during the wildfire events in this study.

Comparing Fig. 2a and b, we found that the gas–particle partitioning behaviors of compounds are better described by the model in the strong BB case, especially for compounds with  $\log_{10}C^0$  between 1 and 3 (data points are closer to the  $\gamma = 1$  line; also shown in Fig. S7), which is probably due to the dominance of OA in the strong BB case. It is also worth noting that the deviation from the  $\gamma = 1$  line can be partly attributed to the inaccuracy in  $C^0$  estimated by the group contribution models. When this area is not affected by wildfire smoke, inorganic aerosol accounts for a larger fraction of total aerosol mass (Shah et al., 2018). Inorganic compounds, liquid water in aerosol, and black carbon can therefore have stronger effects on the partitioning of the SVOCs in the low BB case. Also, the equilibration timescale of SVOCs between the gas and particle phases increases with decreasing mass of PM (Shiraiwa and Seinfeld, 2012). That can also limit the capability of the equilibrium absorption model in predicting the partitioning of many BB SVOCs under relatively clean scenarios. In addition, although  $\log_{10}C^*$  has a roughly linear relationship with  $\log_{10}C^0$ , the slope is only 0.35, even under the strong BB scenario. This nonideal behavior is in agreement with the trend observed in multicomponent temperature-programmed desorption experiments in which researchers found that larger dicarboxylic acids have higher  $\gamma$  than smaller ones (Cappa et al., 2008). Together, these results suggest that the inclusion of activity coefficients is needed to better predict the partitioning behaviors of individual organic compounds.

### 3.2 Temporal variations of $F_p$ values and factors influencing $F_p$ values of individual compounds

To explore the temporal variations in the individual compounds, we first plotted the correlation matrix of the  $F_p$  time series. The  $F_p$  values of lighter alkanes ( $\text{C}_{13}$ – $\text{C}_{20}$ ) correlate well with each other, while semivolatile alkanes ( $\text{C}_{22}$ – $\text{C}_{26}$ ) form a separate group (Fig. S8). This is probably because the lighter alkanes are so volatile that they do not respond to the levels of ambient aerosol like the semivolatile compounds do. As shown in Fig. 3, the  $F_p$  values of most PAHs have an anticorrelation with the  $F_p$  values of oxygenated compounds. Similar trends are observed in the FIREX-AQ 2018 study (Fig. S9), in which the partitioning behaviors of oxygenated compounds were not well-correlated with compounds with-



**Figure 2.** The median effective saturation concentration  $C^*$  (converted to 298 K) of each compound derived from Eq. (2) during the October 2017 northern California wildfires vs. the estimated saturation concentration over pure compound  $C^O$  at 298 K on a logarithmic scale. Lines show the theoretical  $C^*$  as a function of  $C^O$ . (a) Weak BBOA scenario. (b) Strong BBOA scenario.  $C^*$  and  $C^O$  are in micrograms per cubic meter.

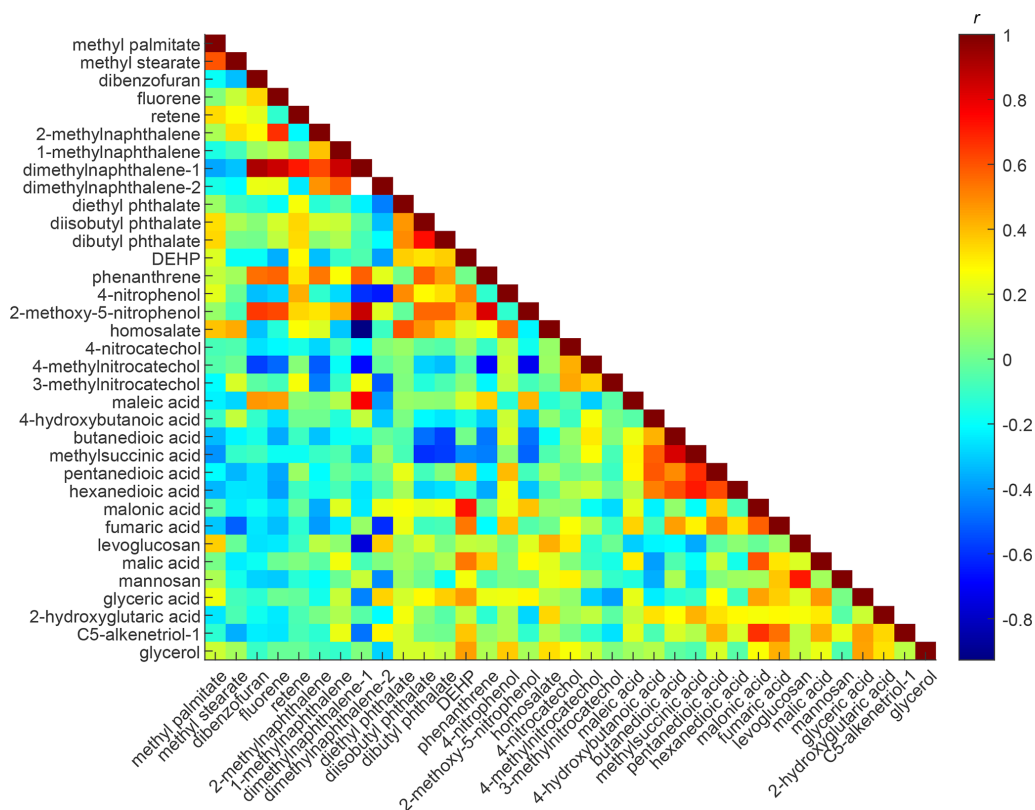
**Table 1.** Mean and standard deviation of concentrations of levoglucosan,  $PM_{10}$ , and levels of temperature and RH during different levels of BBOA influence in Berkeley during the 2017 northern California wildfires study.

| Level      | Percentiles of particle-phase levoglucosan | Mean $\pm$ SD levoglucosan ( $\mu\text{g m}^{-3}$ ) | Mean $\pm$ SD $PM_{10}$ ( $\mu\text{g m}^{-3}$ ) | Mean $\pm$ SD temperature $^{\circ}\text{C}$ | Mean $\pm$ SD RH  |
|------------|--|---|--|--|-------------------|
| Low BB     | 0 %–40 %                                   | $0.03 \pm 0.02$                                     | $4.7 \pm 2.4$                                    | $17.5 \pm 3.8$                               | $49 \% \pm 22 \%$ |
| Light BB   | 40 %–60 %                                  | $0.18 \pm 0.07$                                     | $12.0 \pm 4.6$                                   | $15.8 \pm 2.9$                               | $44 \% \pm 20 \%$ |
| Medium BB  | 60 %–75 %                                  | $0.72 \pm 0.21$                                     | $24.9 \pm 9.0$                                   | $16.0 \pm 1.5$                               | $45 \% \pm 24 \%$ |
| High BB    | 75 %–90 %                                  | $1.36 \pm 0.21$                                     | $36.1 \pm 18.3$                                  | $16.3 \pm 2.2$                               | $36 \% \pm 22 \%$ |
| Extreme BB | 90 %–100 %                                 | $2.61 \pm 0.43$                                     | $56.3 \pm 21.0$                                  | $14.8 \pm 1.5$                               | $45 \% \pm 20 \%$ |

out  $-\text{OH}$  groups. This is likely related to the varying ambient aerosol composition. Whether preexisting organic aerosol can alter the partitioning behavior of SVOCs depends on the aerosol composition because different components of organic aerosol are not always miscible (Ye et al., 2016; Mahrt et al., 2022a, b). Relatively nonpolar compounds in the ambient aerosol may change the OA matrix, thus enhancing the absorption of nonpolar into the particle phase, while they hardly enhance the condensation of polar compounds. In addition, the  $F_p$  values of methyl palmitate and methyl stearate correlated very well with each other, but their  $F_p$  values correlated poorly with the  $F_p$  values of other compounds (Fig. 3). Methyl palmitate and methyl stearate are emitted from cooking (Kristensen et al., 2019). Their  $F_p$  values were elevated (making their  $\gamma$  closer to 1) in the evening, and during breakfast or lunch hours (Fig. S10), which suggests the cooking aerosol enhanced the absorption of fatty acid esters into the particle phase.

We further explored the importance of environmental factors on the  $F_p$  values of each compound using a random forest model implemented in MATLAB R2022a. Details of

the model can be found in the Supplement. The random forest algorithm is an ensemble approach that makes a prediction of the response ( $F_p$  in this study), with predictors (measured variables such as temperature) using the aggregation of multiple decision trees. In the 2017 northern California fires, constrained by the availability of data, we only used time of day, RH, temperature,  $PM_{10}$  and particle-phase levoglucosan concentration as predictors, and the  $F_p$  value of the compounds as the response. Possibly due to the lack of predictors, we were only able to achieve  $R^2 > 0.45$  for predicting the  $F_p$  value of out-of-bag data (data points not used in individual regression trees) of 4 compounds (fluorene, 4-nitrophenol, 1,2,4-benzenetriol, and pentanedioic acid) out of 89 compounds considered. The importance of predictors was estimated by permutation of out-of-bag predictor observations (Fig. S11). For fluorene, the levoglucosan concentration is the dominant predictor with a negative relationship. For other compounds,  $PM_{10}$  is the most important predictor (with a positive relationship). We also applied this model on the  $F_p$  value measured in the FIREX-AQ 2018 study in which more predictors are available. Out of 30 compounds consid-



**Figure 3.** Correlation matrix of  $F_p$  time series for selected compounds observed by SV-TAG in Berkeley, California, between 10 and 19 October 2017. Compounds are ordered by the number of  $-OH$  and  $-COOH$  groups. The white pixel indicates that the  $F_p$  values of those two compounds were both available in fewer than three samples.

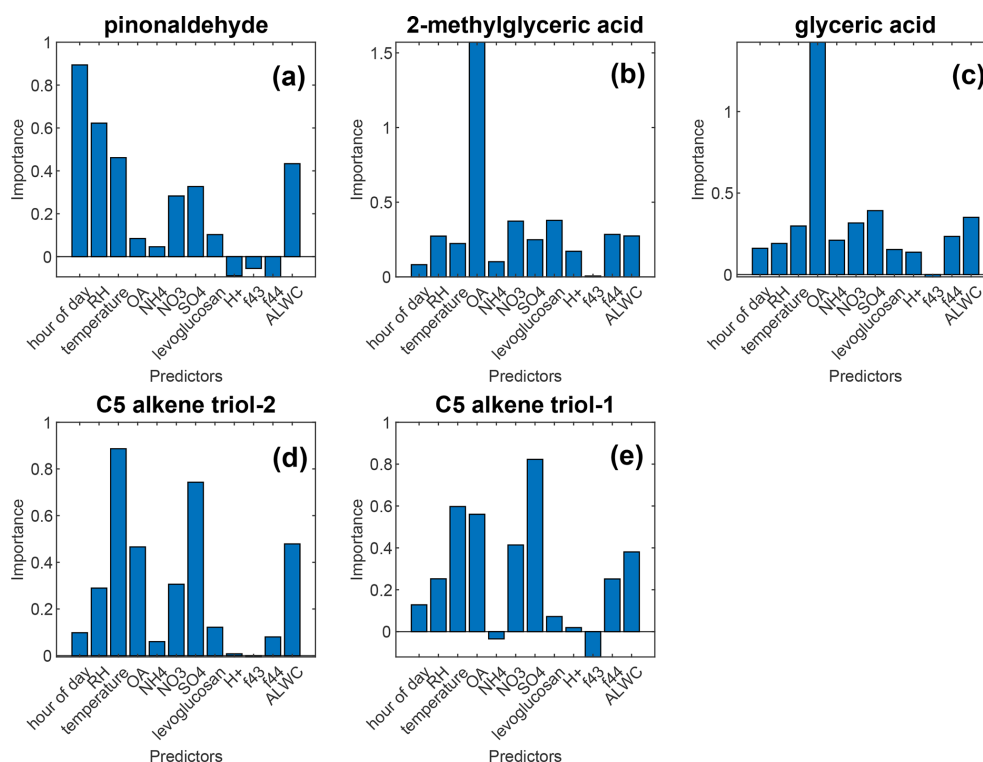
ered, 5 of them were predicted with  $R^2 > 0.45$ . As shown in Fig. 4, the level of OA is the dominant predictor of the  $F_p$  values for 2-methylglyceric acid and glyceric acid, while for the two C<sub>5</sub> alkene triols, temperature and sulfate concentration become the most important factors. As temperature increases, their  $F_p$  value decreases ( $r = -0.53$  between the  $F_p$  value of C<sub>5</sub> alkene triol-1 and temperature and  $-0.67$  for C<sub>5</sub> alkene triol-2). The positive association of sulfate with the  $F_p$  value of pinonaldehyde and C<sub>5</sub> alkene triols are consistent with the catalytic roles of sulfate and water in the formation of these compounds (Yee et al., 2020; Jenkin et al., 2000). Nevertheless, the campaign-averaged particle-phase OA was  $13 \mu\text{g m}^{-3}$ , while the campaign-averaged ALWC was only  $1.5 \mu\text{g m}^{-3}$ , indicating that dissolution in aerosol liquid water was not the dominant gas–particle partitioning mechanism for most compounds considered in this study.

### 3.3 Effect of BBOA on gas–particle partitioning of SVOCs

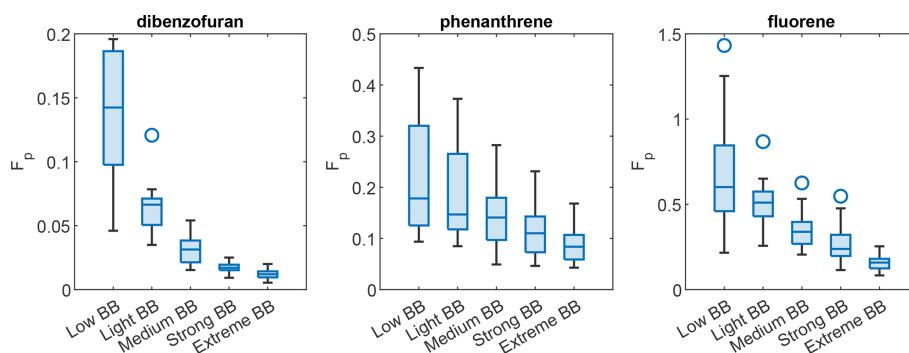
When BBOA dominated the ambient aerosol, the gas–particle partitioning of many compounds was different from

clean periods. But such changes are not all in the same direction. Figure 5 shows that the  $F_p$  value of nonpolar compounds decreased with the level of wildfire smoke influence during the 2017 northern California wildfires. The  $F_p$  value of dimethylnaphthalene (isomer 1) and 2-methylnaphthalene also decreases with increasing BBOA ( $p = 0.021$  and  $0.026$ , respectively, for the linear fit between mean BBOA percentile in each bin and mean  $F_p$  value in each bin), although the changes are smaller. Conversely, levoglucosan, mannosan, 4-nitrocatechol, and hexanedioic acid (adipic acid), each with at least two  $-OH$  groups, did not follow this trend (Fig. S12). Environmental factors such as temperature and RH remained very similar at different BB influence levels in Berkeley (Table 1). They are therefore unlikely major causes for this trend. The high O : C ratios of the BBOA and the low O : C ratios of these nonpolar compounds, which makes them not miscible, is a more plausible cause of the decrease in their  $F_p$  value during the wildfires. We also compared the  $F_p$  value of these compounds in different ranges of PM concentrations, and a slightly weaker though still clear dependence of the  $F_p$  value on the PM concentration is observed for phenanthrene and fluorene (Fig. S13). The slightly weaker dependence of the  $F_p$  value on PM than on levoglucosan also suggests that BBOA is different from local PM and that it caused the de-





**Figure 4.** The importance scores of factors on the  $F_p$  value of selected compounds, measured in the FIREX-AQ 2018 study. A high-importance score means that the factor is important. Importance scores close to zero and negative importance scores suggest low importance. Full descriptions of these predictors can be found in the Supplement.



**Figure 5.** Box plots showing the effect of BBOA on the particle-phase fraction on some nonpolar compounds measured in the 2017 northern California wildfire study. Each box plot shows the interquartile range, with whiskers extended to  $1.5 \times$  the interquartile range. Central horizontal lines show the medians. Circles denote outliers.

crease in the  $F_p$  value of these nonpolar compounds. Different effects of BBOA on gas–particle partitioning behaviors on relatively polar and relatively nonpolar compounds are also seen in the FIREX-AQ 2018 study. The median  $F_p$  value of the polar compounds increased almost monotonically from the low BB bin to the high BB bin, while the  $F_p$  value of the less polar compounds did not (Fig. S14). Nevertheless, we did not see the  $F_p$  value of these nonpolar compounds decrease monotonically when BB influence increased. It is probably because the aerosol in McCall, Idaho,

did not have many contributions from urban sources such as traffic and cooking emissions, and the background aerosol (when BB influence was low) had similar compositions with the BBOA that impacted McCall. This is supported by the narrow distribution of  $f_{44}$  (the fraction of the OA mass spectrum signal at  $m/z$  44) measured by the ACSM, and therefore also the O/C ratio (Aiken et al., 2008), and the lack of statistically significant difference of  $f_{44}$  under different BB influence levels (Fig. S15). Thus, the activity coefficients of

these nonpolar compounds did not change as much as in the 2017 California study.

In the 2017 California study, from the low BB scenario to the high BB scenario, the level of  $PM_{10}$  and therefore  $C_{OA}$  increases by 10 times ( $C_{OA}$  might have increased by more than 10 times because a smaller fraction of local  $PM_{10}$  is OA). According to Eq. (2), to make the  $F_p$  value smaller,  $\gamma$  must increase by more than 10 times. We simulated the  $\gamma$  of these nonpolar compounds using the AIOMFAC model (in the Supplement). The model successfully predicted that  $\gamma$  of these compound increases was the fraction of the BBOA in the total ambient aerosol increases. However, the increase was predicted to be approximately a factor of 2 only (Fig. S16). It is worth noting that the simulations were based on the assumed composition of aerosol because many properties of the aerosol (e.g., bulk composition and liquid water content) were not measured in Berkeley, which introduced uncertainty to the simulations.

The high viscosity of BBOA may contribute to the  $\gamma$  of these compounds increasing with fraction of BB influence. Under the RH and temperature in Berkeley during these fires, the BBOA is expected to be semisolid (Li et al., 2020). This can affect the evaporation and condensation of the SVOCs (Zelenyuk et al., 2012). However, the ages of the plumes are at least 2 h (Liang et al., 2022b), which are long enough for these nonpolar compounds to reach gas–particle partitioning, even when the particles are mainly in a semisolid state (Shiraiwa and Seinfeld, 2012). If the effect of viscosity is important here, then the compounds mainly emitted in the particle phase from wildfires would have  $F_p$  values higher than what is predicted from the equilibrium absorptive model, while compounds later condensed into the particle phase (such as SVOCs mainly emitted in the urban area, including phthalates and fatty acid esters) would have  $F_p$  values lower than prediction when BBOA dominates. However, we did not find such a trend, which may suggest the effect of viscosity on  $F_p$  value is overshadowed by other factors.

The observed partitioning phenomenon of BBOA has at least two interesting implications. First, the high gas-phase fraction of the nonpolar compounds can reduce their atmospheric lifetimes. For instance, the gas-phase lifetimes of fluorene and phenanthrene are 21 and 9 h under a typical atmospheric environment ( $[OH] = 10^6 \text{ molecules cm}^{-3}$ ,  $[O_3] = 50 \text{ ppb}$ ), respectively (Keyte et al., 2013). However, these compounds have much longer lifetimes in the particle phase because OA can shield them against heterogeneous oxidation and evaporation (Zelenyuk et al., 2012). The low-particle-phase fraction of these compounds in the BB plumes can shorten their lifetimes (May et al., 2012), thus reducing their long-range transport. Nonetheless, the gas-phase PAHs can react with oxidants to form oxygenated and nitrated PAHs, which may have higher toxicity than the parent PAHs (Idowu et al., 2019). Also, the result implies that PAHs, which have urban sources, do not necessarily partition more to the condensed phase when the urban area is affected

by wildfire smoke. The phase distribution of compounds directly affects where they will deposit in the respiratory tract, what kind of cells will be exposed to these compounds, and, consequently, the health impacts (Liu et al., 2017). However, the PAHs observed in this research are not as toxic as heavy PAHs such as benzopyrenes. Future work on the gas–particle partitioning behaviors of more toxic PAHs is still needed.

## 4 Conclusions

In this work, we evaluated the measured phase partitioning behaviors of various BB markers. Levoglucosan, mannosan, and nitrocatechols were found to stay mainly in the particle phase, even when the ambient OA was relatively low, which suggests that they are good marker compounds for source apportionment. The phase partitioning of SVOCs is observed to be strongly dependent on the saturation vapor pressure, yet despite whether BBOA is abundant, the equilibrium absorption model underpredicts the particle-phase fraction of the majority of the compounds measured. BBOA enhanced the condensation of polar compounds into the particle phase but not some nonpolar compounds, such as PAHs.

Although we were able to qualitatively find out the factors influencing the gas–particle partitioning of compounds, with the current dataset, even using machine learning algorithms, we were not able to accurately predict the partitioning behavior of the majority of compounds. Future longer-term measurements of gas-phase and particle-phase SVOCs simultaneously with measurements of environmental factors and bulk aerosol composition may help to further address this problem.

**Data availability.** Data used in this research are available from the corresponding authors upon request.

**Supplement.** The supplement related to this article is available online at: <https://doi.org/10.5194/acp-23-12441-2023-supplement>.

**Author contributions.** YL, RAW, NMK, and AHG designed the research. YL, RAW, KK, NMK, PLC, and SCH performed the research. YL, RAW, AWHC, NLN, and AHG analyzed the data. YL and AHG wrote the paper, with suggestions from all coauthors.

**Competing interests.** At least one of the (co-)authors is a member of the editorial board of *Atmospheric Chemistry and Physics*. The peer-review process was guided by an independent editor, and the authors also have no other competing interests to declare.

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