Gas/Particle Partitioning of Semivolatile Organic Compounds To Model Inorganic, Organic, and Ambient Smog Aerosols

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Gas/particle (G/P) partitioning is an important process that affects the deposition, chemical reactions, long-range transport, and impact on human and ecosystem health of atmospheric semivolatile organic compounds (SOCs). Gas/particle partitioning coefficients \( K_p \) were measured in an outdoor chamber for a group of polynuclear aromatic hydrocarbons (PAHs) and \( n \)-alkanes sorbing to three types of model aerosol materials: solid ammonium sulfate, liquid dioctyl phthalate (DOP), and secondary organic aerosol (SOA) generated from the photooxidation of whole gasoline vapor. \( K_p \) values were also measured for ambient \( n \)-alkanes sorbing to urban particulate material (UPM) during summer smog episodes in the Los Angeles metropolitan area. Based on the \( K_p \) values obtained for the aerosols studied here, for environmental tobacco smoke (ETS), and for a quartz surface, we conclude that G/P partitioning of SOCs to UPM during summer smog episodes is dominated by absorption into the organic fraction in the aerosol. Comparisons of the partitioning of SOCs to three different types of aerosols demonstrate that (1) DOP aerosol may be a good surrogate for ambient aerosol that consists mainly of organic compounds from primary emissions; (2) ETS particles may be a good surrogate for SOA; and (3) the sorption properties of ambient smog aerosol and the chamber-generated SOA from gasoline are very similar. The similarities observed between ambient smog aerosol and chamber-generated SOA from gasoline support the use of \( K_p \) values for ambient aerosol that consists mainly of amorphous organic carbon. The temperature dependence of \( K_p \) for a given compound to UPM generally contains a significant amount of amorphous organic carbon. The \( K_p \) values that were measured for PAHs partitioning to three types of aerosols demonstrate that (1) DOP aerosol may be a good surrogate for ambient aerosol that consists mainly of organic compounds from primary emissions; (2) ETS particles may be a good surrogate for SOA; and (3) the sorption properties of ambient smog aerosol and the chamber-generated SOA from gasoline are very similar. The similarities observed between ambient smog aerosol and chamber-generated SOA from gasoline support the use of \( K_p \) values for ambient aerosol that consists mainly of amorphous organic carbon.

Introduction

Gas/particle (G/P) partitioning is an important process that affects the deposition, chemical reactions, long-range transport, and human and ecosystem health effects of atmospheric semivolatile organic compounds (SOCs). Much of the thinking about the G/P partitioning of SOCs has involved the assumption that the partitioning process involves simple physical adsorption (1–3). While this sorption mechanism will be important when atmospheric particulate material is comprised solely of mineral materials, urban particulate material (UPM) generally contains a significant amount of amorphous organic carbon (4, 5). Thus, it seems likely that absorptive partitioning must be at some role in urban air and also in air affected by urban sources. An equation that has been used successfully to parameterize G/P partitioning is (2, 6, 7)

\[
K_p = \frac{F/TSP}{A}
\]

where \( K_p \) (m³/µg) is the G/P partitioning coefficient for a given compound; \( F \) and \( A \) are the P- and G-phase concentrations (ng/m³) of the compound, respectively; and TSP (µg/m³) is the total suspended particulate material concentration. When both adsorptive and absorptive partitioning are operative, \( K_p \) can be parameterized as (8)

\[
K_p = \frac{C_1 + C_2}{p_c^r}
\]

where \( p_c \) (torr) is the pure compound subcooled liquid vapor pressure. The terms \( C_1/p_c^r \) and \( C_2/p_c^r \) represent the adsorptive and absorptive contributions to \( K_p \), respectively. Detailed theoretical expressions for \( C_1 \) and \( C_2 \) are given by Pankow (8). Within a given compound class (e.g. polycyclic aromatic hydrocarbons (PAHs), \( n \)-alkanes, etc.), \( C_1 \) and \( C_2 \) will be only weakly compound dependent, and \( \log (F/TSP)/A \) values will tend to be correlated with \( \log p_c^r \), according to

\[
\log K_p = m_1 \log p_c^r + b_1
\]

where \( m_1 \) is usually near 1. The temperature dependence of \( K_p \) for a given compound sorbing to a given type of atmospheric particulate material can be expressed as (2, 8)

\[
\log K_p = \frac{m_2}{T} + b_0
\]

Most of the temperature dependence explicitly present in eq 4 is implicitly present in eq 3 due to the temperature dependence of \( p_c^r \). Based on a comparison of organic matter-phase-normalized \( K_p \) values that were measured for PAHs partitioning to environmental tobacco smoke (ETS) and estimated for partitioning to UPM, Liang and Pankow (9) have suggested that absorption may frequently be the dominant G/P partitioning mechanism for SOCs in urban air. In addition, Odum et al. (10) recently showed from smog chamber studies that the condensation of the photooxidation products which form secondary organic aerosol (SOA) is very well described by an absorptive G/P partitioning model. In this paper, we discuss smog chamber experiments that were performed to examine the partitioning of two classes of SOCs (PAHs and \( n \)-alkanes) to three different model aerosol materials: solid ammonium sulfate; nonpolar, liquid dioctyl phthalate (DOP); and polar, amorphous SOA generated from the photooxidation of whole gasoline vapor. G/P partitioning was also measured for UPM collected during two heavy smog episodes in the Los Angeles metropolitan area at a sampling site in the city of Pasadena. The UPM data were then compared to the data for the model aerosols to obtain information about both the operative G/P partitioning mechanism during summer smog episodes and the nature of UPM.

Experimental Section

Smog Chamber Experiments. Five experiments were conducted in the Caltech 60-m³ flexible Teflon smog chamber.
described in detail elsewhere (10–12). Prior to each experiment, the chamber was flushed with 4–5 bag volumes of purified compressed air while being allowed to warm to 31–38 °C under sunlight for at least 12 h. The purified compressed air was obtained by passing laboratory compressed air through four consecutive packed sorbent beds that contained (in order) Purafil, Drierite, 13X molecular sieve, and activated charcoal. After purification but before entering the chamber, the air was reheumidified to a relative humidity (RH) of approximately 10% for the chamber temperature of 31–36 °C using distilled/deionized water. The warm temperatures prevented the use of higher RH values because such use would have led to condensation in the sample lines between the chamber and the instrument.

Solid, inorganic aerosol was generated by atomizing an aqueous solution of 5 g/L ammonium sulfate using a stainless steel, constant rate atomizer (10). Before entering the Teflon chamber, the aerosol was passed via a heated copper tubing first through a diffusional dryer packed with indicating silica gel and then through a 85Kr charge neutralizer. The initial number concentration in the chamber was ∼7 × 107/cm3. The geometric mean diameter was 111 ± 1.5 nm.

Liquid DOP aerosol was generated by atomizing a 0.005% by weight solution of DOP in pentane using a stainless steel, constant rate atomizer. Before entering the Teflon chamber, the DOP aerosol was passed through a diffusional dryer packed with activated charcoal to remove pentane, then through a 85Kr charge neutralizer. The initial number concentration in the chamber was ∼107/cm3. The geometric mean diameter was 480 ± 1.6 nm.

Amorphous SOA was generated by first adding seed (NH4)2SO4(s) particles at a concentration in the chamber of 107/cm3 (total aerosol volume ∼20 µm3/cm3). These particles served as condensation nuclei. The chamber was then covered with a black tarpaulin so that photochemical reactions would not begin until all reactants had been added and given time to become well mixed. This was followed by injections of propene, NO, and NO2 into the chamber to obtain initial concentrations of 300, 640, and 340 ppbv, respectively. These chemicals lead to the generation of sufficient hydroxyl radicals and ozone to sustain the subsequent photochemical reactions. SOA was generated using “industry-average” gasoline obtained from the Auto/Oil Air Quality Research Improvement Program (13). A total of 700 µL of the gasoline was completely volatilized in a glass dilution bulb that was gently heated on a hot plate to allow G/P equilibrium to be reached. G- and P-phase organic compounds were collected through a copper tube (i.d. = 0.5 in.) that extended ∼30 cm into the chamber. Sampling event durations and conditions are listed in Table 1. The sampler utilized a 102 mm diameter glass fiber filter (GFF) (with an identical backup filter) to collect P-phase organic compounds. The data obtained with the backup filter were used to correct for the adsorption of G-phase SOCs to the front filter as described for quartz filters by Hart and Pankow (15). (On average, the mass of a compound of interest found on the backup GFF was ∼15% of that on the front GFF; no compound dependence was observed in this number.) Following the filters, two parallel trains were used to collect G-phase compounds. One train employed two sequential adsorption/thermal desorption (ATD) cartridges containing ∼1 g each of Tenax-GC to collect the more volatile compounds. The second train employed two sequential 198 cm3 polyurethane foam (PUF) plugs to collect the less volatile compounds. The GFFs were precleaned by baking at 370 °C overnight and then held at room temperature in a desiccator containing indicating silica gel prior to sampling. The PUF plugs were Soxhlet-extracted with methylene chloride for 24 h before use. Each ATD cartridge was cleaned by pumping 1 L of 1:1 hexane:acetone through the cartridge at 2 mL/min, purged with ultra-pure helium at 50 °C for 20 min to remove the solvents, then conditioned at 300 °C for 2 h with a 1 mL/min flow of ultra-pure helium. During sampling, the flow rates were: filters, 70.8; Tenax-GC train, 0.324; and PUF train, 70.8 L/min, respectively. The face velocity for the filters was 18.5 cm/s.

The TSP was measured for each experiment by collecting particles with a 47-mm Teflon-coated GFF (tc-GFF) at a flow rate of 30 L/min for 20 min. Before use, each tc-GFF was Soxhlet-extracted with methylene chloride for 12 h, air-dried at room temperature, and then kept in a Petri dish prior to sampling. The tc-GFFs were weighed both before and after sampling. In each case, prior to weighing, the tc-GFFs were held for 1 h at 23 °C and 40% RH.

Complete aerosol number and size distribution measurements were recorded for every smog chamber experiment with a 1-min sampling frequency using a TSI (St. Paul, MN) Model 3071 cylindrical scanning electrical mobility spectrometer (SEMS) equipped with a TSI Model 3760 condensation nucleus counter. This equipment was housed in an enclosed cart maintained at 25 °C and located directly adjacent to the chamber. The SEMS system was operated with sheath flows of 0.25 L/min to allow for the measurement of aerosol flow rates of 0.25 L/min to allow for the measurement of aerosol size distributions in the 30–850 nm range. The aerosol size distribution data were used to provide independent measures of the TSP values.

Ambient Aerosol Sampling. Low-volume sampling was performed on the roof of Keck Laboratory during two separate smog episodes. The same equipment and filter type employed to sample the smog chamber were used. The sampling period for both episodes was 12 PM to 4 PM. The average RH for
the two days was 42%.

Other sampling information is given in Table 1. The GFFs were weighed before and after sampling as with the teflon-GFFs in the chamber experiments.

A single high-volume aerosol sample for carbon analysis and specific surface area measurement was taken at the same location from 12 PM to 4 PM during a third smog episode. A single high-volume aerosol sample for carbon analysis and specific surface area measurements was taken at the same location from 12 PM to 4 PM during a third smog episode.

TABLE 2. Measured Log \( K_p \) Values for Different Types of Aerosols

<table>
<thead>
<tr>
<th>compound</th>
<th>( \log p^* ) (torr)(^a)</th>
<th>( \log K_p ) (m(^3)/µg)</th>
</tr>
</thead>
<tbody>
<tr>
<td>((NH_4)_2SO_4(s), Chamber (32 \degree C and 10% RH))</td>
<td></td>
<td></td>
</tr>
<tr>
<td>heptadecane C(_{17})</td>
<td>−3.07</td>
<td>4.48</td>
</tr>
<tr>
<td>octadecane C(_{18})</td>
<td>−3.57</td>
<td>4.06</td>
</tr>
<tr>
<td>nonadecane C(_{19})</td>
<td>−4.06</td>
<td>3.88</td>
</tr>
<tr>
<td>eicosane C(_{20})</td>
<td>−4.55</td>
<td>3.58</td>
</tr>
<tr>
<td>heneicosane C(_{21})</td>
<td>−5.04</td>
<td>2.76</td>
</tr>
<tr>
<td>docosane C(_{22})</td>
<td>−5.53</td>
<td>2.30</td>
</tr>
<tr>
<td>tetracosane C(_{24})</td>
<td>−6.51</td>
<td>1.23</td>
</tr>
<tr>
<td>fluorene</td>
<td>−1.91</td>
<td>5.48</td>
</tr>
<tr>
<td>phanthenrene</td>
<td>−2.74</td>
<td>4.46</td>
</tr>
<tr>
<td>anthracene</td>
<td>−2.76</td>
<td>4.52</td>
</tr>
<tr>
<td>fluoranthene</td>
<td>−3.87</td>
<td>3.31</td>
</tr>
<tr>
<td>pyrene</td>
<td>−4.05</td>
<td>3.27</td>
</tr>
</tbody>
</table>

**Ambient Smog.** Ambient (37 \degree C and 42% RH)

<table>
<thead>
<tr>
<th>compound</th>
<th>( \log p^* ) (torr)(^a)</th>
<th>( \log K_p ) (m(^3)/µg)</th>
</tr>
</thead>
<tbody>
<tr>
<td>octadecane C(_{18})</td>
<td>−3.31</td>
<td>5.07</td>
</tr>
<tr>
<td>eicosane C(_{20})</td>
<td>−4.23</td>
<td>3.36</td>
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<td>heneicosane C(_{21})</td>
<td>−4.71</td>
<td>4.01</td>
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<tr>
<td>docosane C(_{22})</td>
<td>−5.18</td>
<td>3.55</td>
</tr>
<tr>
<td>tetracosane C(_{24})</td>
<td>−6.13</td>
<td>2.11</td>
</tr>
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</table>

**DOP.** Chamber (36 \degree C and 10% RH)

<table>
<thead>
<tr>
<th>compound</th>
<th>( \log p^* ) (torr)(^a)</th>
<th>( \log K_p ) (m(^3)/µg)</th>
</tr>
</thead>
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<tr>
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<tr>
<td>tetracosane C(_{24})</td>
<td>−6.30</td>
<td>1.24</td>
</tr>
<tr>
<td>fluorene</td>
<td>−1.80</td>
<td>4.95</td>
</tr>
<tr>
<td>phanthenrene</td>
<td>−2.62</td>
<td>4.23</td>
</tr>
<tr>
<td>anthracene</td>
<td>−2.65</td>
<td>4.31</td>
</tr>
<tr>
<td>fluoranthene</td>
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<td>2.82</td>
</tr>
<tr>
<td>pyrene</td>
<td>−3.92</td>
<td>2.80</td>
</tr>
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**Gasoline SOA.** Chamber (31 \degree C and 10% RH)

<table>
<thead>
<tr>
<th>compound</th>
<th>( \log p^* ) (torr)(^a)</th>
<th>( \log K_p ) (m(^3)/µg)</th>
</tr>
</thead>
<tbody>
<tr>
<td>hexadecane C(_{16})</td>
<td>−2.63</td>
<td>5.53</td>
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<tr>
<td>heptadecane C(_{17})</td>
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<td>4.79</td>
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<tr>
<td>octadecane C(_{18})</td>
<td>−3.62</td>
<td>4.36</td>
</tr>
<tr>
<td>nonadecane C(_{19})</td>
<td>−4.11</td>
<td>3.71</td>
</tr>
<tr>
<td>eicosane C(_{20})</td>
<td>−4.61</td>
<td>3.70</td>
</tr>
<tr>
<td>heneicosane C(_{21})</td>
<td>−5.10</td>
<td>2.59</td>
</tr>
<tr>
<td>docosane C(_{22})</td>
<td>−5.60</td>
<td>2.72</td>
</tr>
<tr>
<td>tetracosane C(_{24})</td>
<td>−6.58</td>
<td>1.06</td>
</tr>
<tr>
<td>pentacosane C(_{25})</td>
<td>−7.08</td>
<td>0.44</td>
</tr>
<tr>
<td>naphtalene</td>
<td>−0.21</td>
<td>6.28</td>
</tr>
<tr>
<td>acenaphthene</td>
<td>−1.58</td>
<td>6.10</td>
</tr>
<tr>
<td>fluorene</td>
<td>−1.95</td>
<td>5.58</td>
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<tr>
<td>phanthenrene</td>
<td>−2.78</td>
<td>4.57</td>
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<td>anthracene</td>
<td>−2.80</td>
<td>4.01</td>
</tr>
<tr>
<td>fluoranthene</td>
<td>−3.92</td>
<td>3.30</td>
</tr>
<tr>
<td>pyrene</td>
<td>−4.10</td>
<td>3.15</td>
</tr>
<tr>
<td>chrysene</td>
<td>−5.34</td>
<td>1.07</td>
</tr>
</tbody>
</table>

\(^{a}\) \( \log p^* \) values are temperature corrected. \(^{b}\) \( \log K_p \) values are the averages for the 08/01, 08/06, and 08/14 experiments. \(^{c}\) \( \log K_p \) values are the averages for the 08/05/96 and 08/14/96 experiments. \(^{d}\) \( \log K_p \) values are the averages for the 08/02/96 and 08/07/96 experiments.

### Results and Discussion

**Adsorptive Partitioning to \((NH_4)_2SO_4(s)\) Aerosol.** If adsorption is the dominant partitioning mechanism, a surface-area-normalized constant \( K_{p,s} \) can be defined as (17)

\[
K_{p,s} (m^3/m^2) = \frac{K_p}{TSP} \times 10^{-6} \mu g/g
\]

where \( a_{tsp} (m^3/g) \) is the specific surface area of the adsorbing solid. In this case, eq 3 can be rewritten as

\[
\log K_{p,s} = m_5 \log p^*_s + b_5
\]

where \( m_5 = m \) and \( b_5 = b_0 - \log (a_{tsp} / 10^6) \).

Table 2 gives the \( \log K_p \) values for the PAHs and \( n \)-alkanes for sorption to \((NH_4)_2SO_4(s)\) aerosol. Table 3 provides the corresponding \( m \) and \( b \) values. Log \( K_{p,s} \) values were calculated by normalizing the measured \( K_p \) values with \( a_{tsp} \) according to eq 5. The value of \( a_{tsp} \) was estimated to be 17.5 m\(^2\)/g by assuming that the \((NH_4)_2SO_4(s)\) particles were spherical with a density of 1.77 g/cm\(^3\). It is well known that the partitioning of organic compounds to inorganic surfaces is RH-dependent (17, 18) and that the sorption usually becomes stronger as the RH decreases. Since the partitioning experiments performed in the chamber with \((NH_4)_2SO_4(s)\) were conducted at the relatively low RH of 10% and since the ambient smog \( n \)-alkane data were collected at RH 42%, extrapolations of the \((NH_4)_2SO_4(s)\) log \( K_{p,s} \) values to RH 42% were necessary to permit a comparison of the two data sets.
Because of the presence of sorbed water on inorganic surfaces even at relatively low RH values, the adsorption of nonpolar organic gases has been found to be fundamentally similar for a range of inorganic surfaces including quartz, silica, and clay (19). We will therefore assume that adsorption to (NH₄)₂SO₄(s) particle surfaces is similar to adsorption to quartz surfaces. Storey et al. (17) have provided a detailed examination of the RH dependence of the adsorption of a range of PAHs and n-alkanes to a quartz surface. The needed compound-specific extrapolations were therefore based on the 30–70% RH data of Storey et al. (17). It was assumed that (1) only bₙ is a function of RH and (2) the change in bₙ with RH for (NH₄)₂SO₄(s) is similar to that for quartz.

The extrapolated compound-dependent increases in log Kₚ,s for quartz between RH 42% and RH 10% were therefore subtracted from the log Kₚ,s values for (NH₄)₂SO₄(s) obtained in this study. The results are plotted in Figure 1. The aᵥ TSP-normalized ambient smog n-alkane data are also plotted in Figure 1. Sheffield and Pankow (20) have reported that aᵥ TSP = 2.1 m²/g for UPM collected during the summer in Portland, OR. Corn et al. (21) reported that aᵥ TSP = 1.9 m²/g for UPM collected during the summer in Pittsburgh, PA. These values are consistent with measurements made in this study. Therefore, the value of aᵥ TSP for the ambient smog sample in this study was taken to be 2.0 m²/g.

Normalizing ambient smog aerosol log Kₚ values by an estimate of aᵥ TSP allows one to test whether adsorption is an important partitioning mechanism for UPM. As seen in Figure 1, the calculated log Kₚ,s values for the n-alkanes sorbing to the ambient smog samples at RH 42% are much larger than those for sorption to (NH₄)₂SO₄(s) aerosol as extrapolated to RH 42%. (A similar comparison for the PAHs is not possible because the PAH concentrations were too low to be quantified accurately with the methods used.) Considering that the inorganic portion of the midday smog aerosol was probably well represented by (NH₄)₂SO₄(s), we conclude that adsorption to inorganic surfaces probably cannot account for the sorption of the n-alkanes observed in the ambient samples. This suggests that adsorption was not the operative mechanism controlling the partitioning of SOCs to these ambient smog aerosol samples and that the log Kₚ,s values calculated for the n-alkanes sorbing to these samples are not physically meaningful quantities.

Absorptive Partitioning to Organic Aerosols. When absorption into a liquid (or at least amorphous) organic matter (om) phase is the dominant sorption mechanism, an om-phase-normalized constant Kₚ,om can be defined as (10, 22)

$$K_{p,om} = \frac{K_p}{f_{om}}$$

(7)

where fₗ is the weight fraction of the TSP that comprises the om phase. In this case, eq 3 can be rewritten as

$$\log K_{p,om} = m_{om} \log p_c^o + b_{om}$$

(8)

where mₗ = mₗ and bₗ = bₗ - log fₗ. Based on theory developed by Pankow (8), it can be shown that (22)

$$K_{p,om} = \frac{760RT}{M_{om} p_c^o 10^b}$$

(9)

where Mₗ is the number-averaged molecular weight of the om phase, c is the mole fraction scale activity coefficient of the compound of interest in that phase, and R is the gas constant (8.306 × 10⁻⁵ m³ atm mol⁻¹ K⁻¹).

The measured log Kₚ values for the liquid DOP aerosol and the amorphous SOA generated from the photooxidation of whole gasoline are given in Table 2. The corresponding log Kₚ vs log p_c plots are given in Figures 2 and 3. Also given are the regression lines for the partitioning of PAHs and n-alkanes to environmental tobacco smoke (ETS) as determined by Liang and Pankow (9). For the DOP, SOA, and ETS aerosols, fₗ was taken to be 1.0, yielding Kₚ = Kₚ,om. For the ambient smog samples, the carbon analyses revealed that the amount of IC in those samples was essentially zero so that TC = OC + EC. On average, TC constituted 21% of the TSP, with 87% of the TC being organic (OC/EC = 6.7). When all the aerosol is secondary in nature, Izumi and Fukuyama (23) have found that the mass of the om phase is 2.0 times the mass of total OC mass alone assuming that the density of om is 1.0 g/cm³. On the other hand, when most of the primary organic aerosol in LA in the summertime is coming from diesel exhaust and meat cooking (24), then most of the carbon is heavy alkanes and aldehydes. For these kinds
of aerosol, the mass of the om phase is 1.2 times the mass of total OC alone (25). In this context, allowing for the presence of other elements in the organic matrix (e.g., hydrogen and oxygen, including that in the water in the om phase), the mass of the om phase in Los Angeles aerosol was taken to be 1.6 times the mass of the total OC mass alone since 53–58% of the total OC was estimated to be secondary in nature for our ambient smog samples (see text below). These considerations lead to the estimate $f_{om} = 0.29$ for the ambient smog samples. This value was used with eq 8 and the assumption that the partitioning to the ambient smog was absorptive in nature to calculate log $K_{p,om}$ values for the ambient smog samples. The results are plotted in Figure 3.

The slopes and intercepts for the three regression lines in Figure 2 were compared using the “analysis of covariance” method described by Snedecor and Cochran (26). The slopes of the three regression lines in Figure 2 are all close to $-1$ and not significantly different (95% confidence level). This indicates that within each of the three organic aerosols phases, $\zeta$ remained approximately constant for the range of PAHs of interest for a given RH. The intercepts of the three regression lines in Figure 2 are moreover not significantly different (95% confidence level), indicating that the factor $M_{om,n}$ is approximately the same for the three organic aerosols considered. The partitioning of the n-alkanes to the organic aerosols is considered in Figure 3. On average, the partitioning of the n-alkanes to the organic aerosols was found to be weaker than that of the PAHs. For the n-alkanes, the partitioning was found to be similar for the gasoline SOA, ambient smog, and ETS aerosols. Within this group, the slopes of $\log K_{p,om}$ vs $\log p^*$ regression lines are all close to $-1$, and both the slopes and intercepts are not significantly different (95% confidence level) with the analysis of covariance. The sorption of the n-alkanes to the DOP aerosol, however, was much stronger than to the other three aerosols. Since it is very likely that $M_{DOP}$ was larger than the $M_{om,n}$ values for the other aerosols, the increase of partitioning was probably due to $\zeta$ values for the n-alkanes that were significant lower (and closer to 1) in the DOP aerosol as compared to the other aerosol phases.

During summer smog episodes in the South Coast Air Basin, a significant fraction of the ambient organic aerosol is often of secondary origin (27–32). Using data from Claremont, CA, Turpin and Huntzicker (32) suggested a method to estimate the secondary OC content based on the total OC ($= OC_{pri} + OC_{sec}$) and a derived correlation between $OC_{pri}$ and EC. For our ambient smog samples, by assuming that the $OC_{pri}/EC$ ratio for Pasadena was similar to that observed by Turpin and Huntzicker (32) in Claremont, we estimate that 53–58% of total OC was secondary in origin. Therefore, because the anthropogenic volatile organic gas profile in the South Coast Air Basin is very similar to that of whole gasoline vapor (33), it does seem likely that the SOA generated in the basin was similar to the SOA that was generated from whole gasoline in the smog chamber as suggested by Figure 3. Furthermore, the similarity indicates that it may be valid to extrapolate SOA yield data from smog chamber studies (10) to predict the extent of SOA formation during summer midday smog episodes.

O’Brien et al. (34) have reported that mono- and dicarboxylic acids make up an appreciable portion (22% and 10%, respectively) of the organic content of particulate material found under photochemical smog conditions in the Los Angeles area. Schuetzle et al. (35) have found high concentrations of organic alcohols and difunctional organic compounds in SOA. Based on the significant organic content of the ambient smog samples examined here, it is likely that the polarities of the om-portion of the ambient smog aerosol and the SOA generated from gasoline were much higher than the polarity of the DOP aerosol. Moreover, because of the presence of the two long alkane chains in DOP molecules, DOP aerosols are likely to be more hospitable to n-alkanes, the $\zeta$ values for the n-alkanes in DOP are expected to be closer to 1.0 as well as lower than in the other aerosol phases.

The overall spread in the log $K_p$ vs log $p^*$ correlation lines in Figure 3 is greater than that in Figure 2. This result may be the consequence of a greater sensitivity in the absorptive uptake of n-alkanes to changes in the polarity of the om-phase than is the case for the PAHs. It would be useful to know all of the $M_{om}$ values for the organic aerosols studies here so that this hypothesis could be tested directly by calculating the average $\zeta$ values for the PAHs and n-alkanes for each different aerosol type. (Methods for estimating $\zeta$ values in liquid mixture of known composition are also available (36–38).) Unfortunately, the only $M_{om}$ value that is known with some certainty at the present time is $M_{DOP}$, though it may be possible to estimate the molecular weight properties of diesel particle (39), smoke aerosol (40), atmospheric fine particle (41, 42), and secondary aerosol (34, 35). For DOP, we note that the solubility of DOP in water is only 3 mg/L at 25 °C (43). Since the mutual solubilities of water and organic compounds are well correlated (44), it can be concluded that the solubility of water in DOP will also be fairly low. Because the RH in the chamber experiments was only ~10%, it is very reasonable to assume that $M_{DOP}$ was very close to the molecular weight of pure DOP, namely, 390 g/mol. When log $p^*$ = −3, the regression parameters in Table 3 for DOP give log $K_{p,om}$ = −3.75 and −4.21 for the PAHs and the n-alkanes, respectively. These values yield $\zeta$ = 0.28 and 0.8, respectively. These results are consistent with literature values for $\zeta$ in DOP. In particular, the $\zeta$ values for benzene and octane in DOP have been reported to be 0.57 (60 °C) and 1.4 (120 °C) (45).

While it is instructive as well as useful to have a detailed understanding of the individual values of $\zeta$ and $M_{om}$, for a given compound or class of compounds sorbing to a given organic aerosol, there will be many circumstances for which knowledge of the product of $M_{om} \cdot \zeta$ alone will be sufficient. In this context, we note that the sorption properties of ETS and the SOA generated from gasoline were very similar, both for the PAHs and n-alkanes. This suggests that ETS may be a useful surrogate phase for SOA. It may therefore be possible to use ETS and the methods of Liang and Pankow (9) to
estimate the log $K_{p,om}$ values for SOA for other classes of compounds (e.g., organic acids and carboxylic acids). Similarly, DOP aerosol may be a useful surrogate for ambient organic aerosol that consists mainly of moderately nonpolar organic compounds of primary emission origin (e.g., diesel aerosol, which is about 50% organic carbon on average (46)).

In summary, we conclude that the G/P partitioning of SOC to UPM containing a significant fraction of secondary organic carbon will be dominated by absorptive partitioning. This finding is consistent with the conclusions of Liang and Pankow (9) for UPM. This suggests that G/P partitioning in urban environments is correctly described by eqs 7–9 and supports the absorptive model for SOA formation proposed by Odum et al. (10).

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