

Simultaneous measurements of gas- and aerosol-phase water-soluble organic nitrogen in the winter urban atmosphere of Chengdu, China

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Abstract: Partitioning gaseous water-soluble organic nitrogen (WSON) to the aerosol phase is a major formation pathway of atmospheric secondary organic aerosols (SOA). However, the factors influencing this WSON partitioning process remain unclear. We conducted simultaneous wintertime measurements of WSON in both gas and aerosol phases at an urban site in Chengdu (CD), located in the Sichuan Basin (SCB), China, to investigate the concentration, gas-particle partitioning processes, and driven factors of WSON. The average concentration of particulate WSON ($2.3 \pm 1.4 \mu\text{gN m}^{-3}$) was about twice that of gaseous WSON ($1.2 \pm 0.9 \mu\text{g N m}^{-3}$) and increased significantly as $\text{PM}_{2.5}$ increased. In winter, WSON readily partitions into the particle phase ($F_{\text{WSON}} = 0.70 \pm 0.14$), partitioning into the organic phase under warm, dry conditions and into aerosol liquid water under cold, humid conditions, with higher F_{WSON} in the latter. Amines (methylamine, MA; dimethylamine, DMA) were predominantly present in the gas phase but exhibited enhanced partitioning into the particle phase during the $\text{PM}_{2.5}$ polluted periods. Amines preferentially partitioned into aerosol liquid water throughout the entire campaign, stably existing as amine-sulfate or amine-nitrate salts in $\text{PM}_{2.5}$. The influence of aerosol liquid water content (ALWC) on the partitioning behavior of both WSON and amines was substantially greater than that of temperature (T), relative humidity (RH), and aerosol acidity (pH), as further confirmed by thermodynamic modeling with S-curve analysis. Furthermore, NH_4NO_3 was identified as the primary contributor to ALWC, suggesting that controlling NO_x and NH_3 emissions is crucial for reducing ALWC and subsequent WSON formation.

1 Introduction

Organic nitrogen (ON) is a ubiquitous component of fine particulate matter (PM_{2.5}), accounting for 10%–50% of total nitrogen (TN) in global aerosols (Leung et al., 2024). Atmospheric nitrogen-containing organic compounds (NOCs) encompass a wide range of species, including amines, N-heterocycles, nitroaromatics, organic nitrates, and so on (Jickells et al., 2013; Yu et al., 2024). These species partition between the gas and condensed phases depending on their volatility and water solubility. For example, low-molecular-weight amines and nitroaromatic compounds are semi-volatile and partition dynamically between phases (Liu et al., 2018; Chen et al., 2025), while monofunctional organic nitrates predominantly reside in the gas phase, whereas multifunctional ones preferentially partition into the particle phase (Rollins et al., 2013; González-Sánchez et al., 2021). Among these, water-soluble organic nitrogen (WSO_N) is of particular interest, as it actively participates in aqueous-phase reactions and significantly contributes to secondary organic aerosol (SOA) formation (Yu et al., 2020).

Amines, as a major class of water-soluble NOCs in the atmosphere, play a pivotal role in aerosol formation and evolution. The H₂SO₄-amine-H₂O ternary nucleation mechanism markedly enhances new particle formation (NPF) rates by stabilizing critical clusters, with ambient observations demonstrating more than a thousand-fold increase in formation rates compared to ammonia (Almeida et al., 2013; Schobesberger et al., 2013; Yao et al., 2018; Kanawade and Jokinen, 2025; Yu et al., 2025). Through acid-base reactions, amines readily form stable aminium salts with inorganic acids, altering particles hygroscopicity, density, and optical properties (Qiu and Zhang, 2013). Additionally, Maillard-like reactions between amines and carbonyls in the aqueous phase contribute substantially to brown carbon (BrC) formation (Tang et al., 2022; Xiao et al., 2025). Despite these recognized roles, the gas-aerosol partitioning behavior of amines remains critically underexplored. Previous studies suggest that amine partitioning is influenced by aerosol acidity (pH), relative humidity (RH), and condensed-phase composition, and is strongly pH-dependent, favoring the particle phase under acidic conditions (Ge et al., 2011; Pankow, 2015). However, amine gas-aerosol partitioning remains poorly understood, hindering accurate assessments of their contributions to PM formation and climate, and thus warranting urgent investigation.

In contrast, the gas-aerosol partitioning of bulk water-soluble organic compounds (WSOC) has been more extensively investigated. Recent simultaneous online

measurements in the Yangtze River Delta (YRD) and North China Plain (NCP) have systematically elucidated the driving mechanisms of WSOC partitioning (Lv et al., 2022a; Lv et al., 2022b; Li et al., 2025; Li et al., 2026). These studies revealed that under dry conditions ($RH < 80\%$), WSOC partitioning is governed by temperature (T) and proceeds via condensation into the organic phase; under humid conditions ($RH > 80\%$), it shifts to being dominated by aerosol liquid water content (ALWC), with uptake into the aqueous phase (Lv et al., 2022a). Ammonia (NH_3) not only elevates particle pH through neutralization to promote organic acid dissociation, but also participates in aqueous-phase reactions to form nitrogen-containing organics (e.g., imidazoles), contributing to WSON and BrC light absorption (Lv et al., 2022a; Li et al., 2026). Nationwide data show a strong linear correlation ($R = 0.77$) between $WSOC_p$ and NH_4NO_3 mass fractions, confirming the critical role of NH_3 in regional SOA formation (Lv et al., 2022a). However, the gas-aerosol partitioning behavior of WSON has received far less attention, with most studies limited to particle-phase measurements and thus unable to capture the dynamic exchange between phases.

This knowledge gap is especially pronounced in regions with complex aerosol chemistry and high precursor emissions, such as the Sichuan Basin (SCB), a topographically enclosed basin characterized by high RH, frequent winter haze, and substantial agricultural and industrial NH_3 emissions. To address this gap, we conducted an intensive winter observation campaign in Chengdu (CD) from December 2024 to January 2025, simultaneously measuring gas- and particle-phase WSON and two amines. Our work aims to elucidate the pollution characteristics of WSON and amines, and the driving factors governing their gas-particle partitioning behavior in the atmosphere.

2 Materials and methods

2.1 Observation site description.

Winter sampling was conducted in CD, located in the SCB, from December 14, 2024, to January 9, 2025. The observation site was situated on the roof of the Civil Engineering Building at Southwest Jiaotong University's Jiuli Campus ($30.7^\circ N$, $105.05^\circ E$) (Fig. S1), at a height of 25 meters above ground level. It was located in the central area of CD, near the North Second Ring Road, surrounded by residential areas, shopping malls, gas stations, and restaurants. During the sampling period,

meteorological conditions in CD were stagnant with low wind speeds and high RH (Table 1). The SCB features a typical deep-basin topography (with a basin depth exceeding 2000 m), surrounded by mountains on all sides, which severely hampers the dispersion of air pollutants (Rao et al., 2026). In winter, the Qinling Mountains block the intrusion of cold air, allowing warm and moist air masses to accumulate within the basin, leading to high humidity, frequent fog, and recurrent temperature inversions (with a winter inversion frequency of 95.4% and an inversion layer depth of 289.4 m) (Feng et al., 2020). During pollution episodes, the boundary layer is significantly compressed, with its height reduced by approximately 50% (Lu et al., 2024), resulting in a stable atmospheric stratification that is extremely unfavorable for pollutant diffusion. The synergistic effects of increased emissions, stagnant meteorological conditions, and enhanced aqueous-phase reactions under high relative humidity (> 80%) can readily cause extreme PM_{2.5} episodes (Shu et al., 2025), and the humid environment further promotes the hygroscopic growth and secondary transformation of particulate matter (Wang et al., 2021). In CD, the major sources of PM_{2.5} are vehicle emissions (25.6%) and secondary sources (40.5%) (Zhang et al., 2024). In winter, secondary inorganic ions account for 40% – 60% of PM_{2.5} mass, of which approximately half is nitrate (Rao et al., 2026), providing abundant precursors for the secondary formation of NOCs (Li et al., 2023). The combined topographic-meteorological-emission characteristics impart unique regional representativeness to the study of gas-aerosol partitioning of NOCs under the humid and stagnant conditions typical of CD. Therefore, the observation site can effectively capture the influence of the basin-specific atmospheric chemical processes on the partitioning of WSON.

2.2 Measurements of the gas- and aerosol-phase species.

Continuous measurements of inorganic ions, WSON, methylamine (MA) and dimethylamine (DMA) in the gas and aerosol phases were conducted at the campus site. Online samples of the aforementioned compounds were collected via the In-situ Gas and Aerosol Compositions monitor (IGAC, model 63GA, Fortelice International Co., Ltd., Taiwan, China) coupled with an ICS–5000+ ion chromatograph (Thermo Scientific, USA) and a TOC/TN analyzer (model TOC/TN–L CPH, Shimadzu, Inc., Japan) for simultaneous measurements of online samples. WSON was measured with a time interval of 3 hours, while other species were measured hourly. In this study, the

equation for calculating WSON concentration is as follows:

$$WSON_{g,p} = WSTN_{g,p} - WSIN_{g,p} \quad (1)$$

where, g and p represent the gas phase and particle phase, respectively; WSTN represents total water-soluble nitrogen measured by TOC/TN analyzer; WSIN represents the total water-soluble inorganic nitrogen measured by IGAC–IC, when calculating this value, the average is taken over a 3 h WSON sampling intervals. The unit is $\mu\text{gN m}^{-3}$. Due to the differing forms in which inorganic nitrogen exists in the gas phase and particulate phase, the calculation methods will vary accordingly, as follows:

$$WSIN_g = \left(\frac{[\text{NH}_3]}{17} + \frac{[\text{HNO}_3]}{63} + \frac{[\text{HNO}_2]}{47} \right) \times 14 \quad (2)$$

$$WSIN_p = \left(\frac{[\text{NH}_4^+]}{18} + \frac{[\text{NO}_3^-]}{62} + \frac{[\text{NO}_2^-]}{46} \right) \times 14 \quad (3)$$

The gas-particle phase partition coefficient (F_p) for WSON is calculated as follows, similarly, similar calculations apply to amines:

$$F_p = \frac{C_p}{C_g + C_p} \quad (4)$$

Five anions (SO_4^{2-} , NO_3^- , NO_2^- , Cl^- , F^-), five cations (NH_4^+ , Na^+ , K^+ , Mg^{2+} , Ca^{2+}), and two amines (MA and DMA) were measured in this study. $\text{PM}_{2.5}$ and meteorological parameters (including T and RH) were measured by the E–BAM continuous environmental $\text{PM}_{2.5}$ monitor (Met One Instruments, United States) and an automatic weather station (MILOS520, Vaisala, Inc., Finland), respectively. In this study, pollution periods (PP) were defined as days with a daily $\text{PM}_{2.5} > 75 \mu\text{g m}^{-3}$, clean periods (CP) as $\text{PM}_{2.5} < 35 \mu\text{g m}^{-3}$, and transition periods (TP) as $\text{PM}_{2.5}$ between 35 and $75 \mu\text{g m}^{-3}$. The organic carbon (OC) and elemental carbon (EC) of the $\text{PM}_{2.5}$ filter samples were analyzed by DRI Model 2015 Carbon Analyzer (Atmoslytic, Inc., Calabasas, USA) with IMPROVE_A protocol (Chow et al., 2007; Li et al., 2025).

To minimize background interference and systematic errors in the instrument, a HEPA filter was installed at the IGAC sampling port. Measurement results indicated that the background levels in the tube system were comparable to the blank concentration of deionized water. The background concentrations of WSOC and WSTN in the tubes were $0.049 \pm 0.030 \text{ mg L}^{-1}$ and $0.007 \pm 0.002 \text{ mg L}^{-1}$, respectively, significantly lower than the sample concentrations. Additionally, the collection

efficiency of IGAC system for both gas and particle phase species is $> 73\%$ (Lv et al., 2022a; Lv et al., 2022b), and the data during the observation period were corrected accordingly. For detailed information on instrument applications and method accuracy tests, please refer to the Support Information Text S1.

2.3 Calculations on ALWC and acidity of PM_{2.5}.

The ALWC and pH of PM_{2.5} were estimated using the thermodynamic ISORROPIA-II model based on hourly data of aerosol chemical composition (including Na⁺, NH₄⁺, K⁺, Ca²⁺, Mg²⁺, SO₄²⁻, NO₃⁻, and Cl⁻), gaseous NH₃ concentration, and meteorological parameters (T; RH) (Fountoukis and Nenes, 2007). The forward metastable mode was employed for the estimation due to its lower sensitivity to measurement errors compared to the reverse mode (Hennigan et al., 2015). Data corresponding to RH greater than 95% were excluded on the basis that such conditions are associated with increased uncertainty in the model estimation of both ALWC and pH (Guo et al., 2015). The contribution of organic compounds to ALWC (ALWC_{org}) was determined using the mass concentration of organic matter (OM) which was calculated on the basis of OC (OM = OC $\times$ 1.6) (Lv et al., 2022a; Lv et al., 2022b), as described in detail in the Supporting Information Text S2.

2.4 S-Curve analysis of amines.

To further elucidate the effects of pH and ALWC on the gas-particle phase distribution of organic amines, this study assumes that the gas and particle phases of amines are in thermodynamic equilibrium. $\varepsilon(\text{CH}_3\text{NH}_3^+)$ and $\varepsilon((\text{CH}_3)_2\text{NH}_2^+)$, the theoretical values of the gas-particle phase distribution coefficients, are calculated using a thermodynamic model and analyzed via S-curve analysis with aerosol pH. The calculation equations are as follows:

$$\varepsilon(\text{CH}_3\text{NH}_3^+) = \frac{\left(\frac{\gamma_{\text{H}^+}}{\gamma_{\text{CH}_3\text{NH}_3^+}} 10^{-\text{pH}} + \frac{K_a}{\gamma_{\text{CH}_3\text{NH}_2}} \right) \frac{H_{\text{CH}_3\text{NH}_2}}{K_a} W_i RT \times 0.987 \times 10^{-14}}{1 + \left(\frac{\gamma_{\text{H}^+}}{\gamma_{\text{CH}_3\text{NH}_3^+}} 10^{-\text{pH}} + \frac{K_a}{\gamma_{\text{CH}_3\text{NH}_2}} \right) \frac{H_{\text{CH}_3\text{NH}_2}}{K_a} W_i RT \times 0.987 \times 10^{-14}} \quad (5)$$

$$\varepsilon((\text{CH}_3)_2\text{NH}_2^+) = \frac{\left(\frac{\gamma_{\text{H}^+}}{\gamma_{(\text{CH}_3)_2\text{NH}_2^+}} 10^{-\text{pH}} + \frac{K_a}{\gamma_{(\text{CH}_3)_2\text{NH}}} \right) \frac{H_{(\text{CH}_3)_2\text{NH}}}{K_a} W_i RT \times 0.987 \times 10^{-14}}{1 + \left(\frac{\gamma_{\text{H}^+}}{\gamma_{(\text{CH}_3)_2\text{NH}_2^+}} 10^{-\text{pH}} + \frac{K_a}{\gamma_{(\text{CH}_3)_2\text{NH}}} \right) \frac{H_{(\text{CH}_3)_2\text{NH}}}{K_a} W_i RT \times 0.987 \times 10^{-14}} \quad (6)$$

where, H_i is the Henry's law constant for organic amine i at 298.15 K, K_a is acid dissociation constant at 298.15 K, γ_i represents the activity coefficient, which can be calculated using the E-AIM model; W_i is the ALWC of the particles; R is the gas constant, and T is the temperature. Detailed derivations of Equations (5) and (6) are provided in the Support Information Text S3.

2.5 Oxidation ratio calculation.

The higher nitrogen oxidation ratio (NOR) and sulfur oxidation ratio (SOR) represent more efficient conversion of gaseous species into secondary aerosols. The calculation equations are as follows:

$$\text{NOR} = \frac{[\text{NO}_3^-]}{[\text{NO}_3^-] + [\text{NO}_2]} \quad (7)$$

$$\text{SOR} = \frac{[\text{SO}_4^{2-}]}{[\text{SO}_4^{2-}] + [\text{SO}_2]} \quad (8)$$

where, $[i]$ refers to the molar concentration of species i .

3 Results and discussion

3.1 Chemical characteristics of PM_{2.5} pollution during the campaign.

The temporal variations of meteorological parameters, major atmospheric pollutants during the whole campaign are shown in Fig. 1 and Table 1, respectively. During the campaign, the average PM_{2.5} concentration in CD was $76 \pm 38 \mu\text{g m}^{-3}$ (Table 1), with 14 pollution days accounting for 51.9% of the total sampling days. The average PM_{2.5} concentration during PP was $100 \pm 33 \mu\text{g m}^{-3}$, with the highest pollution level reaching $199 \mu\text{g m}^{-3}$. Concurrently, NO_3^- concentrations (Table S1) and contributions (Table S1) began to exceed those of SO_4^{2-} and exhibited an overall increasing trend, indicating that NO_3^- has become the dominant inorganic salt in PM_{2.5}. In contrast, SO_4^{2-} remained relatively stable and was significantly lower in 2024 compared to previous years (Table S1). NH_4^+ has remained relatively stable since its decline began in 2016, as has Cl^- . In this study, SO_4^{2-} , NO_3^- , and NH_4^+ accounted for $7.2 \pm 3.1\%$, $22.8 \pm 6.6\%$, and $11.5 \pm 4.6\%$ of the PM_{2.5} mass, respectively (Fig. 1b). Additionally, the observed concentration of OM in CD was $21.1 \pm 9.6 \mu\text{g m}^{-3}$ (Table 1), contributing $27.3 \pm 7.4\%$ to the total PM_{2.5} mass. As shown in Table S1, previously reported values in the SCB

also exceeded 25%. These results clearly indicate that NO_3^- and OM are the major components of $\text{PM}_{2.5}$ in the SCB, accounting for over 50% of the total $\text{PM}_{2.5}$ mass.

The temporal variation of WSO_Np (Fig. 1c) exhibited the same trend as $\text{PM}_{2.5}$ (Fig. 1b), with a significant positive correlation between WSO_Np and $\text{PM}_{2.5}$ ($R = 0.78$, $p < 0.05$, Fig. S2). WSO_Np contributed to 0.5 – 12.4% of the $\text{PM}_{2.5}$ mass, with the average value was 3% (Fig. S3a–d). The concentration of WSO_Np ($2.3 \pm 1.4 \mu\text{gN m}^{-3}$) was more than twice that of WSO_Ng ($1.2 \pm 0.9 \mu\text{gN m}^{-3}$) (Fig. 1c and Table 1), indicating that WSON in CD mainly exists in the particle phase, as observed at Mt. Fuji in Japan (Matsumoto et al., 2020). WSON contributed 9.3% and 14.7% to the TN in the gas phase and particulate phase, respectively. In CD, the $\text{WSO}_\text{N}/\text{WSOC}$ ratios for the gas and particle phases were 0.05 and 0.15 (Fig. S3e–l), respectively. Notably, the $\text{WSO}_\text{N}/\text{WSOC}_\text{p}$ ratio increases as pollution worsens (Fig. S3i–l), indicating that more NOCs are enriched during the accumulation of OM. Our study shows that the wintertime WSO_Np concentration of $\text{PM}_{2.5}$ in CD ($2.3 \pm 1.4 \mu\text{gN m}^{-3}$) was slightly higher than those in central Chinese cities such as Nanjing ($1.7 \pm 0.6 \mu\text{gN m}^{-3}$) (Guan et al., 2022) and Shanghai (SH, $1.9 \pm 1.8 \mu\text{gN m}^{-3}$) (Lv et al., 2022a), as well as in the northern city of Wuhai ($1.7 \pm 1.1 \mu\text{gN m}^{-3}$) (Xie et al., 2025), but substantially higher than those in southern coastal cities such as Guangzhou ($0.48 - 0.53 \mu\text{gN m}^{-3}$) (Yu et al., 2017) and Hong Kong ($0.6 \pm 0.6 \mu\text{gN m}^{-3}$) (Leung et al., 2024). This indicates that the wintertime NOCs of $\text{PM}_{2.5}$ in CD are notably prominent, likely due to its unique basin topography, which allows local emissions (e.g., biomass burning (Yu et al., 2017; Xie et al., 2025), cooking fumes (Özel et al., 2010; Ditto et al., 2022), vehicle exhaust (Yu et al., 2017; Xie et al., 2025)) and secondary formation (Liu et al., 2023; Yu et al., 2017; Xie et al., 2025) to continuously accumulate near the surface, resulting in concentrations that exceed those in other cities.

As shown in Fig. 1d and Table 1, the gas-phase concentrations of methylamine (MA_g) and dimethylamine (DMA_g) ($1.23 \pm 0.37 \mu\text{g m}^{-3}$ and $0.41 \pm 0.03 \mu\text{g m}^{-3}$, respectively) were 3-5 times higher than their particle-phase concentrations ($0.45 \pm 0.24 \mu\text{g m}^{-3}$ and $0.09 \pm 0.05 \mu\text{g m}^{-3}$, respectively). This trend was consistent with observations reported from the Nanling Mountains in Guangzhou, China (MA_g : $60.6 \pm 35.0 \text{ ng m}^{-3}$, MA_p : $8.8 \pm 7.8 \text{ ng m}^{-3}$; DMA_g : $71.3 \pm 54.5 \text{ ng m}^{-3}$, DMA_p : $2.4 \pm 3.2 \text{ ng m}^{-3}$) (Liu et al., 2018), suggesting that amines predominantly exist in the gas phase in the atmosphere. Furthermore, the observed amine concentrations were much higher

than those reported in other regions (Liu et al., 2018; Yang et al., 2022; Zhu et al., 2022; Yang et al., 2023; Brean et al., 2025). Studies show that vehicle emissions elevate roadside C₁–C₂ amine concentrations by more than a factor of four relative to background levels (Brean et al., 2025), with a contribution exceeding 70% (Yang et al., 2022; Zhu et al., 2022). As shown in Fig. S4, amines in CD exhibited significant correlations with the vehicular tracers NO₂ and CO ($r = 0.4$ and 0.5 , respectively, $p < 0.05$), indicating that vehicle exhaust contributes to amine emissions to some extent. Except primary emissions, secondary formation also was a major source of amines (Yang et al., 2023). In CD, particulate amines exhibited strong correlation with SNA (sulfate, nitrate, ammonium) ($r > 0.5$, $p < 0.01$; Fig. S5), indicating that secondary formation driven by acid–base neutralization is also an important source.

3.2 Gas-to-aerosol phase partitioning of WSON

Figure 2 illustrates the morphological characteristics of aerosols under two different temperature conditions. The F_{WSON} values were lower during the high-T period (0.61 ± 0.11) than during the low-T period (0.71 ± 0.13) (Figure 2a), suggesting different partitioning processes under the two T regimes. During the high-T process, the OM fraction ($f_{OM} = 0.22 \pm 0.08$) (Figure 2b) exceeded the ALWC fraction ($f_{ALWC} = 0.15 \pm 0.08$) (Figure 2c), indicating that under high-T and low-RH conditions, the aerosol predominantly existed in an organic phase. In contrast, during the low-T and high-RH process, the f_{ALWC} was substantially higher than the f_{OM} , with f_{ALWC} being approximately 4.5 times that of f_{OM} , suggesting that under low-T and high-RH conditions, the aerosol was mainly present in an aqueous phase. Furthermore, Figure 2d shows that the SNA-to-OM ratio during the low-T process (1.16 ± 0.45) was considerably lower than that during the high-T process (1.62 ± 0.42). Although the high-T process exhibited high concentrations of hygroscopic inorganic components, the RH was only $53 \pm 13\%$ (Figure 2e), significantly lower than that during the low-T process ($92 \pm 4\%$) (Figure 2e), which limited the increase in ALWC. Consequently, f_{ALWC} during the high-T process was much lower than that during the low-T process (0.59 ± 0.11) (Figure 2c). Therefore, the elevated OM concentrations during the high-T process may have facilitated the formation of an organic aerosol phase, whereas the combination of high RH and abundant inorganic components was likely the primary

driver for the high ALWC content observed during the low-T process. In summary, water-soluble organic compounds were preferentially partitioned into the organic phase during the high-T process, whereas they were mainly distributed into the aqueous phase during the low-T process.

Table 2 presents the differences in F_{WSO_N} across three Chinese cities, namely CD (representing the SCB), SH (representing the YRD), and Xianghe (XH, representing the NCP). As shown in the Table 2, F_{WSO_N} was highest in CD (0.70 ± 0.14), followed by SH (0.49 ± 0.26), and lowest in XH (0.43 ± 0.23). The variation in F_{WSO_N} among the three cities did not follow the same trends as T and RH, and no significant differences in pH were observed, suggesting that T, RH, and pH exerted only weak influences on F_{WSO_N} . ALWC showed a decreasing trend across the three cities, which was consistent with the regional variation trend of F_{WSO_N} . In addition, it was also observed that within each city, F_{WSO_N} increased with increasing ALWC, implying that ALWC may play a substantial role in modulating F_{WSO_N} variations. As mentioned earlier, CD is located at the bottom of the SCB, and its unique deep-basin topography leads to unfavorable meteorological conditions in winter, such as stagnant conditions (Table 1) and the frequent occurrence of low boundary layers, resulting in the continuous accumulation of pollutants. The strong temperature inversions in CD promote increases in the low-level cloud liquid water path and near-surface RH, exerting a pronounced regulatory effect on the accumulation of ALWC (Guo et al., 2026). The coupling of these topographic and meteorological conditions makes the wintertime ALWC level in CD significantly higher than that in other regions, making the regulatory effect of ALWC on WSON gas-aerosol partitioning more pronounced than that in coastal or northern Chinese cities.

To further investigate the influence of ALWC on the partitioning of WSON in China, we conducted a statistical analysis of the relationship between the spatial distribution of the f_{ALWC} and F_{WSO_N} across the country (Fig. 3a). A significant linear correlation was observed between f_{ALWC} and F_{WSO_N} ($R = 0.78$, $p < 0.05$), indicating that higher ALWC levels promote the partitioning of WSON_g into the particle phase. As illustrated in Fig. 3b, ALWC increased with increasing SNA concentrations during the observation period, particularly under conditions of high RH. Calculations using the ISORROPIA-II thermodynamic model indicate that NH_4NO_3 contributed approximately 61.3% to ALWC, followed by $(\text{NH}_4)_2\text{SO}_4$ (14.4%) and OM (7.7%) during the whole campaign in CD (Fig. 3c). Figure 3d demonstrates that ALWC

increased with the ratio of NH_4NO_3 to $\text{PM}_{2.5}$ at a given RH, suggesting that ALWC in CD during winter is primarily driven by NH_4NO_3 . Therefore, controlling emissions of NH_4NO_3 precursors (NO_x and NH_3) is crucial for both ALWC and WSON.

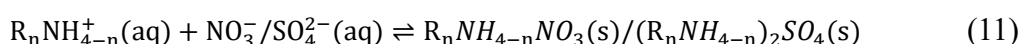
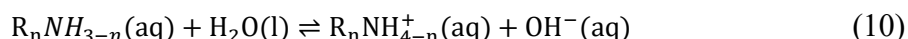
3.3 Gas-to-aerosol phase partitioning of amines

Amines contribute significantly to $\text{WSON}_{\text{g,p}}$ in CD. Specifically, amines-N (MA-N and DMA-N) account for 48.7% and 10.5% of WSON in the gas and particle phases, respectively (Fig. 4). During the winter haze periods in CD, the total concentration of amines increased substantially, and the gas-particle partitioning coefficient (F) nearly doubled (Table 1), indicating enhanced gas-to-particle conversion. It is also evident from Table 1 that the gas-aerosol partitioning coefficients of the basic nitrogen-containing species (NH_4^+ , MA, DMA) followed a decreasing order of $F_{\text{AM}} (0.36) > F_{\text{MA}} (0.25) > F_{\text{DMA}} (0.18)$, indicating that as the hydrogen atoms on the amino group ($-\text{NH}_2$) are progressively substituted by hydrophobic methyl groups ($-\text{CH}_3$), these basic species increasingly tend to remain in the gas phase rather than partitioning into the aerosol phase. Previous studies have demonstrated that with increasing degree of methylation of amines, their relative hydration free energy increases, thereby weakening their affinity for water and enhancing their partitioning from the aerosol phase to the gas phase (Rizzo and Jorgensen, 1999). In addition, as shown in Table S2, the reaction rate of NH_3 with hydroxyl radical ($\cdot\text{OH}$) in the atmosphere is relatively slow ($1.6 \times 10^{-13} \text{ cm}^3 \text{ molecule}^{-1} \text{ s}^{-1}$), corresponding to an atmospheric lifetime of up to 72.3 days. In contrast, MA ($(2.20 \pm 0.22) \times 10^{-11} \text{ cm}^3 \text{ molecule}^{-1} \text{ s}^{-1}$) and DMA ($(6.54 \pm 0.66) \times 10^{-11} \text{ cm}^3 \text{ molecule}^{-1} \text{ s}^{-1}$) react extremely rapidly with $\cdot\text{OH}$, with lifetimes of only a few hours (12.6 h and 4.3 h for MA and DMA, respectively) (Table S2). This implies that the fast chemical loss of their gas-phase concentrations directly reduces their accumulation potential in the aerosol phase, leading to the observed decline in the measured F values with increasing methylation degree.

To investigate the partitioning behavior of amines, we calculated their partitioning coefficients under different diel conditions, as shown in Fig. 5. As depicted in Figs. 5a and 5c, OM exhibited no significant correlation with the F_p of amines during either daytime or nighttime ($R < 0.5$, $p > 0.05$), indicating that OM exerted only a weak influence on the gas-aerosol partitioning of MA and DMA, and that their partitioning into the aerosol organic phase was relatively low. In contrast, both F_{MA} and F_{DMA}

showed a clear increasing trend with increasing ALWC ($R > 0.5$, $p < 0.05$), suggesting that MA and DMA preferentially partitioned into the aerosol aqueous phase.

Furthermore, we found significant linear relationships between F_p and acidic anions (nitrate and sulfate) ($R > 0.5$, $p < 0.01$), and the coefficients increased with NOR and SOR (Fig. 6). This indicates that inorganic salts significantly promote the partitioning of MA and DMA, and that reactive uptake may also serve as one of the driving forces facilitating the partitioning of amines into aerosol. According to the Reactions (9 – 12) (Ge et al., 2011), gaseous amines synergistically neutralize H_2SO_4/HNO_3 together with NH_3 to generate particulate amine salts (Almeida et al., 2013; Glasoe et al., 2015; Liu et al., 2018). Figure S5 presents a comparison between amines and inorganic salts. Our results show that amines exhibited good linear correlations with both nitrate and sulfate ($R > 0.5$, $p < 0.01$), suggesting that MA and DMA are likely present in particles predominantly as amine nitrate and/or amine sulfate salts (Huang et al., 2022). Furthermore, ALWC was found to promote their formation (Fig. S5). The humid and stagnant meteorological conditions in CD (Table 1) provide an ideal environment for the above transformations, promoting efficient enrichment of amines in the aerosol phase and making them a significant component of water-soluble ions in $PM_{2.5}$.



To further identify the key factors governing the partitioning of amines, we analyzed the correlations between F_p and meteorological conditions (i.e., RH and T) as well as pH. As shown in Figures S6a–f, no significant linear or nonlinear correlations were observed between F_{MA} and T, RH, or pH during either daytime or nighttime ($R < 0.5$). Similarly, F_{DMA} did not show notable correlations with T, RH, or pH during the daytime ($R < 0.5$, $p > 0.05$). However, during the nighttime, F_{DMA} exhibited a significant nonlinear relationship with RH, with an exponential increase when RH exceeded 80%, while the influences of T and pH remained relatively weak ($R < 0.5$, $p > 0.05$). Figures 7a and 7b present the theoretical values of the gas-particle partitioning coefficients for MA and DMA ($\epsilon(CH_3NH_3^+)$ and $\epsilon((CH_3)_2NH_2^+)$, respectively) calculated using a thermodynamic model and plotted against aerosol pH in an S-curve

analysis. The observed partitioning coefficients for amines in CD fell within the range predicted by theoretical calculations; however, they did not exhibit the anticipated decrease with increasing pH. Theoretically, the partitioning of amines is co-regulated by pH, ALWC, and T. Under acidic conditions with low pH, amines are protonated to form ionic species, with ϵ approaching 100%, favoring their partitioning into the aerosol phase. As pH increases, deprotonation occurs, forming volatile free bases, with ϵ approaching 0, leading to their transfer to the gas phase. However, given the acidity of atmospheric aqueous phases (pH 0–5), amines exist almost entirely in their protonated form ($R-NH_3^+$), within the observed pH range and the uncertainty of the available measurements, no significant independent pH dependence was detected (Tilgner et al., 2021). Generally, lower temperatures enhance the effective solubility of trace gases. However, (Tilgner et al., 2021) explicitly noted that aqueous-phase partitioning coefficients are higher in clouds (typically around 285 K) compared to near-surface conditions during hot summers, and higher in winter compared to summer. This indicates that the influence of T on partitioning coefficients is manifested in vertical altitude gradients and seasonal variations. Within a single season, its impact is less pronounced compared to that of ALWC (Tilgner et al., 2021). In this study, the partitioning coefficients of amines showed no significant correlation with T but exhibited a significant positive correlation with ALWC (Fig. 5). Consequently, variations in pH and T exert minimal influence on amine partitioning (Fig. S6), with ALWC emerging as the dominant limiting factor (Fig. 5).

According to the (Pankow, 1994), aerosol mass loading may also be a factor affecting the gas-particle partitioning of organic compounds (Pankow, 1994; Lutz et al., 2019; Li et al., 2025). This relationship is typically described using Raoult's law, with the calculation equation as follows (Pankow, 1994):

$$K_i = \frac{[i]_{particle}}{[i]_{gas} TSP} \quad (11)$$

where K_i is the partitioning constant of compound i , $[i]_{particle}$ and $[i]_{gas}$ represent the concentrations of compound i in the particle and gas phases, respectively, and TSP is the concentration of total suspended particulate material (PM_{2.5} mass concentrations are used in this study). Figure S7 shows the relationship between the $[i]_{particle}/[i]_{gas}$ ratios of MA and DMA and PM_{2.5} mass concentrations during the clean, transition, and

pollution periods, respectively. For MA, the correlations between its $[i]_{particle}/[i]_{gas}$ ratio and $PM_{2.5}$ were weak across all the three periods, with $R < 0.5$ and $p < 0.01$. The $[i]_{particle}/[i]_{gas}$ ratio of DMA during the clean period showed a correlation with $PM_{2.5}$ ($R = 0.56$, $p < 0.01$); however, according to the application criteria for the K_i value proposed by (Lutz et al., 2019), an R^2 greater than 0.7 is required for accurate K_i determination and meaningful interpretation, indicating that the results did not fall within the applicability range in this study. During the transition and pollution periods, the $[i]_{particle}/[i]_{gas}$ ratio of DMA also showed no significant correlation with $PM_{2.5}$ ($R < 0.4$, $p < 0.01$), suggesting that aerosol mass loading did not exert a significant influence on amines uptake during the sampling period.

4 Summary and implications

Atmospheric WSON accounts for approximately one-quarter to one-third of TN in global wet deposition. Its chemical composition is highly complex, and its qualitative and quantitative characterization remains limited to date (Yu et al., 2020). The environmental behavior of WSON in the atmosphere (e.g., residence time, transformation pathways, and deposition mechanisms) is influenced by its F_{WSON} . However, relevant studies are still lacking, and most existing research has only measured particle-phase WSON. The CD, characterized by its unique topography and frequent stagnant, high-humidity meteorological conditions in winter, is one of the regions most severely affected by haze in China.

In this study, IGAC was employed to simultaneously measure, for the first time in CD, the concentration levels, chemical composition, and partitioning characteristics of wintertime gas- and particle-phase WSON. The evolution of WSON gas-particle partitioning across different pollution stages and its key driving factors were systematically elucidated. This study found that in high-humidity environments, ALWC plays a much stronger role in promoting the partitioning of WSON and amines into the particle phase than pH or T. During the PP, the particle-phase partitioning coefficients of amines increased significantly. Owing to their stronger basicity compared to NH_3 , amines can more effectively undergo acid-base reactions with acidic molecules such as sulfuric acid, forming more stable sulfuric acid–amine–water clusters, thereby substantially enhancing new particle formation rates (Almeida et al., 2013; Schobesberger et al., 2013; Yao et al., 2018). This indicates that amines are important

precursors for SOA formation during heavy pollution episodes. Thermodynamic model (ISORROPIA-II) and S-curve analysis revealed that ALWC is the dominant factor controlling the partitioning of NOCs. This finding suggests that in high-humidity polluted regions, aqueous-phase chemical processes contribute significantly to nitrogen-containing organic aerosol formation. Previous studies have similarly demonstrated that ALWC significantly promotes the secondary formation of particle-phase WSON (Yu et al., 2017; Xu et al., 2020; Leung et al., 2024). NH_4NO_3 contributes approximately 61% of ALWC, making it the largest contributor. Therefore, for CD, synergistic control of NO_x and NH_3 emissions can not only directly reduce nitrate in $\text{PM}_{2.5}$ but also indirectly inhibit the particle-phase enrichment of WSON and amines by reducing ALWC, thereby effectively interrupting the self-amplifying cycle of SNA formation. These findings provide a new scientific basis for haze mitigation in high-humidity polluted regions.

Data availability.

The data including meteorological data, gaseous pollutants, and major chemical components in $\text{PM}_{2.5}$ are freely available from Gehui Wang.

Supplement.

The supplementary material is published as separate file with this article.

Author contributions.

GW designed the experiment and supervised the research. KC, RL, LC, RL, BX, SL and JZ collected the samples. KC, RL, LC, CW and GW conducted the sample analysis. KC, RL, LC, CW and GW performed the data interpretation. KC and GW wrote the paper. ZL provided helpful assistance in writing the paper. All the authors contributed to the paper with useful scientific discussions.

Competing interests.

The contact author has declared that none of the authors has any competing interests.

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Table 1. Concentrations of major pollutants and meteorological parameters in Chengdu (CD) during winter.

Specie	Whole	Clean Period	Transition Period	Pollution Period
I. Gas pollutants ($\mu\text{g m}^{-3}$)				
NH ₃	12.3 ± 3.6 (2.4–31.2)	8.1 ± 2.7 (2.9–11.5)	11.8 ± 3.5 (3.4–31.2)	13.5 ± 3.0 (5.1–21.6)
WSON _g ^a	1.2 ± 0.9 (0.1–5.8)	0.7 ± 0.5 (0.1–1.7)	1.0 ± 0.7 (0.1–3.9)	1.4 ± 1.1 (0.1–5.8)
WSOC _g ^b	22.6 ± 9.8 (4.4–69.8)	16.6 ± 7.9 (4.4–31.3)	20.3 ± 8.8 (4.8–51.5)	25.5 ± 9.8 (6.2–69.8)
CH ₃ NH ₂	1.23 ± 0.36 (0.46–2.90)	0.84 ± 0.21 (0.46–1.26)	1.19 ± 0.37 (0.46–2.90)	1.35 ± 0.32 (0.59–2.32)
(CH ₃) ₂ NH	0.41 ± 0.03 (0.34–0.68)	0.38 ± 0.02 (0.34–0.42)	0.40 ± 0.03 (0.34–0.68)	0.42 ± 0.03 (0.35–0.53)
II. Major components of PM _{2.5} ($\mu\text{g m}^{-3}$)				
PM _{2.5}	76 ± 38 (7–199)	29 ± 11 (7–49)	55 ± 18 (9–101)	100 ± 33 (30–199)
SO ₄ ²⁻	4.9 ± 2.0 (1.5–11.9)	2.9 ± 1.2 (1.5–6.0)	3.7 ± 1.0 (1.6–6.7)	6.2 ± 1.7 (2.9–11.9)
NO ₃ ⁻	16.7 ± 8.3 (2.9–39.0)	6.4 ± 2.1 (2.9–11.0)	12.0 ± 4.0 (3.4–22.3)	22.3 ± 7.2 (6.4–39.0)
Cl ⁻	1.3 ± 1.0 (0.1–6.1)	0.5 ± 0.2 (0.1–0.9)	1.0 ± 0.7 (0.1–6.0)	1.8 ± 1.1 (0.3–6.1)
NH ₄ ⁺	7.2 ± 3.0 (1.4–15.5)	3.5 ± 0.8 (2.1–5.5)	5.6 ± 1.7 (1.4–10.3)	9.1 ± 2.6 (3.0–15.5)
Other ions ^c	2.2 ± 1.1 (0.8–9.1)	1.2 ± 0.2 (0.8–1.9)	1.7 ± 0.5 (0.9–3.5)	2.8 ± 1.2 (1.1–9.1)
OM ^d	21.1 ± 9.6 (6.7–47.4)	9.5 ± 1.9 (6.7–11.3)	15.1 ± 3.5 (6.9–21.0)	25.6 ± 9.3 (8.1–47.4)
WSON _p ^a	2.3 ± 1.4 (0.1–6.3)	1.0 ± 0.9 (0.1–4.3)	1.5 ± 0.8 (0.1–4.9)	3.0 ± 1.4 (0.3–6.3)
WSOC _p ^b	13.5 ± 7.2 (1.2–56.2)	8.1 ± 4.5 (1.4–21.0)	10.9 ± 6.6 (1.2–56.2)	16.3 ± 6.8 (2.7–54.8)
CH ₃ NH ₃ ⁺	0.45 ± 0.24 (0.03–1.27)	0.18 ± 0.06 (0.10–0.34)	0.33 ± 0.13 (0.03–0.67)	0.61 ± 0.23 (0.15–1.27)
(CH ₃) ₂ NH ₂ ⁺	0.09 ± 0.05 (0.02–0.34)	0.06 ± 0.01 (0.04–0.09)	0.07 ± 0.04 (0.02–0.29)	0.12 ± 0.04 (0.05–0.34)
III. Gas-particle partitioning coefficients of WSON, WSOC, ammonium (AM), methylamine (MA), dimethylamine (DMA)				
F _{WSON} ^e	0.70 ± 0.14 (0.28–0.98)	0.62 ± 0.17 (0.29–0.88)	0.66 ± 0.14 (0.28–0.90)	0.73 ± 0.13 (0.42–0.98)
F _{WSOC} ^e	0.36 ± 0.09 (0.10–0.81)	0.32 ± 0.10 (0.19–0.65)	0.34 ± 0.09 (0.13–0.75)	0.39 ± 0.09 (0.10–0.81)
F _{AM} ^e	0.36 ± 0.08 (0.14–0.52)	0.31 ± 0.10 (0.19–0.52)	0.32 ± 0.07 (0.14–0.49)	0.39 ± 0.05 (0.20–0.49)
F _{MA} ^e	0.25 ± 0.08 (0.07–0.44)	0.18 ± 0.06 (0.09–0.31)	0.21 ± 0.06 (0.07–0.39)	0.30 ± 0.06 (0.12–0.40)
F _{DMA} ^e	0.18 ± 0.06 (0.04–0.45)	0.13 ± 0.02 (0.10–0.19)	0.15 ± 0.06 (0.04–0.42)	0.21 ± 0.05 (0.12–0.45)
IV. Meteorology parameters and aerosol properties				
T (°C)	6.4 ± 3.6 (-2.4–16.5)	6.2 ± 2.7 (0.2–12.3)	6.4 ± 3.4 (-1.3–15.4)	6.4 ± 3.9 (-2.4–16.5)
RH (%)	78 ± 19 (20–100)	78 ± 22 (26–99)	76 ± 19 (20–100)	79 ± 18 (29–100)
WS (m/s)	0.5 ± 0.4 (0.0–1.7)	0.6 ± 0.3 (0.0–1.4)	0.6 ± 0.4 (0.0–1.7)	0.3 ± 0.3 (0.0–0.9)
pH of PM _{2.5}	4.4 ± 0.2 (3.4–5.0)	4.2 ± 0.3 (3.4–4.5)	4.4 ± 0.2 (3.6–4.9)	4.4 ± 0.2 (3.8–5.0)
ALWC ^f	64.0 ± 71.5 (0.5–422.2)	34.0 ± 35.8 (1.2–130.5)	44.4 ± 49.2 (0.5–215.9)	87.6 ± 85.1 (1.6–422.2)

^a WSON_g is the gas-phase organics, WSON_p is the organics in PM_{2.5}, ($\mu\text{gN m}^{-3}$). ^b WSOC_g is the gas-phase organics, WSOC_p is the organics in PM_{2.5}, ($\mu\text{gC m}^{-3}$). ^c Other ions are the sum of NO₂⁻, F⁻, K⁺, Na⁺, Mg²⁺, and Ca²⁺. ^d Organic matter: OM = OC × 1.6. ^e $F_p = C_p / (C_g + C_p)$. ^f ALWC: aerosol liquid water content of PM_{2.5}.

Table 2. Concentrations Regional comparison of key physicochemical parameters (PM_{2.5}, T, RH, pH, ALWC) and the water-soluble organic nitrogen partitioning coefficient (F_{WSON}) in atmospheric aerosols at Chengdu (CD), Shanghai (SH), and Xianghe (XH).

Specie	City	Whole	Clean Period	Transition Period	Pollution Period
F _{WSON}	CD	0.70 ± 0.14 (0.28–0.98)	0.62 ± 0.17 (0.29–0.88)	0.66 ± 0.14 (0.28–0.90)	0.73 ± 0.13 (0.42–0.98)
	SH	0.49 ± 0.26 (0.01–0.99)	0.37 ± 0.21 (0.01–0.93)	0.52 ± 0.29 (0.01–0.99)	0.64 ± 0.23 (0.15–0.97)
	XH	0.43 ± 0.23 (0.03–0.99)	0.33 ± 0.18 (0.03–0.69)	0.46 ± 0.20 (0.04–0.78)	0.48 ± 0.25 (0.05–0.99)
PM _{2.5}	CD	76 ± 38 (7–199)	29 ± 11 (7–49)	55 ± 18 (9–101)	100 ± 33 (30–199)
	SH	54 ± 40 (4–225)	23.9 ± 13.5 (4.0–91.0)	50 ± 26 (5–119)	100 ± 38 (35–225)
	XH	63 ± 56 (0–239)	18 ± 15 (0–69)	61 ± 27 (4–123)	118 ± 52 (1–239)
T	CD	6.4 ± 3.6 (-2.4–16.5)	6.2 ± 2.7 (0.2–12.3)	6.4 ± 3.4 (-1.3–15.4)	6.4 ± 3.9 (-2.4–16.5)
	SH	7.8 ± 3.9 (-1.2–19.9)	8.6 ± 4.4 (-1.2–19.9)	7.6 ± 3.4 (-0.91–16.6)	7.1 ± 3.8 (0.2–17.7)
	XH	-2.0 ± 5.2 (-14.2–9.8)	-5.3 ± 4.6 (-14.2–6.5)	-0.8 ± 3.9 (-7.6–9.8)	0.9 ± 4.3 (-8.3–9.1)
RH	CD	78 ± 19 (20–100)	78 ± 22 (26–99)	76 ± 19 (20–100)	79 ± 18 (29–100)
	SH	83 ± 14 (42–100)	88 ± 11 (53–100)	83 ± 14 (45–100)	77 ± 15 (42–98)
	XH	54 ± 21 (16–92)	47 ± 20 (24–91)	60 ± 21 (16–92)	57 ± 19 (18–92)
pH	CD	4.4 ± 0.2 (3.4–5.0)	4.2 ± 0.3 (3.4–4.5)	4.4 ± 0.2 (3.6–4.9)	4.4 ± 0.2 (3.8–5.0)
	SH	4.2 ± 0.3 (3.2–4.8)	4.0 ± 0.3 (3.3–4.8)	4.2 ± 0.3 (3.2–4.6)	4.4 ± 0.2 (3.8–4.8)
	XH	5.0 ± 0.9 (1.7–8.9)	4.5 ± 0.8 (1.7–6.0)	5.2 ± 0.6 (4.6–7.6)	5.4 ± 0.7 (3.8–8.9)
ALWC	CD	64.0 ± 71.5 (0.5–422.2)	34.0 ± 35.8 (1.2–130.5)	44.4 ± 49.2 (0.5–215.9)	87.6 ± 85.1 (1.6–422.2)
	SH	51.6 ± 51.4 (2.0–329.4)	23.2 ± 22.4 (2.0–130.1)	45.2 ± 37.6 (5.4–165.8)	80.4 ± 64.3 (6.6–329.4)
	XH	38.7 ± 73.6 (0.0–439.8)	9.4 ± 17.7 (0.0–103.3)	33.9 ± 36.2 (0.1–129.6)	73.2 ± 106.2 (0.0–439.8)

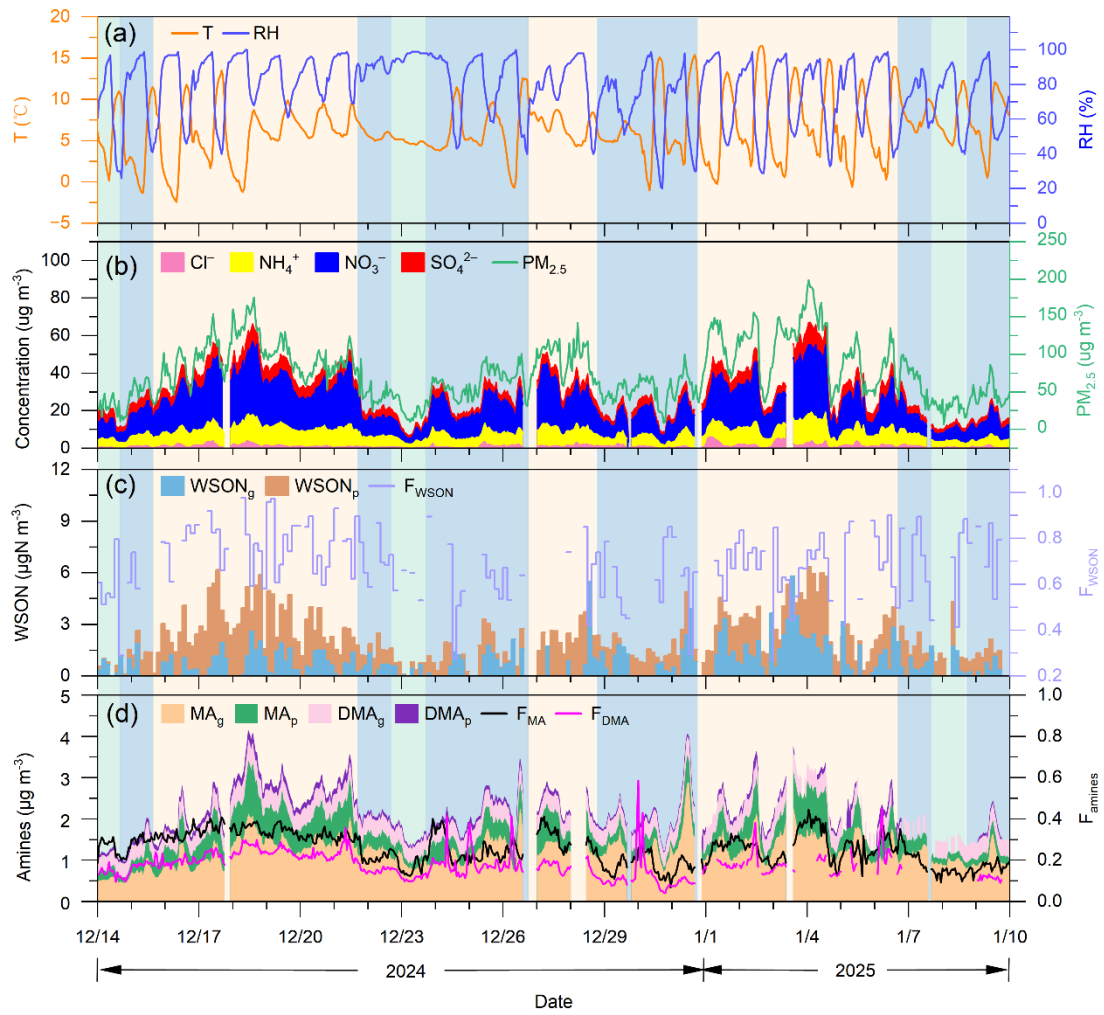


Figure 1. Temporal variation of major pollutants in Chengdu, China, during the winter of 2024 – 2025. $F_p = C_p / (C_g + C_p)$, C_p and C_g are the concentrations of WSON and methylamine (MA) or dimethylamine (DMA) in the particle phase and gas phase, respectively. Green shadows represent clean periods with $PM_{2.5} \in [0, 35) \mu g m^{-3}$. Blue shadows represent transition periods with $PM_{2.5} \in [35, 75) \mu g m^{-3}$. Orange shadows represent pollution periods with $PM_{2.5} \in [75, 200) \mu g m^{-3}$.

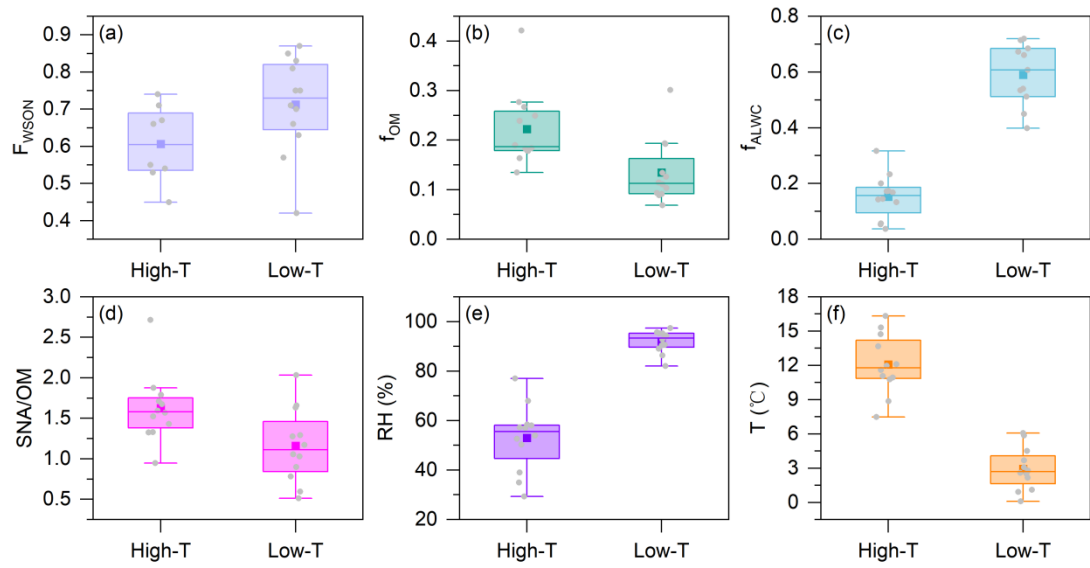


Figure 2. Analysis of the parting behavior of WSON under different temperature conditions during the pollution period (2025.1.2—2025.1.6).

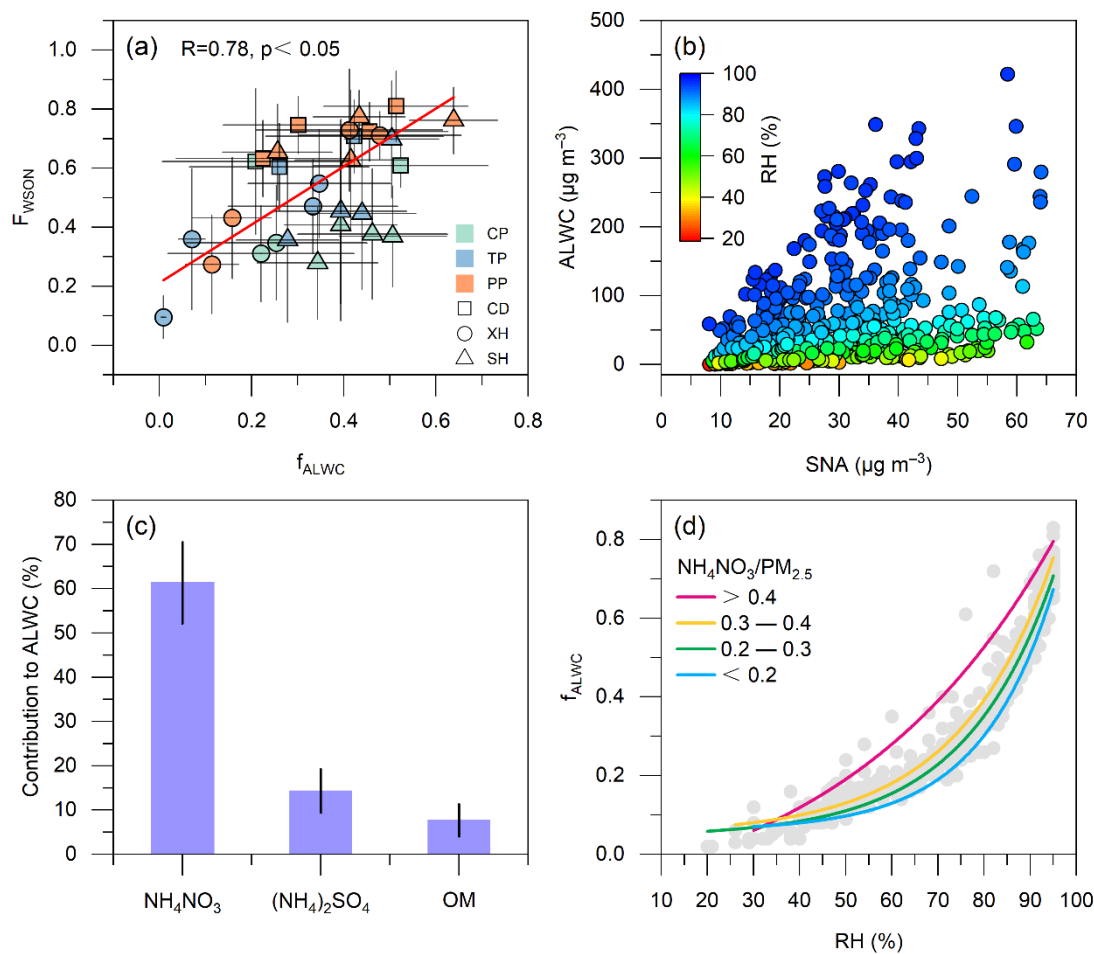


Figure 3. Chemical drivers of ALWC and its associations with WSON. (a) correlation between the relative abundance of ALWC to $PM_{2.5}$ ($f_{ALWC} = ALWC / (ALWC + PM_{2.5})$) and F_{WSON} . Symbols represent different geographical regions (CD: Chengdu; XH: Xianghe; SH: Shanghai), colors denote different pollution periods (PP: polluted period; TP: transition period; CP: clean period). (b) relationship between ALWC and SNA concentrations during the whole campaign of CD, colored by RH. (c) mean percentage contribution (± 1 SD) of different chemical components (NH_4NO_3 , $(NH_4)_2SO_4$, OM) to ALWC during the whole campaign of CD. (d) variation of f_{ALWC} with RH under different relative abundances of NH_4NO_3 to $PM_{2.5}$ during the whole campaign in the CD (solid lines represent the fitted curves, and gray dots represent the raw data).

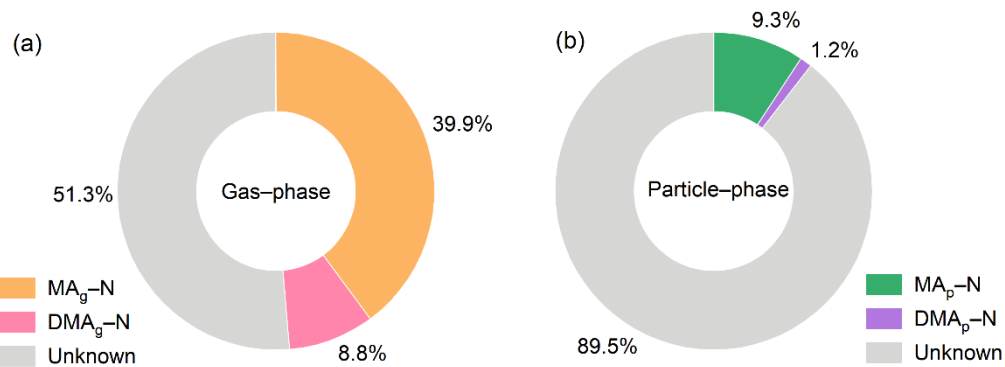


Figure 4. Contributions of amines N concentrations to gas-phase (a) and particulate-phase (b) WSON.

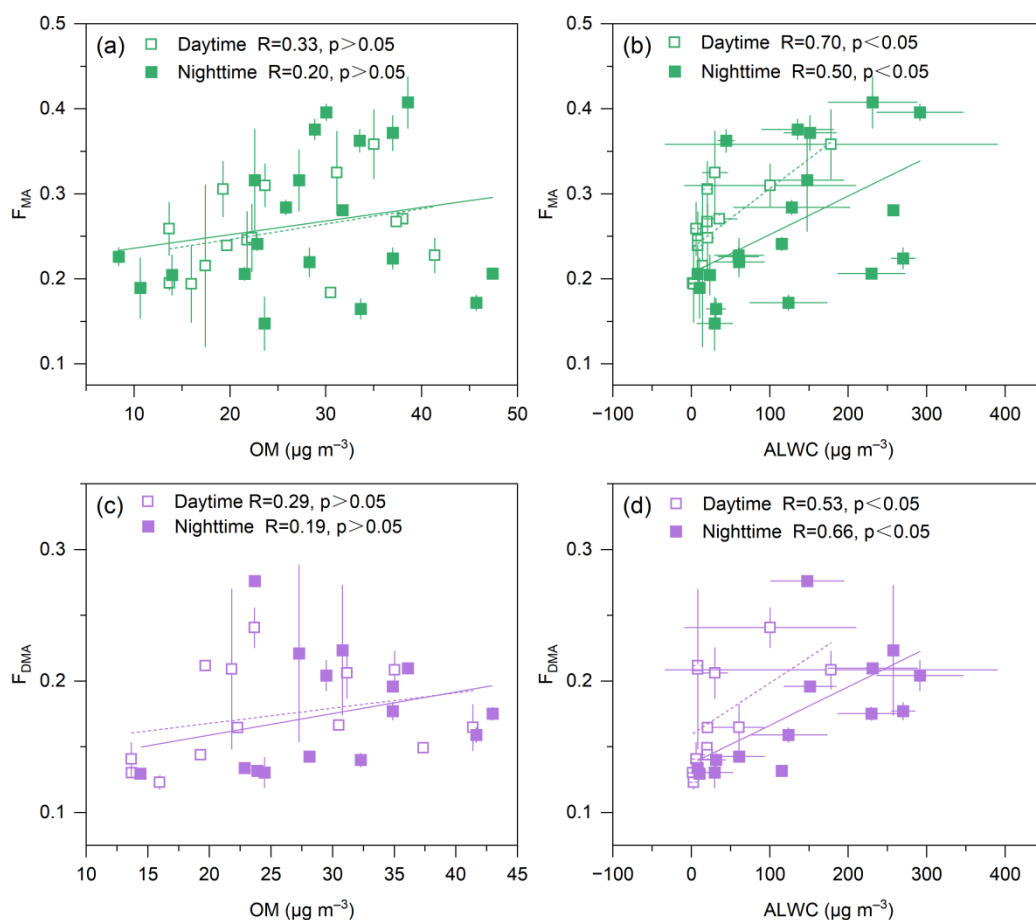


Figure 5. Linear regression analysis for F_p with OM and ALWC under the daytime and nighttime during the pollution period (2025.1.2—2025.1.6).

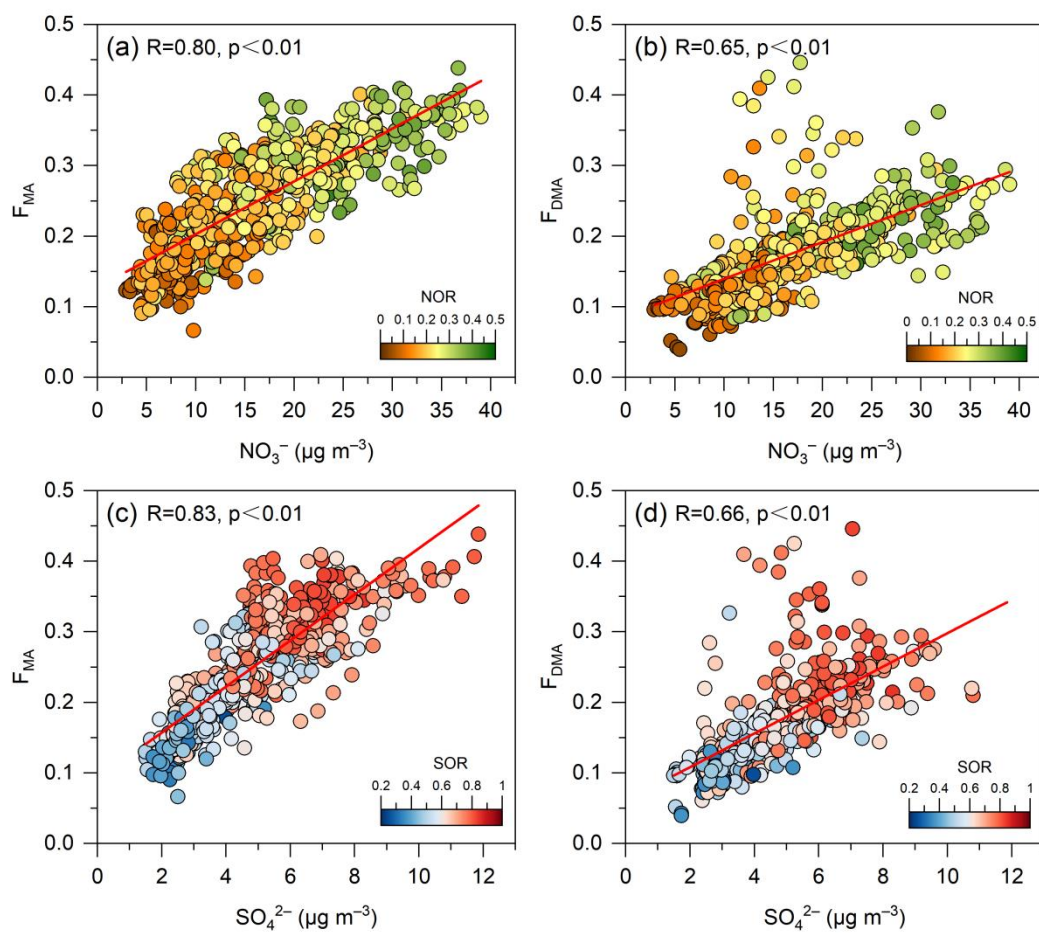


Figure 6. Linear regression analysis for the gas-aerosol partitioning coefficient (F_p) of amines with inorganic salts.

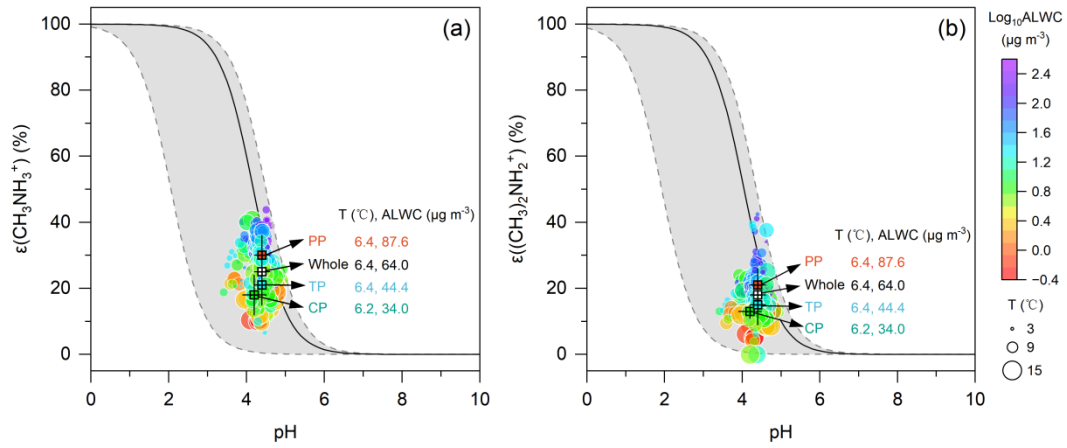


Figure 7. Analytically calculated S-curves of $\varepsilon(\text{CH}_3\text{NH}_3^+)$ and $\varepsilon((\text{CH}_3)_2\text{NH}_2^+)$ and ambient data during the entire observation period. Circles represent the measured data for methylamine (a) and dimethylamine (b), respectively. For the analytically calculated S-curve, we used $\gamma_{\text{CH}_3\text{NH}_3^+} = \gamma_{(\text{CH}_3)_2\text{NH}_2^+} = 0.338$, $\gamma_{\text{CH}_3\text{NH}_2} = \gamma_{(\text{CH}_3)_2\text{NH}} = 1.240$, and $\gamma_{\text{H}^+} = 0.836$ (E-AIM model predicted). The black solid line is the S-curve calculated based on the average temperature and ALWC (6.4 ± 3.6 °C and 63.99 ± 71.53 $\mu\text{g m}^{-3}$, respectively). The gray dashed line represents the S-curve calculated using 1 standard deviation of the average temperature and ALWC. Due to the value of ALWC minus 1 standard deviation being less than 0, the minimum ALWC value (i.e., $W_i = 0.48$ $\mu\text{g m}^{-3}$) was used in the calculation. Different colors and blocks represent different sampling periods and their corresponding average values. CP: clean period, TP: transition period, PP: pollution period, and Whole: whole campaign period.