Measurement report: Impact of domestic heating on dust deposition sources in hyper-arid Qaidam Basin, northern Qinghai-Xizang Plateau

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Highlights

1. The spatiotemporal distribution of carbonaceous aerosols in the basin's atmospheric dust deposition was analyzed using various carbon indicators.

2. Domestic heating significantly increased associated pollutants in rural atmospheric dust deposition.

3. The basin's unique energy structure suggests its emissions and deposition could affect regional glaciers on the Qinghai-Xizang Plateau.

Abstract

Given the unique energy structure of the Qaidam Basin (QDB), this study systematically reveals the spatiotemporal variations in the chemical composition of atmospheric dust deposition and clarifies the key contributions of coal and biomass burning to carbonaceous aerosols, as well as their potential impacts on the Qinghai-Xizang Plateau (QXP) and global atmospheric systems. Monthly dust deposition samples were collected at six sites in the southern QDB between 2020 and 2023. Results indicated heightened carbon emissions and the higher char/soot ratio during the heating period (HP, 5.06 ± 4.08) than the non-heating period (NHP, 4.42 ± 3.09), indicating intensified seasonal solid fuel consumption. Spatially, the organic carbon (OC) and elemental carbon (EC) ratio was significantly lower in urban (3.97 ± 2.04) than that in rural areas (10.99 ± 10.00). Char-EC dominated EC (80.44%), especially in urban areas (85.00%), while secondary organic carbon (SOC) dominated OC (72.61%), particularly in rural areas (87.32%). The coal combustion (15.19%) and biomass burning (33.55%) as major contributors in rural areas, strongly associated with domestic heating, whereas urban dust predominantly originated from traffic (46.83%) and industrial emissions (16.41%). Coal consumption in QDB was greater during the HP relative to other dust sources on the QXP leads to increased atmospheric pollutant emissions, which may accelerate regional glacier melting. Consequently, integrating QDB carbonaceous aerosols into future environmental policies and climate models for the QXP is essential. This study provides a reference for investigating carbonaceous aerosols in climatically similar hyper-arid basins with intensive human activity and salt lake regions.

Keywords: Qinghai-Xizang Plateau; Qaidam Basin; Biomass burning; carbonaceous elements; atmospheric dust deposition.

Short summary

This study collected dust samples from six sites in the Qaidam Basin over three years to investigate the impact of domestic heating on atmospheric dust in hyper-arid region. Our results indicate that rural dust is significantly influenced by heating, particularly from coal and biomass burning, which accounts for over 70% of total sources. The unique energy structure here has resulted in distinct environmental effects from the emitted carbonaceous aerosols and useful for similar dry areas.

1. Introduction

Atmospheric dust, a key component of particulate matter (PM), comprises solid particles, typically ranging in size from below 1 μm to 100 μm, which may become airborne depending on their origin, physical characteristics and ambient conditions (Xu, 2014). Within this size spectrum, larger particles (10-100 μm) that settle under gravity are defined as atmospheric dust deposition. Notably, during long-distance transport, these coarse particles can undergo fragmentation into fine particles ($PM_{2.5}$) and subsequently actively participate in atmospheric chemical and climatic processes (Noll and Fang, 1989). As the dominant natural component of PM, atmospheric dust deposition serves not only a crucial indicator of regional air quality but also a key biogeochemical process linking the atmosphere, cryosphere, hydrosphere, and biosphere, with global-scale influences that significantly shape environmental and climate systems (Feng et al., 2019).

Arid and semi-arid regions are the primary global sources of atmospheric dust (Griffin et al., 2002; Schepanski, 2018). Long-distance transport of this dust via wind currents exerts multi-faceted impacts on the environment and human society. A particularly critical effect is the alteration of the cryosphere: dust deposition on snow and ice lowers surface albedo and modifies ice crystal structure, thereby accelerating glacier and snowpack melt (Tuzet et al., 2017). This process disrupts regional snow energy balance, which is directly linked to glacier retreat and water resource security. Dust deposition also influences the atmospheric energy budget by attenuating solar radiation reaching the surface and participates in the global carbon cycle through biogeochemical pathways, such as delivering nutrients to oceans and affecting marine primary productivity (Mahowald et al., 2009; Parajuli et al., 2022). For human health, harmful components associated with dust can induce respiratory and cardiovascular diseases and even damage cellular DNA (Shahram et al., 2016). For terrestrial ecosystems, the shading effect of dust on leaves can inhibit plant photosynthesis and reduce biological productivity.

Given the complexity and regional variability of atmospheric dust impacts, identifying dust sources (source apportionment) is fundamental for understanding its environmental behavior and effects. Recent advancements in understanding PM characteristics, particularly chemical composition (e.g. water-soluble ions, carbonaceous components, and trace elements) and source apportionment, have been driven by the integrated application of methodological tools. These

receptor models include principal component analysis (PCA), chemical mass balance (CMB), and positive matrix factorization (PMF) models, are used to quantify source contributions. Specifically, the PMF model mathematically deconstructs the chemical composition matrix of ambient samples to achieve this. Additionally, multivariate statistical approaches like backward trajectory simulations (e.g., the HYSPLIT model) trace air mass transport pathways to identify potential source regions (Lai et al., 2016; Yao et al., 2016; Zhang et al., 2015a). This multi-method approach has greatly enhanced the precision of dust deposition source analysis. For instance, PMF analysis of atmospheric dust in urban areas such as Lanzhou, Taiyuan, and Jinan have identified diverse sources, including coal combustion, industrial emissions, construction dust, windblown dust, vehicle emissions, and resuspended road dust. Seasonal variations indicate that coal combustion during the domestic heating period and regional meteorological conditions significantly influence dust deposition (Hu and Liu, 2022; Chen et al., 2024; Yang et al., 2024; Zhang et al., 2022). These findings underscore the urgency of region-specific pollution control strategies. When considering the global scale, arid and semi-arid regions are unequivocally the dominant sources, contributing over 60% of the global atmospheric dust flux (Zan et al., 2025). Therefore, a comprehensive investigation of atmospheric dust processes in these key regions, encompassing emission intensity, physicochemical properties, transport pathways, and environmental effects, is indispensable for elucidating global dust cycle mechanisms and assessing their profound impacts on the cryosphere (e.g., glacial melting) and regional climate.

The Qinghai-Xizang Plateau (QXP) often referred to as the “Roof of the World” due to its immense elevation, plays a crucial role in regulating the regional and global climate by altering large-scale atmospheric circulation. Its vast glaciers and snow cover influence regional energy balance through the albedo effect, and as the source of many major Asian rivers, it is known as the “Asian Water Tower” (Liu et al., 2019; Liu et al., 2020b). However, rapid glacier retreat on the plateau poses risks to the Asian hydrological cycle and the monsoon system, with potential adverse impacts if unchecked (Luo et al., 2020). Beyond climate warming and moistening, black carbon (BC), a light-absorbing carbonaceous aerosol component emitted from incomplete combustion processes of solid and liquid fuels during household cooking, heating, and coke production (Bond et al., 2013), significantly accelerates glacial melt. By depositing on ice, BC reduces surface albedo, enhances radiative absorption and thus raises it temperatures, promoting glacial melt (Bond and

Bergstrom, 2006; Chen et al., 2015). Notably, biomass burning in South and Central Asia during winter serves as a major source of BC, further exacerbating glacier decline on the plateau (Zhang et al., 2015b; Zheng et al., 2017; Xu et al., 2018b). In addition to these external sources, local dust sources within the QXP itself remain significant. Among these, the Qaidam Basin (QDB) in the northeastern plateau is a particularly important contributor, identified as a key dust source for the plateau (Wei et al., 2017; Zheng et al., 2021) and a critical, unique arid dust source area.

The QDB, known as the "Treasure Basin" of the QXP, is rich in mineral resources (e.g., copper, iron, and tin) as well as abundant oil and gas reserves. It serves as a key economic development zone in northwest China, accounting for approximately 30% of the plateau's industrial and agricultural output despite comprising only about 8% of its registered population (Fu, 2023). Extensive resource extraction has rendered its ecosystem fragile (Li and Sha, 2022), exacerbating the impact of atmospheric pollution. The region is also highly sensitive and vulnerable to climate change, with severe and extreme vulnerability zones covering 45.98% of its area (Xu et al., 2024). Unlike South Asia, Central Asia, and Xizang, where biomass fuels dominate, the QDB relies primarily on distinct mix of coal (about 60%) and biomass fuels like yak dung, firewood (about 35%) for winter domestic heating, reflecting a unique energy structure (Liu et al., 2008; Xiao et al., 2015; Behera et al., 2015; Kerimray et al., 2018; Jiang et al., 2020; Shen et al., 2021). The combustion of coal releases significant pollutants, including light-absorbing aerosols like BC and brown carbon (BrC) (Munawer, 2018; Ye et al., 2020; Zhou et al., 2025). Brown carbon refers to organic compounds that can absorb light, particularly at shorter wavelengths, resulting in a reddish orange or brown appearance (Donahue, 2018). Both BC and BrC can influence solar radiation absorption and cloud properties, exerting a positive radiative forcing on climate (Ramanathan and Carmichael, 2008; Bond et al., 2013). When deposited on snow and ice, they further accelerate glacial melt (Qian et al., 2015; Kang et al., 2020a). Additionally, BC and BrC are significant contributors to global warming and can exacerbate adverse health effects by carrying toxic components (Ramanathan and Carmichael, 2008; Shrivastava et al., 2017). Notably, QDB's widespread use of yak dung as a fuel, a practice less common in other coal-intensive heating regions of northern China, releases pollutants like carbon monoxide (CO), volatile organic compounds (VOCs), and polycyclic aromatic hydrocarbons (PAHs), further affecting local air quality and health (Zhang et al., 2022). Consequently, we posit that seasonal carbon emissions in QDB, particularly

during winter domestic heating, could exert a unique influence on the climate and ecological stability of the QXP.

The QDB, as a representative arid region with intensive human activity, exhibits climatic and environmental conditions comparable to other hyper-arid basins (e.g., the Tarim and Junggar Basins in Xinjiang, the Great Basin in the United States) and high-altitude salt lake regions (e.g., Uyuni in Bolivia, Atacama in Chile). These regions are characterized by low precipitation, rich mineral resources subject to significant anthropogenic impact, and abundant salt lakes. Research in the Tarim and Junggar Basins has predominantly focused on dust events, their sources, and associated gas emissions (Gao and Washington, 2010; Liu et al., 2016b; Filonchyk et al., 2018; Yu et al., 2019; Zhou et al., 2023). In the Great Basin, studies largely address ozone and dust sources (Hahnenberger and Nicoll, 2012; Vancuren and Gustin, 2015; Miller et al., 2015). Research on salt lake atmospheres has predominantly focused on high-salinity dust emissions resulting from lakebed desiccation due to resource extraction (Löw et al., 2013; Gholampour et al., 2015; Moravek et al., 2019; Christie et al., 2025), with limited research on atmospheric carbon components, their sources, and environmental impacts. Therefore, this research aims to investigate atmospheric carbonaceous aerosols in arid basins with intensive human activity and climates comparable to the QDB, as well as in salt lakes environments.

From January 2020 to March 2023, monthly dust deposition samples were collected at six urban and rural monitoring sites in the southern QDB. Samples were categorized into two seasonal periods: the domestic heating period (HP) and the non-domestic heating period (NHP). Measured parameters included dust deposition flux, soluble ions, trace elements, and key carbonaceous components. The objectives of this study were: (1) to clarify the variation trends of carbonaceous components in atmospheric dust deposition under the unique energy structure of the QDB, and to quantify the contribution of domestic heating; and (2) to identify the major sources of atmospheric dust deposition in the basin and to evaluate their associated environmental impacts. To achieve these aims, we applied OC/EC and char/soot ratios, the HYSPLIT trajectory, and PMF model for source apportionment. Furthermore, these findings offer a scientific basis and reference for examining atmospheric carbonaceous aerosols in arid basins with similar climates and human activities to the QDB, as well as in salt lake regions.

2. Materials and methods

2.1 Sampling

The QDB, situated in the northeastern part of the QXP, is bordered by the Altyn-Tagh, Kunlun, and Qilian mountains, making it one of China's largest intermontane basins (Zhang, 1987). With an average elevation of 3,000 m, the basin features an extremely arid climate characterized by less than 20 mm of annual precipitation in the northwestern region, while evaporation rates exceed 2,000 mm (Feng et al., 2022). The QDB is rich in salt lakes, non-ferrous metals, and hydrocarbon resources, with significant coal deposits. As a major salt lake resource area, it hosts 33 lakes, including Qarhan, Dachaidan, and Caka Salt Lake, and faces notable conflicts between resource extraction and ecological preservation. Agriculturally, it cultivates crops like goji berries, quinoa, and forage grass, and hosts China’s largest resource-rich circular economy pilot zone. The permanent population of the basin is approximately 400,000, primarily using coal, yak dung, and firewood for domestic heating (Jiang et al., 2020). Additionally, annual tourism peaks from May to September, attracting around 17 million visitors (Qinghai Statistical Yearbook, 2023), which likely amplifies atmospheric pollutant emissions.

From January 2020 to March 2023, monthly dust deposition samples were collected from six monitoring stations in the southern QDB. The stations included Xiao Zaohuo (XZH), Golmud (GEM), Da Gele (LTC), Nuo Muhong (NMH), Balong (BLX), and Dulan station (DLX) (Figure 1). Dry deposition collection employed the glass ball method using Marble dust collector (MDCO) designed dust collection cylinders (Sow et al., 2006). The MDCO sampler operates on the principle of gravitational settling. It uses a collection vessel with a known opening to capture particulate matter that settles naturally from the air. In field measurements, surrogate surfaces or deposition traps are commonly employed to better quantify atmospheric deposition. These surfaces mimic the original ground surface, are easy to deploy, and can be integrated without significantly disturbing airflow. Examples include glass beads (Ganor, 1975; Offer et al., 1992), moist filter paper (Goossens and Offer, 1993), plastic surfaces (Gregory, 1961), and water or antifreeze solutions (Smith and Twiss, 1965). The MDCO deposition vessel is based on the original concept of Ganor (1975), using glass beads as a surrogate collection surface. Due to their very low microscopic roughness, the beads help prevent the resuspension of particles once they have been captured (Goossens and Offer, 1993).

The stainless steel collection device (50×30×30 cm) contained a plastic sieve container of identical dimensions, with the sieve base positioned 10 cm above the opening and perforated with 0.5 cm diameter holes (Figure S1). To minimize dust resuspension during high wind events (Qian and Dong, 2004), two layers of 16 mm glass balls were placed within the sieve container. A high-density polyethylene bag was attached to the base for sample collection. According to Sow et al (2006), the collection efficiency of the MDCO decreases with increasing wind speed, dropping below 20% when wind speed exceeds 3 m/s, and it preferentially collects the finer fraction of atmospheric dust particles ranging from 10 to 31 μm in size (Chow, 1995). Furthermore, collection efficiency decreases significantly as the size of dust-fall increases. Based on our findings, more than 50% of the collected dust particles are below 40 μm (Figure S2), whereas the mean particle size of atmospheric dust deposition reported in other studies is approximately 80 μm (Lin et al., 2022a; Lin et al., 2022b). This confirms that the MDCO sampler preferentially collects relatively finer particles. Consequently, this study may place greater emphasis on analyzing the chemical activity of the finer fraction of atmospheric dust, thereby enhancing the detection sensitivity for components such as secondary ions and carbonaceous species in deposited dust. However, this particle-size selectivity introduces a sampling bias that can amplify inter-site differences in deposition flux between sites, complicate local source apportionment, strengthen signals from distant sources (e.g., combustion and vehicle emissions), and weaken the contribution from local, coarse-particle-dominated sources (e.g., resuspended dust and construction dust). These factors may affect the accuracy of both deposition flux estimates and source apportionment. Future research should enhance data comparability and the reliability of ecological assessments through efficiency correction and the combined application of multiple methods. The collection efficiency at each site was calculated based on local dust grain-size distribution and wind speed; details are provided in Supplementary Text S1. This type of collector has been widely used in many studies to evaluate local dust conditions (Abdollahi et al., 2021; Barjoee et al., 2021; Alzahrani et al., 2024).

In this study, dust samples were collected monthly, with each sampling period lasting 30 or 31 days. The installation height and environment of the samplers are provided in Table S1. To ensure only dry dust was collected, collection devices were manually-operated covered during rain or snowfall. A total of 37, 39, 23, 30, 16, and 29 samples were obtained from XZH, GEM, LTC, NMH, BLX, and DLX stations, respectively, and no duplicate samples were collected during the sampling

period. Laboratory protocols incorporated biannual analyses with negative controls and appropriate control samples. As continuous dust monitoring commenced in 2020, site blanks were evaluated during initial sampling. Stations were classified as Urban (GEM, NMH, DLX) and Rural (XZH, LTC, BLX) based on location characteristics. Consistent with the cold-arid climate in QDB, the HP was defined as October-April, while the NHP spanned May-September.

Materials such as plant remnants, microfauna, and bird droppings were removed from the sample bags with tweezers. The samples were then measured on a balance (0.0001 g) to determine the total dust deposition flux (DF) (Eq. 1) (Yu et al., 2016):

$$M=\frac{m\times 30}{S\times K}, \quad (1)$$

where M is dust deposition [g/(m²·30d)]; m is the sample mass (g); S is the area of the dust collection device (m²); and K is the actual number of sampling days per month (d).

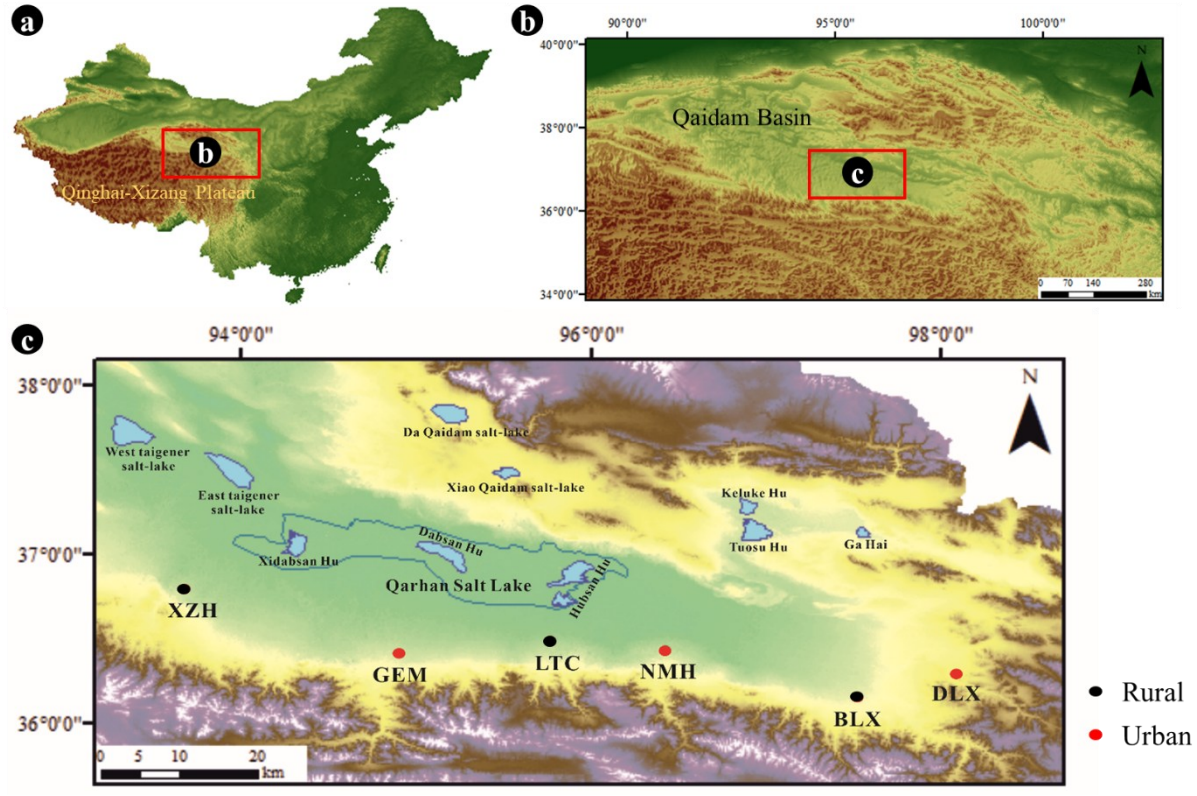

Figure 1. Spatial distribution of monitoring stations in the southern Qaidam Basin. Urban stations (red) and rural stations (black) are labeled as follows: XZH (Xiao Zaohuo), GEM (Golmud), LTC (Da Gele), NMH (Nuo Muhong), BLX (Balong), DLX (Dulan).

2.2 Water-soluble inorganic ions

A 100 mg sample was weighed and transferred into a 250 mL bottle. The mixture underwent ultrasonic extraction for 20 minutes to ensure complete solubilization. The resulting supernatant was then filtered through a 0.45 μm filter for analysis. Based on preliminary experimental results, the concentrations of major ions (K^+, Na^+, Mg^{2+}, and Ca^{2+}) were measured using Inductively Coupled Plasma Optical Emission Spectrometer (ICP-OES, NexIon 2000). Anions (Cl^-, SO_4^{2-}, and NO_3^-) were quantified using ion chromatography (IC). To ensure measurement accuracy, samples were organized in sets of twenty, with one randomly selected sample from each group serving as a replicate, achieving an error margin of less than 10%. The detection limits for the various components were as follows: K^+ (0.0560 mg/L), Na^+ (0.0100 mg/L), Ca^{2+} (0.0037 mg/L), Mg^{2+} (0.0390 mg/L), SO_4^{2-} (0.0090 mg/L), NO_3^- (0.0125 mg/L), Cl^- (0.0100 mg/L). All standard solutions employed in the analysis were sourced from the National Standard Material Center.

2.3 Trace element analysis

According to the Chinese State Standard "Ambient air and waste gas from stationary sources emission-determination of metal elements in ambient particles" (HJ 777-2015), the concentrations of elements such as iron (Fe), aluminum (Al), silicon (Si), titanium (Ti), copper (Cu), cadmium (Cd), chromium (Cr), manganese (Mn), nickel (Ni), zinc (Zn), lead (Pb), and vanadium (V) were quantified using Inductively Coupled Plasma Mass Spectrometry (ICP-MS) and ICP-OES. A dust sample weighing 0.100 g was placed in a Teflon cup, to which 20.0 mL of a nitric acid-hydrochloric acid digestion solution was added. The sample was heated to reflux at 100 ± 5°C for 2 h under a watch glass, then cooled. Following this, the inner walls of the cup were rinsed with water, and approximately 10 mL of water was added, allowing the mixture to stand for 30 minutes for extraction. The extract was then filtered into a 100 mL volumetric flask and diluted to volume with distilled water for analysis. In cases where organic matter content was high, an appropriate amount of hydrogen peroxide was introduced during digestion to decompose the organic materials. Prior to sample analysis, the system was flushed with a rinse solution until the blank intensity value reached a minimum, and samples were analyzed only after the signal stabilized. If the concentration of any element exceeded the calibration range, the sample was diluted and reanalyzed.

2.4 Carbon analysis

This study utilized a combination of wet chemical treatment and thermal/optical reflection (TOR) to analyze carbon elements in dust deposition (Han et al., 2007b; Han et al., 2007a; Han et al., 2016). Dust samples were digested stepwise to remove inorganic materials: first samples were treated with 10 mL of 2 N hydrochloric acid (HCl) for 24 h at room temperature to dissolve carbonates and partial metals, followed by centrifugation (4500 rpm, 12 min) to separate the residue; then with 15 mL of a 1:2 (v/v) mixture of 6 N HCl and 48% hydrofluoric acid (HF) for 24 h at room temperature to dissolve silicates and residual metals; and finally with 15 mL of 4 N HCl for 24 h at 60 °C to remove minerals such as fluorite formed during demineralization. After each step, the mixture was centrifuged, and the supernatant was collected. The solid residue was diluted with 200 mL of deionized water and filtered through a pre-combusted quartz fiber filter (Whatman, 450°C for 4 h, diameter 47 mm) (Han et al., 2007a). This method has been widely applied to measure OC and EC contents in lake sediments and urban soils (Han et al., 2009; Khan et al., 2009; Han et al., 2011). Studies have shown that the EC collection efficiency of this method is approximately 99.6% (Zhan et al., 2013). However, because OC includes polar, basic, and mineral-bound compounds that may dissolve in HCl/HF and be lost during filtration, the supernatant from all three acid-digestion steps was collected and analyzed for water-soluble OC using a carbon-nitrogen analyzer (Shimadzu, TOCN-4200, Japan). Total OC was obtained by summing the dissolved OC and the particulate OC measured by TOR.

The filtered samples were air-dried and analyzed for carbon content using a DRI 2001 thermal/optical carbon analyzer (Atmoslytic Inc., Calabasas, CA) at the Institute of Earth Environment, Chinese Academy of Sciences, adhering to the Interagency Monitoring of Protected Visual Environments (IMPROVE) protocol. A 0.544 cm^2 disc was extracted from the filter and placed in a quartz boat for analysis. During the carbon analysis, the samples were initially heated in a 100% helium atmosphere, resulting in the production of four organic carbon (OC) fractions (OC1, OC2, OC3, and OC4) at four different temperature levels (140, 280, 480, and 580°C). The atmosphere was subsequently changed to 2% O_2/98% He, generating three elemental carbon (EC) fractions (EC1, EC2, and EC3) at three temperatures (580, 740, and 840 °C). Volatile carbon underwent carbonization in an anaerobic environment, indicated by a decrease in laser reflectance,

and is referred to as pyrolyzed organic carbon (OPC). In the oxidative atmosphere, OPC was emitted along with the original EC from the filter. The amount of OPC is defined as the carbon evolved until the laser reflectance returned to its baseline value (Han et al., 2007b). According to the IMPROVE protocol, EC is calculated as the total of the three EC subfractions minus OPC (i.e., EC is defined as the sum of EC1, EC2, and EC3, with OPC subtracted). The method enables differentiation between soot and char, as determined by the gradual oxidation of these black carbon subtypes in standard reference materials during the EC1 and the EC2 plus EC3 steps, where char is defined as EC1 minus OPC and soot as the sum of EC2 and EC3 (Han et al., 2007a; Han et al., 2016).

Please note that in this manuscript, we interchangeably use the terms "EC" and "BC". While these terms do not strictly refer to the same component, they serve as an adequate approximation within the scope of this study (Seinfeld et al., 1998; Bond et al., 2004). We use "EC" when discussing emissions and modeling components, reserving "BC" for climate-related discussions. Throughout this manuscript, the term "OC" refers to the total organic carbon (sum of the dissolved OC and the particulate OC).

2.5 The particle size analysis

The grain size of dust-fall samples was measured using a laser particle analyzer (Malvern Mastersizer 2000, UK). The particle size distribution was calculated for 100 grain size classes within a range of 0.02-2000 μm. Sample preparation for grain size analysis included wet oxidation of organic matter by adding 10 mL of 30% hydrogen peroxide (H_2O_2) per 1.5 g dry sample. Carbonates were dissolved by boiling with 10 mL (10% HCl) over 10 min. The glass beakers were filled with 150 mL distilled water and suspended particles were left to deposit. After siphoning the supernatant water, 10 mL of 0.05 N sodium hexametaphosphat [$(NaPO_3)_6$] were added, and the residue was dispersed for 5 min in an ultrasonic bath before measurement (Lu and An, 1997), the results are expressed as volume percentages.

2.6 Collection of Meteorological Factors

Meteorological data used in this study were obtained from the Qinghai Meteorological Bureau during the sampling period. Monthly meteorological parameters, including wind speed (WS), wind

direction (WD), temperature (Temp), relative humidity (RH), Precipitation (RF), sunshine duration (SUN), and the frequency of visibility ≤10 km (VIS), were collected from four national-level meteorological stations: XZH, GEM, DLX, and NMH. All of these stations are standard meteorological observatories established by the state, equipped with standardized monitoring capabilities, and provided continuous and reliable data for this study (https://data.cma.cn). In addition, two atmospheric dust deposition sampling sites (LTC and BLX) were established specifically for collecting dust samples. Due to field-monitoring constraints and research-budget limitations, no meteorological instruments were deployed at these two sites.

2.7 Statistical analysis

2.7.1 Estimation of Secondary Organic Carbon

Organic carbon consists of primary organic carbon (POC) and secondary organic carbon (SOC). Due to the intricate physical and chemical processes involved, SOC in urban atmospheres cannot be directly measured. Therefore, an indirect estimation method, known as the EC tracer method, has been developed (Turpin and Huntzicker, 1991). If the concentrations of OC and EC are available and primary OC from non-combustion sources ($OC_{non\text{-}comb}$) can be disregarded, EC can be utilized as a tracer for POC from combustion sources, facilitating the estimation of SOC (Turpin and Huntzicker, 1995):

$$POC=EC\times(OC/EC)_{pri,} \quad (2)$$

$$SOC=OC_{total}\text{-}POC, \quad (3)$$

where OC_{total} represents total organic carbon.

Traditional methods for determining $(OC/EC)_{pri}$ involve regressing OC and EC within a fixed percentile range of the lowest (OC/EC) ratio data (typically 5-20%) or relying on sampling days characterized by low photochemical activity and local emissions (Castro et al., 1999; Lim and Turpin, 2002). However, these approaches are limited by their empirical nature, lacking clear quantitative criteria for selecting the data subsets used to establish $(OC/EC)_{pri}$, defined as the hypothetical primary OC/EC ratio. In this study, we employed the minimum R squared (MRS) method (Millet et al., 2005; Wu and Yu, 2016; Wu et al., 2018a) to determine $(OC/EC)_{pri}$. This method calculates a set of hypothetical $(OC/EC)_{pri}$ and SOC values to identify the minimum

correlation coefficient (R^2) between SOC and EC, allowing for the accurate derivation of $(OC/EC)_{pri}$. The computational procedure followed the algorithm developed by Wu and Yu (2016) (available at: https://sites.google.com/site/wuchengust), implemented within the Igor Pro environment (WaveMetrics, Inc., Lake Oswego, OR, USA). Due to the limited dataset size and low temporal resolution, the MRS analysis was performed collectively across all sampling sites. In this approach, the R^2 between SOC and EC was calculated iteratively for a range of $(OC/EC)_{pri}$ values spanning 0 to 10. The minimum R^2 value of 1.33 (Figure S3) identified the optimal $(OC/EC)_{pri}$ representative of the true primary ratio.

2.7.2 Quantifying the contributions of playa salt (ps) and non-playa salt (nps) sources

To differentiate the contributions of salt lake sources to water-soluble ions in atmospheric deposition, we adopted a methodology similar to that used for marine aerosols. This approach relies on the ratio of water-soluble ions (SO_4^{2-}, Ca^{2+}, K^+, Mg^{2+}, Cl^-) to Na^+ in the salt lakes of the QDB, enabling us to assess the contribution of ps-Na^+ components to nps (Zhang, 1987); details in Zhu (2025).

$$nps\text{-}SO_4^{2-}=[SO_4^{2-}]-0.333\times ps\text{-}Na^+, \quad (4)$$

$$nps\text{-}Ca^{2+}=[Ca^{2+}]-0.062\times ps\text{-}Na^+, \quad (5)$$

$$nps\text{-}K^+=[K^+]-0.087\times ps\text{-}Na^+, \quad (6)$$

$$nps\text{-}Mg^{2+}=[Mg^{2+}]-0.051\times ps\text{-}Na^+, \quad (7)$$

$$nps\text{-}Cl^-=[Cl^-]-2.287\times ps\text{-}Na^+, \quad (8)$$

This was accomplished using equations that incorporate total Na^+, total Ca^{2+}, the average crustal ratio $(Na^+/Ca^{2+})_{crust}$ = 0.56 w/w (Bowen, 1979), and the average (Ca^{2+}/Na^+) ratio for Qaidam salt lakes, $(Ca^{2+}/Na^+)_{salt\ lake}$ = 0.06 w/w (Zhang, 1987). Among these, the mass concentration of $[SO_4^{2-}]$, $[Ca^{2+}]$, $[K^+]$, $[Mg^{2+}]$, $[Na^+]$ and $[Cl^-]$ were identified as constituents of dust-fall.

$$ps\text{-}Na^+=[Na^+]-nps\text{-}Na^+, \quad (9)$$

$$nps\text{-}Na^+=nps\text{-}Ca^{2+}\times(Na^+/Ca^{2+})_{crust}, \quad (10)$$

$$nps\text{-}Ca^{2+}=[Ca^{2+}]-ps\text{-}Ca^{2+}, \quad (11)$$

$$ps\text{-}Ca^{2+}=ps\text{-}Na^+\times(Ca^{2+}/Na^+)_{salt\ lake}, \quad (12)$$

2.7.3 HYSPLIT backward trajectory model

Backward trajectory clustering analysis was conducted on sampling points using the TrajStat plugin within Meteoinfo software. Daily backward trajectories for 48 hours were calculated from January 2020 to March 2023 and classified monthly based on differences in the horizontal movement direction and velocity of air masses. The trajectories were initiated at Universal Time (UTC) 00:00, with a 6-hour increment, originating from 500 m above sea level (Yang et al., 2014). Meteorological data utilized in this research were obtained from the Global Data Assimilation System (GDAS) provided by the U.S. National Centers for Environmental Prediction (NCEP), covering the period from December 2019 to February 2023 (Meteoinfo software website: http://meteothink.org).

2.7.4 PMF model

The PMF is a multivariate factor analysis tool that decomposes a matrix of speciated sample data into two matrices: factor contributions (G) and factor profiles (F). The goal of the PMF model is to solve the chemical mass balance between measured species concentrations and the respective source profiles, with the purpose of minimizing the objective function Q (Eq. 13) based upon the uncertainties (uij) of measured species (Paatero and Tapper, 1994).

$$e_{ij}=x_{ij}-\sum_{h=1}^{p} g_{ih}f_{hj};\ Q=\sum_{i=1}^{m}\sum_{j=1}^{n}(e_{ij}/h_{ij}s_{ij})^2, \quad (13)$$

where x_{ij} is the measured concentration of the j_{th} species in the i_{th} sample at receptor sites. f_{kj} is the source profile of the j_{th} species in the k_{th} factor and g_{ik} is the mass contribution of the k_{th} factor in the i_{th} sample. e_{ij} is the difference between modeled concentrations and measured concentrations.

The uncertainty for individual species (u_{ij}) was defined as the sum of two components: the x_{ij} multiplied by an error fraction, and one-third of the method detection limit (MDL). For data below the MDL, concentrations were replaced by 1/2 MDL and the corresponding uncertainty was set to 5/6 MDL (Reff et al., 2007). An extended description of the PMF parameters used in this study and error estimate based on the model's Q value, displacement (DISP), and bootstrapping (BS) tests (DISP-BS) are provided in the Supplementary Information. The error assessment and uncertainty data for the PMF source apportionment can be found in Tables S2 and S3, respectively.

2.7.5 Statistical Analysis

A one-way analysis of variance (One-Way ANOVA) was performed to examine the differences in dust deposition flux, soluble ions, carbonaceous components, and trace elements among various monitoring sites and periods. All statistical analyses were conducted in SPSS (IBM SPSS Statistics 26.0.0), with the significance level (P) set at 0.05. Prior to ANOVA, the normality and homogeneity of variance were tested for each dataset. The Shapiro-Wilk test was used to assess normality. If the data met the normality assumption ($P > 0.05$), parametric ANOVA was applied directly. If not, the data were log-transformed and retested; if the transformed data still violated this assumption, the non-parametric Kruskal-Wallis H test was used instead. In this study, all datasets were confirmed to follow a normal distribution. Homogeneity of variance was verified using Levene's test; meeting this assumption ($P < 0.05$) is a prerequisite for conducting ANOVA. When the overall ANOVA result was significant ($P < 0.05$), indicating that at least two group means differed statistically, a post-hoc test was performed. Given the potentially unequal sample sizes among groups, Tukey's Honestly Significant Difference test was used for pairwise comparisons to identify specific inter-group differences. Data in this study are presented as mean ± 1 standard deviation (Mean ± 1 SD).

Furthermore, to investigate the influence of meteorological conditions on the chemical composition of atmospheric dust, Pearson correlation analysis was applied to quantitatively assess the linear relationships between meteorological factors (e.g., wind speed and direction, relative humidity) and chemical components in dust (e.g., OC, EC, water-soluble ions). The underlying assumptions of the Pearson correlation (continuous variables, bivariate normal distribution, and linearity) were verified. All variables were tested for normality using the Shapiro-Wilk test and screened for outliers. For data that did not meet the normality assumption, log transformation was applied. The significance of correlation coefficients was tested using the t-test, with $P < 0.05$ considered statistically significant.

3. Results and discussion

3.1 Atmospheric dust deposition flux and water soluble ions concentration

The DF in the southern QDB is 5.41 ± 5.33 g/m^2·30d, slightly lower than that of the Lake Aibi Basin (10.77 g/m^2·30d) (Li et al., 2022), but higher than the surrounding areas of Akatama Salt Lake (2.93 g/m^2·30d) (Wang et al., 2014). Specifically, DF was 4.67 ± 4.96 g/m^2·30d in rural and 5.97 ±

5.73 g/m²·30d in urban areas. During the HP, DF in rural and urban areas were 4.62 ± 4.15 g/m²·30d and 4.95 ± 5.25 g/m²·30d, respectively. In contrast, NHP showed increased DF values of 4.77 ± 4.42 g/m²·30d (rural) and 7.66 ± 6.09 g/m²·30d (urban) (Figure 2). Notably, Urban DF during NHP demonstrated a 54.6% increase relative to HP, while rural DF rose by 7.1% (not statistically significant), contrasting previous findings that associated elevated DF with HP coal combustion (Cheng et al., 2009; Gao et al., 2013; Guo et al., 2010; Qi et al., 2018). We hypothesize that the increase in DF during the NHP is attributed to seasonal meteorological variations such as wind speed (Yang et al., 2024), and heightened anthropogenic emissions associated with tourism (Zhang et al., 2011). The NHP coincides with the peak tourist season (May-September) in the QDB, which receives about 17 million visitors annually, a number that continues to rise (Qinghai Statistical Yearbook, 2023). This elevated human activity likely enhances local emissions, contributing to increased dust deposition during this period. The influence of meteorological factors on dust deposition will be further examined in Section 3.5.

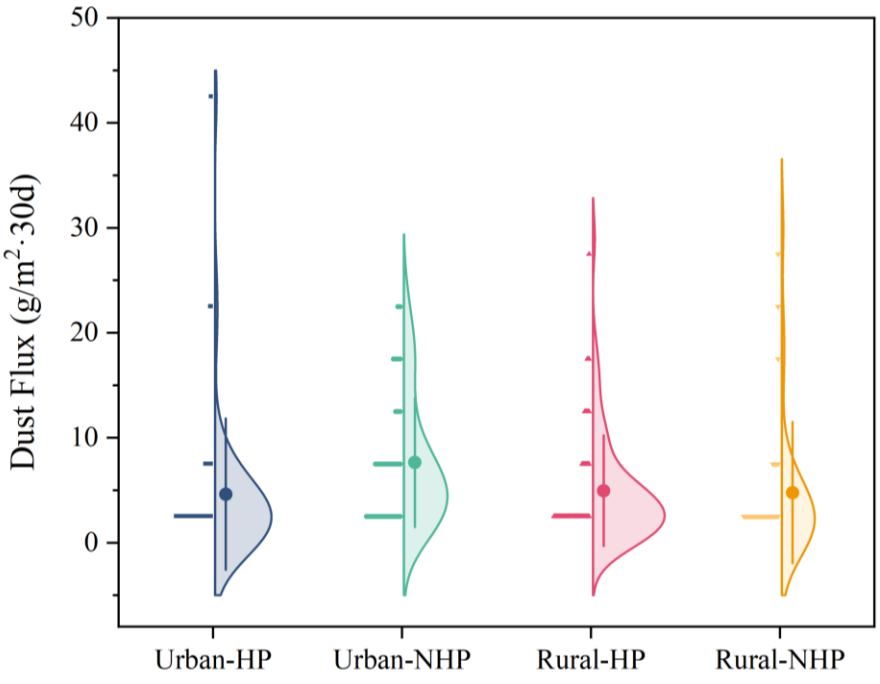

Figure 2 Dust flux distribution in urban and rural. The distribution of dust flux in four contexts: Urban with domestic heating period (Urban-HP), Urban with non-HP (Urban-NHP), Rural with domestic heating period (Rural-HP), and Rural with non-HP (Rural-NHP). Each violin plot illustrates the density distribution of dust flux values, with the central dot representing the mean, and the vertical lines indicating the interquartile range.

Water-soluble ion concentrations differed between rural (115.31 mg/g) and urban (72.81 mg/g) areas (not statistically significant). In rural, water-soluble ion content was greater during the HP than in the NHP, while the opposite trend was observed in urban areas (not statistically significant) (Figure 3 and S5). We categorized the water-soluble ions in dust deposits into ps and nps based on their sources, following the model of marine aerosols (Zhu et al., 2025). Playa salt content consistently surpassed nps in rural areas across both periods, while urban areas showed the opposite trend (not statistically significant). Notably, during the NHP, ps content in urban and rural increased by 54.46 and 36.86% respectively (not statistically significant). Backward trajectory analysis indicated that airflow in both rural and urban areas primarily originated from the northwest QDB and the eastern Tarim Basin during the HP, while during the NHP, it was influenced more broadly by the salt lake of the QDB (Figure S6 and S7), aligning with the observed variations in ps content. A similar increase in summer sea salt and non-sea salt ions has been reported in Rajkot, India, attributed to ocean wind direction (Gupta and Dhir, 2022).

The ratio of nps-SO_4^{2-}/NO_3^- in soluble ions is used to differentiate between coal combustion (fixed sources) and vehicular emissions (mobile sources) (Arimoto et al., 1996; Shen et al., 2008). The higher nps-SO_4^{2-}/NO_3^- ratio in urban areas compared to rural areas (Figure S8) indicates that during the study period, stationary sources (e.g., industrial emissions, coal, and biomass combustion) contributed more significantly to the ionic composition of urban dust-fall (Pipal et al., 2019). In contrast, the ratio was considerably lower during the NHP than during HP, suggesting that mobile sources likely played a more important role in the dust ionic composition during the NHP. Generally, emissions from coal combustion and biomass burning was intensified in northern China during the HP (Liu et al., 2016), while vehicle emissions dominate during the NHP (Xu et al., 2012), supporting the reliability of this analysis. Nevertheless, further investigation integrating atmospheric emission inventories, source tracers, aerosol physicochemical processes, and meteorological conditions is warranted.

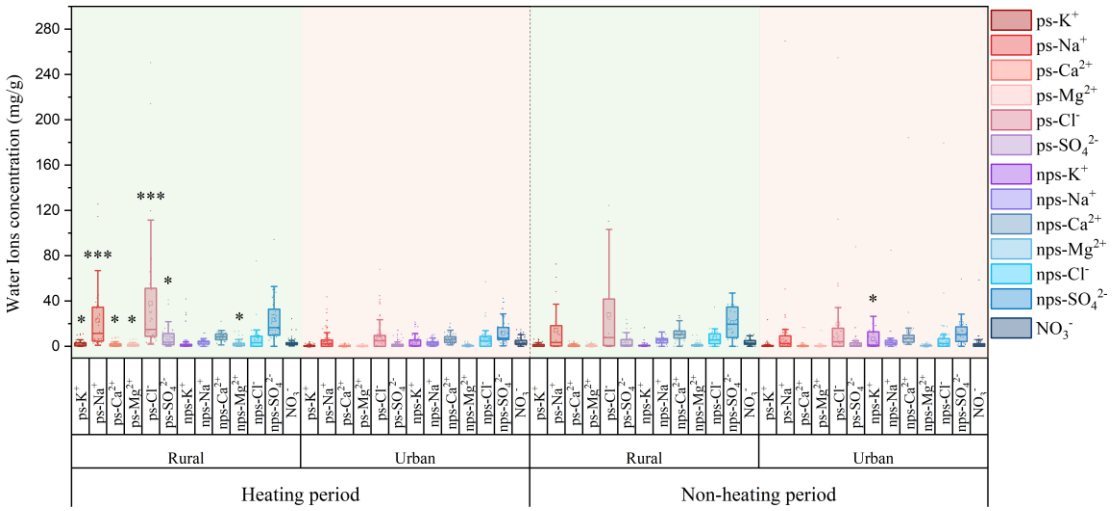

Figure 3 Concentrations of water ions in rural and urban across different seasonal periods (domestic heating and non-domestic heating periods). Asterisks indicate statistically significant differences between sites, with *, P <0.05; **, P <0.01; ***, P < 0.001.

3.2 Carbonaceous element compositions

3.2.1 Organic carbon and elemental carbon

The average total carbon (TC) concentration in the southern QDB was 4.87 ± 4.18 mg/g, significantly lower than that of Huangshi, Hubei province (25.15 ± 11.79 mg/g) and Washington (157 ± 3.2 mg/g), Kumasi in West Africa (28 mg/g) and Xi'an (14.6 ± 5.8 mg/g) (Han et al., 2007a; Han et al., 2009b; Zhan et al., 2016; Bandowe et al., 2019), suggesting relatively low carbon emissions in the southern QDB. Average OC and EC levels in QDB (3.48 and 1.41 mg/g, respectively) are markedly lower than those in Xi'an (7.4 and 7.2 mg/g, respectively), Wuhu (33.26 and 22.49mg/g, respectively), and Nanchang (25.15 and 11.46 mg/g, respectively) (Han et al., 2009a; Deng et al., 2014; Zhang, 2014), but significantly higher, particularly for EC, than in Nam Co (0.35 mg/g) (Chen et al., 2015).

In urban, TC content (4.75 ± 4.47 mg/g) was marginally lower than the rural level (5.02 ± 3.79 mg/g), although this difference was not statistically significant. Contrasting spatial patterns emerged for carbon components: EC dominated urban areas (1.46 ± 1.60 mg/g), while OC prevailed in rural (2.25 ± 1.92 mg/g) (not statistically significant) (Figure 4 and S9). This disparity may be attributed to the long-term combustion of biomass, coal, and wood in rural settings (Na et al., 2004). It may also be associated with meteorological conditions, particularly heightened solar radiation resulting

from reduced primary emissions in rural areas, which facilitates the formation of SOC (Xu et al., 2018a; Wang et al., 2019). Overall, the contents of OC and EC showed small difference between rural (3.73 and 1.33 mg/g, respectively) and urban aeras (3.28 and 1.47 mg/g, respectively).

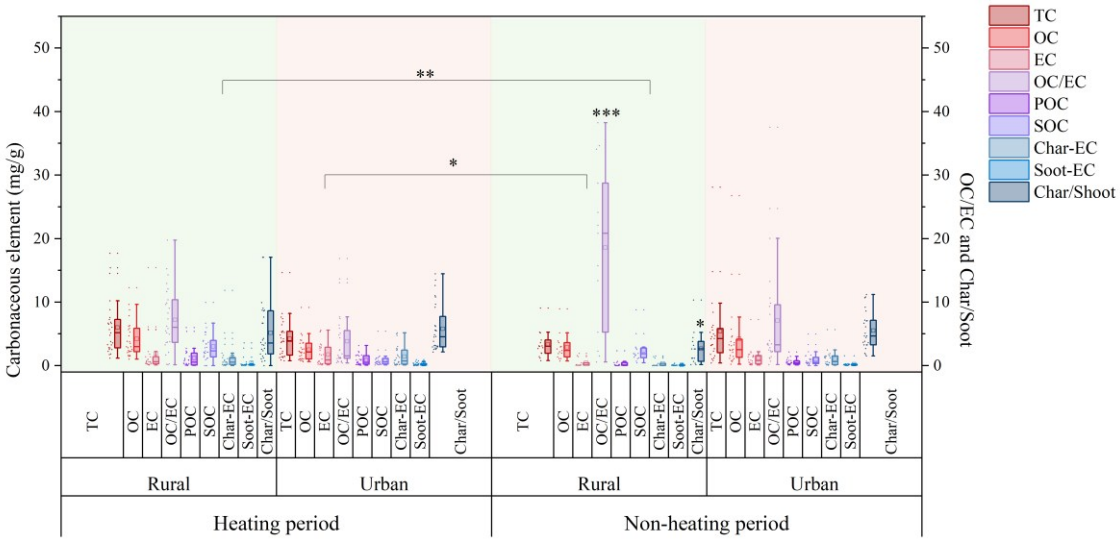

Figure 4 Concentrations of organic carbon (OC), elemental carbon (EC), secondary organic carbon (SOC), primary organic carbon (POC), char-EC, soot-EC and OC/EC, char/soot ratios in different sites (Rural, Urban) and seasonal variations (domestic heating and non-domestic heating period). Significant differences are indicated by asterisks (*p < 0.05; **p < 0.01; ***p < 0.001).

Seasonal analysis revealed elevated carbonaceous compound concentrations, specifically OC and EC, during the HP (not statistically significant). This increase is primarily due to local domestic heating activities coupled with adverse meteorological conditions, such as low temperature, weak winds (Oliveira et al., 2007; Gong et al., 2017), weak atmospheric turbulence, and frequent atmospheric inversions (Guo et al., 2016). Rural emissions primarily stem from coal and biomass burning for heating and cooking (Zhang et al., 2000; He et al., 2004), contributing to reduced OC and EC content in the NHP, whereas elevated EC levels in urban areas are primarily linked to vehicular and industrial sources. Spatiotemporal transport dynamics show EC depletion during rural ward pollutant migration due to atmospheric dispersion, particularly affecting coarse particulate fractions.

Notably, rural carbon emissions exceed urban levels in the southern basin, potentially

attributable to extended HP duration (7 months) compelling low-grade fuel (crop residues, wood, raw coal and yak dung) utilization (Na et al., 2004). In contrast, urban areas benefit from solar/wind energy infrastructure and government-led clean heating initiatives (suitable electricity for electricity policy), achieving 66.63% clean heating penetration (Statistical Yearbook of Haixi Xizang Autonomous Prefecture of Qinghai Province, 2023), leading to a comparatively lower TC content.

The OC/EC ratio is a valuable indicator of carbonaceous aerosol sources. In this study, Urban areas exhibited stable OC/EC ratios ranging from 0.15 to 16.89 (mean = 3.97), whereas rural areas showed significantly higher ratios during NHP (18.58 ± 12.80) compared to HP (7.20 ± 4.99) ($P < 0.001$) (Figure 4 and S11). Typically, the OC/EC ratio for vehicle emissions ranges from 0.7 to 2.4, for coal combustion emissions from 0.3 to 7.6, and for biomass burning from 3.8 to 14.5 (Schmidl et al., 2008; Gonçalves et al., 2010; Pio et al., 2011; Popovicheva et al., 2016).

These findings suggest that urban OC/EC ratios (0.15-16.89) are primarily associated with vehicle and coal combustion, while rural ratios (0.14-38.25) are predominantly linked to coal and biomass burning. The rural OC/EC ratios are significantly higher than those in urban areas ($P < 0.001$), which is consistent with the general distribution pattern of carbonaceous aerosols and reflects a greater contribution of SOC (Zhang et al., 2008; Sandrini et al., 2014). The higher OC/EC ratio typically indicates a greater contribution from biomass combustion; in this study, the OC/EC ratio of rural was 10.99 ± 10.00, which is significantly lower than values recorded in Nam Co in Xizang (16.3 ± 4.4) (Chen et al., 2015), yet higher than those observed in India (8.47) and Shanxi (0.7-1.6), Beijing (1.9-2.7), and Tianjin rural (2.66) (Zhang et al., 2007; Saud et al., 2013; Cheng et al., 2015; Wang et al., 2021). This finding indicates that carbonaceous aerosols in rural QDB derive from both fossil fuel combustion and biomass burning, with a pronounced influence from the burning of wood, yak dung, and similar biomass fuels, suggesting source specificity.

3.2.2 Char-EC and Soot-EC

Elemental carbon is classified into soot and char (Han et al., 2009b), with char-EC and soot-EC defined as EC1 minus OPC and the sum of EC2 and EC3, respectively (Han et al., 2007). Char-EC is typically produced from biomass burning at relatively low temperatures, whereas soot-EC originates from high-temperature coal combustion and automotive emissions (Zhu et al., 2010; Cao et al., 2013). The char-EC/soot-EC ratio, like the OC/EC ratio, serves as an indicator of carbon

aerosol sources. Since char and soot are mainly generated through combustion processes, their ratio is typically influenced by two key factors: the primary emission source and the deposition removal efficiency. For localized PM, such as in urban areas, the removal rate is generally negligible (Han et al., 2009a).

Char-EC constitutes 75.88% of rural EC (74.71% HP; 78.84% NHP) and 85.00% of urban EC (85.58% HP; 84.11% NHP) (Figure 4 and S10), demonstrating its dominance across spatial and temporal scales (not statistically significant). Research suggests that char-EC constitutes a larger proportion of coarse PM, while soot-EC is more predominant in fine particles, resulting in extended atmospheric residence times for soot-EC due to reduced deposition velocities (Han et al., 2009b). The increased levels of char-EC during the urban HP are linked to complex sources, including biomass fuel usage and transportation emissions, resulting in elevated char concentrations in urban areas and along busy roadways (Kim et al., 2011).

The char/soot ratios for automobile emissions, coal combustion, and biomass burning are 0.60, 1.9, and 11.6, respectively (Chow et al., 2004; Chuang et al., 2014). Generally, high-temperature combustion (e.g., vehicle and industrial processes) yields lower char and soot concentrations, while low-temperature combustion (e.g., household cooking and biomass burning) results in higher ratios (Han et al., 2016; Han et al., 2012; Han et al., 2010; Han et al., 2009a). Differences in char/soot ratios between urban and rural areas across seasons may be linked to wheat straw burning, contrasting with minimal vegetation combustion impacts in cities like Xi'an (Cao et al., 2005). The char/soot ratio for dust-fall observed in this study (4.97; Figure 4 and S11) is slightly higher than those recorded in Jinchang (3.84) (Han et al., 2009a) and Daheihe, Inner Mongolia (3.2) (Han et al., 2008). The relatively stable concentration of soot-EC in this study, along with the elevated char/soot ratio, suggests a correlation with higher coal consumption among local residents and industries. This indicates that, in comparison to other regions, carbon emissions in the remote QDB are predominantly sourced from coal and biomass burning, supporting OC/EC ratio findings. Furthermore, the char/soot ratio is elevated during HP (not statistically significant), highlighting the predominant influence of coal and biomass burning in rural areas during HP, while fossil fuel impacts are more pronounced in NHP.

3.2.3 Primary and secondary organic carbon

Aerosol samples with low OC/EC ratios typically exhibit low concentrations of POC, which mainly comprises primary carbonaceous compounds. The MRS method enables discrimination of OC into POC and SOC (Method 2.7.1) (Yoo et al., 2022; Liu et al., 2023). Secondary organic carbon constitutes a dominant fraction of OC in atmospheric aerosols. Research on carbonaceous aerosols in various Chinese cities indicates that SOC contributes 67% (53-83%) and 57% (48-62%) to rural and urban OC, respectively (Zhang et al., 2008), marginally lower than the 72.61% observed in this study (Figure 4 and S12). This is likely because SOC formation relies on solar radiation, the QDB experiences high levels of solar energy (Liu et al., 2017), which may facilitate photochemical oxidation of VOCs into SOC (Hama et al., 2022).

During the NHP, rural areas exhibit the highest SOC/OC ratio of 87.32% (Figure S13), while urban areas record the lowest ratio of 52.94% during the HP. This trend reflects elevated potential for photochemical activity and reduced contributions from POC, likely due to local emission sources, such as traffic and coal combustion (Mbengue et al., 2018). The high SOC/OC ratio suggests that SOC largely displaces OC. Our findings indicate that SOC levels are greater in rural areas (78.44%) compared to urban regions (67.78%) (not statistically significant), which consistent with the observation of higher OC/EC ratios in rural areas likely attributable to significant coal consumption for domestic heating, which enhances emissions of semi-volatile organic compounds and organic gases (Dan, 2004). As for seasonal variations, studies in California show an increase in SOC levels during warmer months, which is consistent with our results (Na et al., 2004). This contrasts with the broader observation that higher temperatures typically lead to lower SOC concentrations (Strader et al., 1999; Sheehan and Bowman, 2001). This discrepancy may stem from varying sources of SOC emissions throughout the seasons, necessitating further investigation in conjunction with other carbonaceous indicators.

This study conducted a comparative analysis of carbonaceous element concentrations in atmospheric dust-fall and road dust between the QDB and other global regions (Table S4). To ensure data comparability, the selected road dust samples consisted of directly collected in-situ dust without resuspension treatment. The results revealed that the concentrations of TC, OC, and EC in QDB (4.87, 3.48, and 1.41 mg/g, respectively) were significantly lower than those in industrial or urban areas such as Bolu, Turkey; New Delhi, India; and Ezhou, China, and were even lower than many other Chinese cities (Table S4). The low concentrations of OC and EC in QDB indicate minimal

anthropogenic pollution influence in this region, and the data can represent the regional background values of carbonaceous components in atmospheric dust-fall in the arid inland areas of East Asia (Chen et al., 2019a). This is crucial for global models assessing the emission fluxes of carbonaceous aerosols from dust source regions. In contrast, extremely high values of carbonaceous elements were found primarily in urban road dust from locations like Bolu, Turkey (TC: 605.2 mg/g), Gwangju, South Korea (TC: 31.97 mg/g), and Xi'an, China (TC: 36.53 mg/g), indicating strong influences from traffic emissions (mainly non-exhaust emissions) (Wei et al., 2015; Lee et al., 2018; Demir et al., 2022). For atmospheric dust-fall in major cities like New Delhi, India, and Wuhan, China, the carbonaceous components are affected by the combination of traffic emissions (diesel vehicle emissions being a major source of EC), industrial activities, and emissions from dense populations (Deng et al., 2014; Zhang, 2014; Zhan et al., 2016; Mishra and Kulshrestha, 2017).

The OC/EC ratio in QDB (7.09) is an intermediate level. It is much lower than that in regions dominated by biomass burning, such as Kumasi, West Africa (17.07) and Huainan, China (21.4), but is higher to ratios found in cities like Gwangju, South Korea (5.63) and Ulaanbaatar, Mongolia (5.69)(Lee et al., 2018; Bandowe et al., 2019; Liu et al., 2020). We primarily analyzed the OC/EC ratios in cities across different regions of China to reveal the influence of varying economic development levels.

Figure 5 presents spatial variations in urban OC/EC ratios across China. The findings reveal that the Northwest region, represented by QDB urban and Xi'an (XA) (Han et al., 2009b), exhibits a significantly lower ratio (2.50 ± 1.47) compared to central regions, including Nanchang (NC) (Zhang, 2014), Huangshi (HS) (Zhan et al., 2016), Wuhu (WH) (Deng et al. 2014), Xiaogan (XG) (Zhan et al., 2022), and Huainan (HN) (Liu et al., 2020), where the ratio is 5.86 ± 7.81. This ratio is also lower than that observed in eastern cities such as Beijing (BJ) (Tang et al., 2013), Tianjin (TJ) (Ma et al., 2019), and Shijiazhuang (SJZ) (Guo et al., 2018), which have a ratio of 6.83 ± 2.77. This pattern is consistent with the trends in atmospheric PM OC/EC ratios (Xie et al., 2023), suggesting that the carbon in the dust of the QDB urban primarily results from coal combustion and industrial emissions, leading to elevated EC concentrations and lower OC/EC ratios (Liu et al., 2022). Conversely, cities with higher economic development, such as Beijing and Tianjin, characterized by greater population density and income levels, typically experience secondary pollution, resulting in higher OC/EC ratios.

While the MDCO sampler effectively captures dry deposition flux, the reported OC and EC values must be interpreted with specific limitations in mind. First, passive sampling is governed by aerodynamic drag and gravitational settling; therefore, the collected OC is likely skewed toward the coarse fraction (e.g., re-suspended soil, biological debris) while potentially under-representing fine-mode anthropogenic combustion aerosols which have negligible settling velocities. This implies that the OC fluxes reported here should be interpreted as fluxes of deposited particulate OC, not as a direct surrogate for total atmospheric OC. Second, the extended exposure period inherent to passive sampling may lead to negative artifacts due to volatilization. Semi-volatile organic compounds may partition from the particulate to the gas phase, a process accelerated in environments with high solar radiation and ambient temperatures (Turpin et al., 1994). Consequently, the OC fluxes reported here likely represent a conservative lower limit. Finally, environmental factors such as wind speed and humidity affect collection efficiency. High wind speeds can induce turbulence leading to the resuspension of lighter organic particles from the collector, while high humidity may facilitate chemical or biological degradation of the organic fraction prior to laboratory analysis (Chow et al., 2011). Taken together, these processes imply that the OC collection efficiency of the MDCO sampler is not a fixed quantity but varies with temperature, relative humidity, precipitation, wind direction and other environmental factors. The role of these meteorological factors in atmospheric dust deposition is examined in detail in Section 3.5.

The char/soot ratio in QDB is notably high at 5.04, significantly exceeding that of other regions such as Xi'an (0.99) and Wuhan (0.09) (Wei et al., 2015; Liu et al., 2021). Char-EC primarily originates from incomplete combustion processes like biomass burning and coal combustion. Soot-EC mainly derives from high-temperature combustion sources such as fuel oil and diesel vehicle exhaust (Han et al., 2009). The exceptionally high char/soot ratio in QDB strongly indicates that its limited carbonaceous components predominantly originated from relatively inefficient combustion sources. These potentially included coal or small-scale biomass burning for local residential/expedition activities (e.g., heating, cooking) and possibly long-range transported biomass burning products (e.g., from forest/agricultural fires in South or Southeast Asia) (Han et al., 2009; Han et al., 2006; Han et al., 2016). In contrast, the very low char/soot ratios observed in cities like Wuhan and Xi'an clearly point to traffic source emissions as their primary contributor, a finding

likely influenced by the specific focus of those studies on road dust.

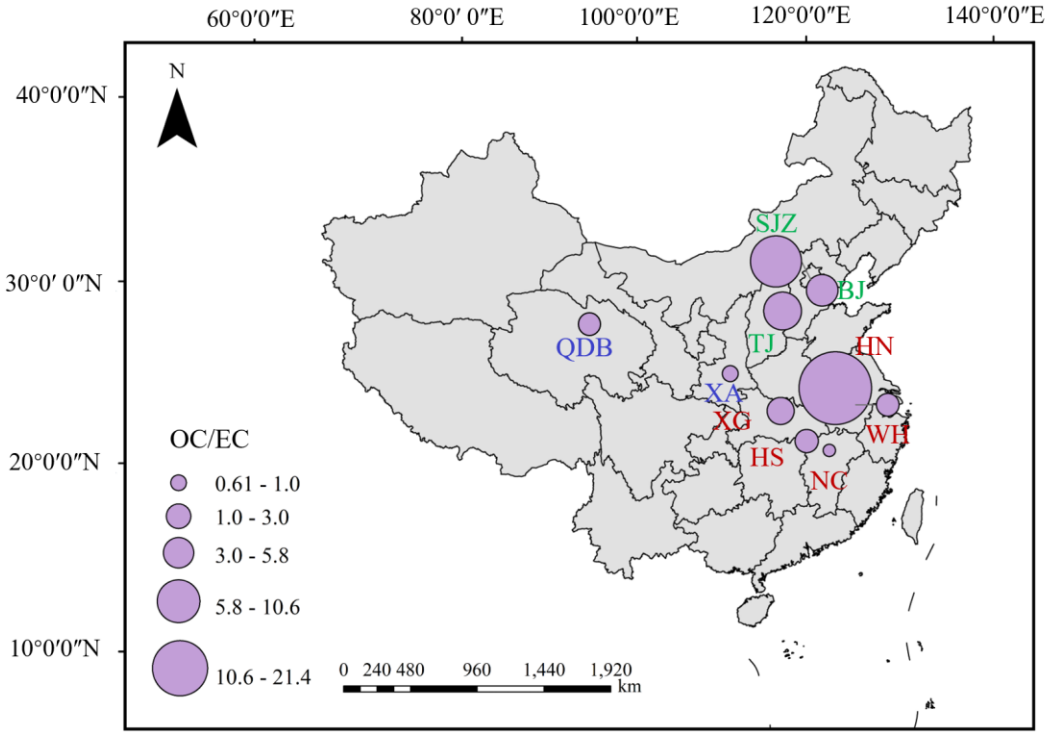

Figure 5 Distribution of OC/EC ratios across various regions of China. Blue designations represent the Northwest region, red indicates the Central region, and green denotes the Eastern region. The circle size reflects the magnitude of the OC/EC ratios.

However, we fully recognize the fundamental differences in sources and composition between road dust and atmospheric dust-fall. Road dust is primarily secondary dust formed from traffic activities, construction dust, soil particles, and resuspended deposited atmospheric particles, with its carbonaceous composition strongly reflecting intense local anthropogenic emissions (Casotti Rienda and Alves, 2021). In contrast, atmospheric dust-fall integrates contributions from local sources, regional transport, and even long-range transport. Therefore, direct comparison between these two may introduce bias when interpreting regional pollution characteristics and the degree of anthropogenic influence, which cannot be overlooked. Building on this analysis, the next phase of this research will focus on the sampling and analysis of fine atmospheric particulate matter ($PM_{2.5}$ and PM_1) to more accurately elucidate the emission levels and environmental and climatic impacts of carbonaceous aerosols in the QDB.

3.3 Trace elements concentration

The total concentration of major (Fe, Si, Al) and trace elements (Ti, Cr, Cd, Cu, Mn, Ni, Pb, V, Zn) was determined to be 8.74 ± 5.82 mg/g, while arsenic (As) remained below the detection limit in all analyzed samples. Crustally derived elements—Fe, Al, Si, and Ti—dominated the elemental profile, aligning with dust composition patterns reported in Ira, Singapore, and Beijing-Hebei regions, China (Joshi et al., 2009; Qiao et al., 2013; Eivazzadeh et al., 2021). In comparison to cities such as Beijing, Shanghai, Xi'an, and Lanzhou, and Junggar Basin the levels of heavy metals in dust from the QDB are relatively low (Jiang et al., 2018) (Supplementary Table S5).

The low heavy metal content in dust deposition within the QDB can be attributed to the following factors. The region has sparse human activity, lacks heavy industrial zones and dense urban clusters, resulting in low total anthropogenic emissions of heavy metals. Furthermore, the surface soil in the QDB itself has a low background level of heavy metals, primarily derived from natural sources with relatively weak influence from human activities (Nuralykyzy et al., 2021; Chen et al., 2021). From the meteorological perspectives, the basin's high altitude, strong winds, and arid conditions with minimal precipitation favor the dispersion of atmospheric pollutants. This makes the formation of prolonged stagnant weather conditions unlikely, thereby preventing the accumulation of pollutants and the occurrence of high concentrations near the ground. A particularly unique aspect of the QDB is its role as a significant source of salt dust. The recent study indicates that salt dust emitted from the playa lakes within the basin contributes substantially to atmospheric dust deposition (Zhu et al., 2025). These salt dust particles, composed mainly of soluble salts like NaCl and gypsum, may dilute the relative concentration of non-salt components, such as heavy metals, when released into the atmosphere in large quantities. The combined effect of these factors leads to the observed low heavy metal content in dust deposition in this region.

Throughout both the HP and NHP, trace elements concentrations in urban areas were consistently higher than in rural areas (not statistically significant), with the exception of Ti (Figure 6 and S14). During the HP, urban levels of Zn, Pb, and Cu were significantly elevated compared to rural areas ($P < 0.05$), and Pb also demonstrated a significant increase in urban during the NHP ($P < 0.01$). In rural, the differences in metal concentrations between HP and NHP were minimal. In

contrast, urban areas exhibited higher concentrations of all elements except for Ti, Cd, and Cr during the HP, with Fe and V showing notably elevated levels compared to other regions (not statistically significant). These variations in average concentrations indicate that coal combustion for domestic heating in urban areas contributes to increased atmospheric heavy metal levels (Duan and Tan, 2013; Meng et al., 2017). In contrast, Cd and Cr exhibited mixed anthropogenic sources with limited coal combustion contributions, while Ti concentrations remained stable across seasons, reflecting minimal anthropogenic influence.

Analysis of the carbonaceous components in QDB dust deposition reveals more intensive coal combustion in rural areas, yet the heavy metal concentrations in atmospheric deposition are lower than in urban area (not statistically significant). This observation can be explained by the following factors. First, pollution sources in rural areas are relatively singular, whereas urban areas are influenced by more complex heavy metal sources. During the heating period, heavy metals in the rural atmosphere of the QDB mainly originate from coal and biomass combustion. In contrast, urban areas are affected by a wider range of sources, including industrial activities, traffic emissions, and others (Liu et al., 2021; Huang et al., 2021a). Additionally, the dense building layout in urban areas hinders pollutant dispersion, leading to accumulation, while the open terrain in rural areas facilitates dilution and diffusion. This pattern, where rural heavy metal concentrations (particularly Pb, Cr, Cd, As, and other elements associated with coal combustion) are lower than those in urban areas during the heating season, has also been observed in studies conducted in Northeast China, Shanghai, Taiyuan, the Yangtze River Delta, and Southern Nigeria (Shi et al., 2012; Liu et al., 2021; Huang et al., 2021a; Liu et al., 2023b; Hilary et al., 2025).

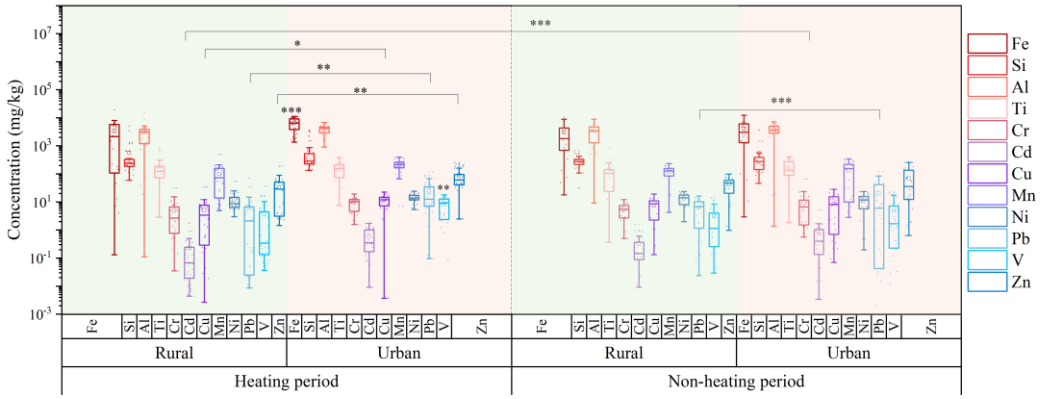

Figure 6 Concentration of heavy metals by rural and urban settings during domestic heating and

non-domestic heating periods. Significant differences are indicated by asterisks (*p < 0.05; **p < 0.01; ***p < 0.001).

3.4 Source apportionment

We conducted a PMF source apportionment analysis on soluble ions, trace and carbonaceous elements present in dust, specifically focusing on ps-SO_4^{2-}, ps-Ca^{2+}, ps-K^+, ps-Mg^{2+}, ps-Cl^-, ps-Na^+, nps-SO_4^{2-}, nps-Ca^{2+}, nps-K^+, nps-Mg^{2+}, nps-Cl^-, nps-Na^+, Fe, Si, Al, Ti, Cr, Cd, Cu, Mn, Ni, Pb, V, Zn, SOC, POC and char-EC, soot-EC. Seven source factors were identified based on prior research and an understanding of potential local sources: salt lakes, soil, traffic emissions, secondary inorganics, biomass and coal burning, and industrial activities (Figure 7 and S15. A plot of the time series is provided in Figure S16. The generation of Figure 7a involved extracting factors identified as the same source from the PMF factor profiles of each site (Urban and Rural) and each heating season (HP and NHP) shown in Figure S15. The arithmetic mean of the contributions from characteristic species (elements and ions) corresponding to each factor was calculated. Species with average contributions exceeding 20% were defined as characteristic species of that source in atmospheric dust over the QDB.

The factor profiles for each element in these source categories represent the arithmetic mean of profiles from six stations, with detailed operational methods provided in Supplementary Text S1. The uncertainty of the source contributions was calculated directly from the standard error of the multiple regression coefficients between the deposition flux (independent variable) and the source contribution (dependent variable) at different monitoring sites (Belis et al., 2015; Manousakas et al., 2017). The regression method assumes that all factors explaining the mass have been identified; however, if a significant portion of the mass not directly related to the species in the PMF analysis is omitted, the source contributions may be overestimated, which could be an important additional source of uncertainty. The results are shown in Table S6. It must be noted that this method captures only a portion of the total uncertainty, as it does not include errors from profile uncertainty or rotational ambiguity. The low errors calculated by this method indicate a good model fit. In this study, the sample numbers used for PMF source apportionment were 690, 780, 480, and 750 for the Urban-NHP, Urban-HP, Rural-NHP, and Rural-HP groups, respectively. Limited sample number can

increase the risk of model overfitting, reduce the representativeness of source profiles, and potentially lead to the merging of multiple sources into a mixed factor (Norris et al., 2014). Future research will extend the observation period and increase the sample number to further enhance the reliability of the source apportionment results.

The ions ss-Na^+, ss-Cl^-, ss-SO_4^{2-}, ss-Ca^{2+}, ss-K^+, and ss-Mg^{2+} are widely acknowledged as indicators of sea salt (Ambade et al., 2022; Aswini et al., 2022; Gluscic et al., 2023). In this study, we identified ps-Cl^- (83.96%), ps-Ca^{2+} (83.71%), and ps-SO_4^{2-} (83.71%), ps-Na^+ (83.70%), ps-Mg^{2+} (83.66%), ps-K^+ (82.35%) as key markers of salt lake sources. The contribution of salt lake sources in rural (12.29%) was higher than in urban (6.87%). During the HP, the proportion of salt lake sources in rural areas was 4.97%, compared to 19.61% in the NHP; urban showed contributions of 2.16% during the HP and 11.58% in the NHP, showing opposite seasonal trends. Backward trajectory simulations indicated that during the HP, air mass in both urban and rural areas mainly originated from the northwestern part of the basin and the eastern Tarim Basin, whereas during the NHP, they were broadly influenced by the salt lake regions within the basin (Figure S6). The minor wind direction differences and the inter-distributed sampling points (Figure 1), suggests no substantial geographical disparity between urban and rural sites. Additionally, the ion content derived from playa salts in dust deposits increased during the NHP in both areas. Therefore, we propose that the anomalous increase in salt lake contribution during the urban HP may be closely related to human activities. The enhanced Urban Heat Island (UHI) effect and temperature inversion structures during the HP can alter boundary layer height, turbulence, and deposition conditions, thereby increasing the residence time of externally transported particles within the urban boundary layer and elevating their measured contribution (Cichowicz and Bochenek, 2024). Urban heat sources and heating emissions may also modify local transport pathways, leading to more concentrated deposition of dust originating from playa regions over urban areas. Furthermore, dry road surfaces, increased traffic, and construction activities during the HP can promote the resuspension of previously deposited playa dust. The use and subsequent resuspension of road de-icing salts (e.g., NaCl, $CaCl_2$) may further amplify the contribution of tracer ions indicative of playa salts (Gertler et al., 2006; Casotti Rienda and Alves, 2021).

The second factor pertains to soil dust, characterized by trace element such as Fe (47.03%) and Si (36.22%), along with ions such as nps-Na^+ (54.21%), nps-Ca^{2+} (44.05%), nps-Mg^{2+} (32.30%),

(Pervez et al., 2018; Tian et al., 2021). Additionally, the proportions of elements such as SOC (32.58%) and POC (20.84%) suggest that the dust is likely mixed with fossil fuel emissions. Furthermore, V (24.62%) was detected at multiple sampling sites, indicating contributions from both soil dust and traffic emissions. Mg, Ca^{2+}, Al, Si, and Fe are typical tracers for soil dust (Liu et al., 2003; Heo et al., 2009). Notably, the contribution of soil dust in rural areas (21.14%) exceeded that in urban areas (12.97%), indicating that soil dust is a major source of atmospheric deposition. In urban areas, the contribution during the NHP was relatively low (6.84%), likely due to higher wind speeds and the effectiveness of frequent summer precipitation (Zhang et al., 2015a).

The third factor is traffic emissions, which are particularly pronounced in urban areas. Key characteristic elements and ions include nps-Cl^-(56.71%), Zn (50.13%), Cd (49.41%), Pb (47.84%), Ni(44.22%), Cu(37.61%), nps-Mg^{2+} (30.54%), V (29.63%), nps-K^+ (27.74%), NO_3^- (24.35%) and char-EC (46.71%), soot-EC (38.12%), SOC (32.25%), POC (23.72%) (Adeniran et al., 2017). In rural areas, traffic emissions contributed 8.52% to atmospheric deposition during the NHP, whereas in urban areas, the contribution was significantly higher at 46.83%, with 32.91% during the HP and 60.75% in the NHP. These findings correlate with previous studies on OC/EC and char/soot ratios, suggesting that carbonaceous elements in the NHP primarily derive from traffic emissions. The traffic emission factor in the QDB represents a mixture of vehicle exhaust and non-exhaust sources (e.g., tire and brake wear, and resuspended road dust). Elements and ions including V, NO_3^-, Ni, and carbonaceous components primarily associated with vehicle exhaust (Cong et al., 2011; Zhang et al., 2012). For instance, Ni can be emitted from fuel combustion and vehicle exhaust (Pacyna and Pacyna, 2001). In contrast, elements such as Cu, Zn, nps-Mg^{2+}, and nps-K^+ originate from non-exhaust vehicle emissions, including brake and tire wear, as well as the resuspension of road dust (Amato et al., 2014). For example, Zn may derive from the wear of rubber tires on roads (Rogge et al., 1993), Pb emissions may be related to wear (tires/brakes) (Smichowski et al., 2007), and Cu is associated with brake wear (Lin et al., 2015). Furthermore, the presence of crustal elements and ions such as Fe, Si, and nps-Mg^{2+} in the traffic emission factor for Urban-NHP, Urban-HP, and Rural-NHP suggests an additional contribution from resuspended road dust (Chen et al., 2019).

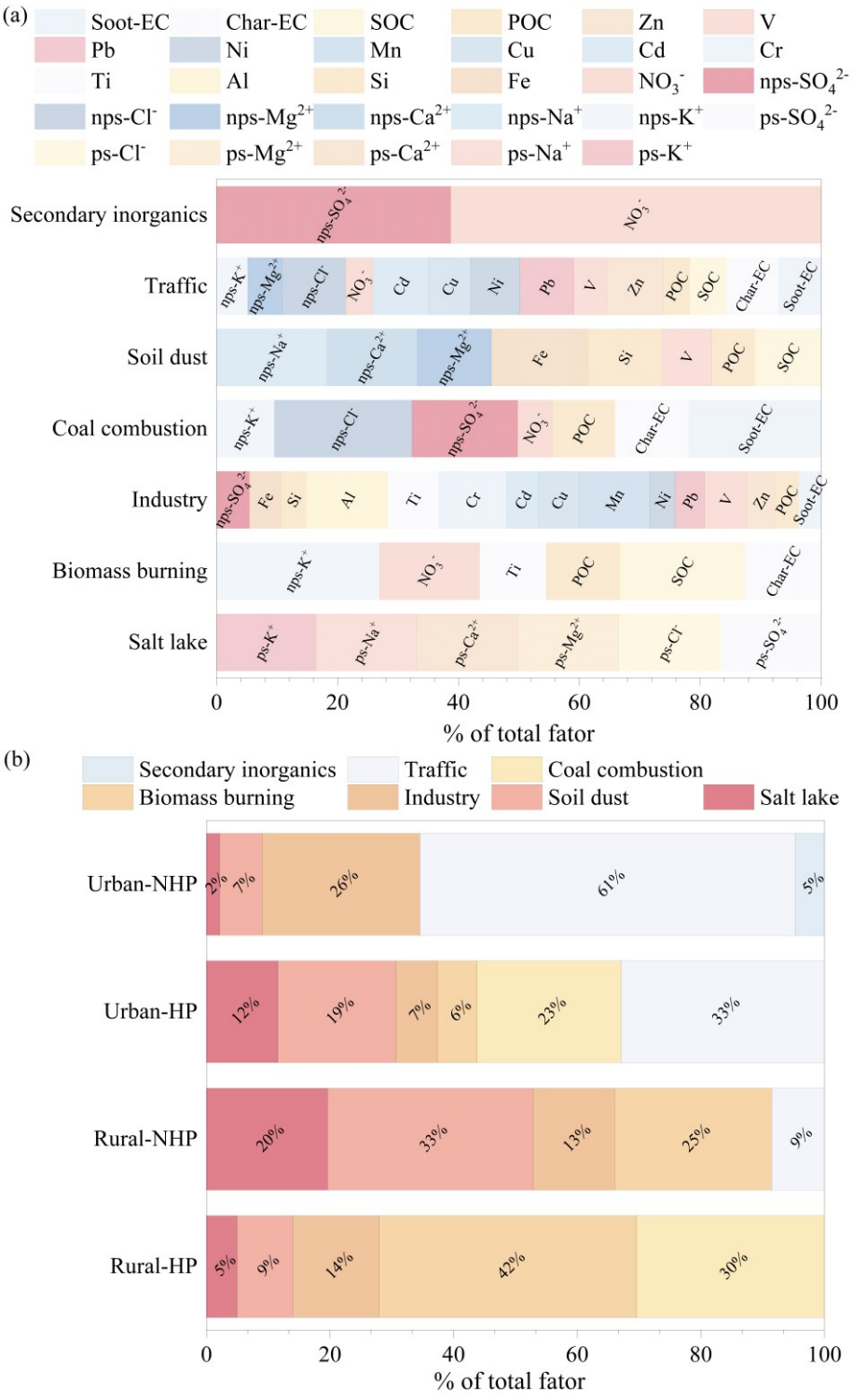

Figure 7 Factor profile and contributions in urban and rural area. (a) presents the factor profiles, represented as the arithmetic mean of individual elements across various locations, highlighting only those elements that constitute more than 20% of each profile. (b) illustrates the contributions of different sources at each location. [HP, domestic heating period; NHP, non-domestic heating period].

The fourth factor is coal combustion, characterized by high concentrations of nps-Cl^- (75.40%), nps-SO_4^{2-} (57.87%), nps-K^+ (31.75%), NO_3^- (19.10%), and soot-EC (72.54%), char-EC (40.63%), POC (34.10%) (Kundu and Stone, 2014; Contini et al., 2016). Zhang et al (2023) found that coal combustion emits particles rich in Cl^-. Coal combustion was more intense at site LTC in rural areas and at site DLX in urban areas. Coal combustion occurs exclusively during the HP, contributing 30.38% in rural areas and 23.31% in urban areas. These results align with earlier studies on carbonaceous aerosols, indicating that the carbon content from coal combustion emissions is higher in rural than in urban aeras. Consistent with northern China, air pollution in QDB urban has declined due to the adoption of clean heating technologies (Zhang et al., 2021; Xue et al., 2023). However, rural coal combustion remains a major source of carbonaceous aerosols during the HP.

The fifth factor, biomass burning, is characterized by significant concentrations of non-precipitating species, including nps-K^+ (56.99%), NO_3^-(34.97%), Ti (23.26%) and SOC(43.68%), char-EC (26.78%), POC (25.60%) (Simoneit, 2002; Sulong et al., 2019). K^+ serves as an important tracer for biomass burning (Cachier and Ducret, 1991). Biomass burning contributes 33.55% to rural atmospheric dust deposition, with a higher proportion during the HP (41.67%) compared to NHP (25.43%). In urban, biomass burning is primarily observed during the HP, contributing only 6.41%. These findings underscore biomass burning as a major source of carbon emissions in rural settings, aligning with the prevalent use of biomass fuels for cooking and domestic heating in Northern China's rural areas (Meng et al., 2019), where 70% to 80% of energy demand are fulfilled by materials such as dung cakes, firewood, charcoal, and crop residues (Tao et al., 2018; Shi et al., 2019). Furthermore, increased biomass burning is also associated with the autumn harvest period (Chen et al., 2017; Li et al., 2021).

The sixth factor pertains to industrial emissions, which are characterized by high concentrations of Mn (61.98%), Cr (59.22%), Ti (44.90%), V (37.17%), Cu (34.95%), Cd (28.63%), Fe (27.47%) Pb (25.75%) and Zn (24.03%) (Almeida et al., 2015; Yao et al., 2016). These elements were consistently detected across all sampling sites, alongside Al (71.17%), nps-SO_4^{2-} (28.84%), Si (22.24%) and POC (20.95%), soot-EC (19.97%), which indicate potential contributions from soil dust. Zn, Cu, Fe, and Mn are major chemical components in industrial emission profiles; Cd is a trace element found in metallurgical industries (Xu et al., 2022), while Zn and Mn emitte from oil

combustion, metallurgy, and steel manufacturing processes (You et al., 2017). Pb and Cd are associated with metal smelting and processing (Fang et al., 2021). Industrial emissions showed greater contributions during the HP at XZH and at DLX. In rural areas, industrial emissions constitute 13.57% of carbon output, with contributions of 13.95% during the HP and 23.20% during the NHP. In urban areas, industrial emissions account for 16.12% overall, with 6.69% during the HP and 25.55% in the NHP. Due to the abundance of non-ferrous metal resources (e.g., lead-zinc ores), oil, natural gas, and saline minerals (e.g., potassium, lithium) in the QDB, the primary industrial activities are mining and associated chemical industries. Particularly around the GEM and DLX sites, the presence of numerous Pb-Zn, Fe, and Cu mining enterprises leads to significant contributions from industrial emissions to urban dust fallout, making it one of the major sources of air pollution in the basin.

The seventh factor is secondary inorganic aerosols, primarily composed of NO_3^- (72.46%), nps-SO_4^{2-} (45.79%), (Liu et al., 2015; Liu et al., 2016a). High mass loadings of NO_3^- and SO_4^{2-} are characteristic of typical secondary inorganic aerosols (Huang et al., 2021). Research indicates that NO_3^- and SO_4^{2-} primarily result from the conversion of gaseous precursors, such as SO_2 and NO_x, through photochemical reactions, predominantly sourced from local and regional emission (Liu et al., 2015; Tao et al., 2013). Secondary inorganic aerosols are predominantly observed in urban areas during the NHP, where they contribute 4.70% to total aerosol sources. This increase is likely due to elevated temperatures and enhanced solar radiation during this period, which promote photochemical activity (Pandolfi et al., 2010).

Dust deposition sources exhibit significant seasonal and spatial variations. In the QDB, coal combustion (26.85%) and biomass burning (24.04%) dominate during HP, transitioning to traffic emissions (34.64%), soil dust (20.04%) and industry emission (19.37%) as the primary contributor in NHP. Rural areas predominantly contribute to dust through biomass burning and coal combustion, as well as natural sources like windblown dust and salt lake emissions. This pattern aligns with increased coal usage for winter domestic heating and heightened biomass burning for cooking and heating in rural areas. In urban, dust deposition is briefly influenced by anthropogenic activities, including traffic and industrial emissions, with minimal contributions from domestic heating. Such differences can be attributed to varying economic development models, industrial and energy structures, and levels of human activity (Kataki et al., 2016).

Time series analysis (Figure S16) indicates that atmospheric dust deposition in rural areas primarily originates from natural processes and dispersed anthropogenic combustion due to sparse human activities. During the NHP, the contribution of salt-lake emissions increases in May and June. Strong winds in spring and summer in southern QDB transport saline soil particles, making them a key source of dust deposition (Zhu et al., 2025). Soil-derived dust shows variations highly synchronized with the salt-lake source, with elevated contributions also occurring in May and June, reflecting the natural erosion of exposed surfaces under high wind conditions (Zhang, 2010). Traffic contributions rise noticeably in July and August, likely linked to increased summer tourism and transportation. During the HP, contributions from biomass and coal combustion increase, peaking from November to January. This peak is attributed to the widespread use of biomass fuels such as livestock dung for decentralized heating in rural areas, leading to increased particulate matter emissions (Chen et al., 2023). In contrast, contributions from salt lakes and wind-blown dust decrease significantly, likely due to lower wind speeds and reduced natural source emissions in winter (Jiménez et al., 2018; Li et al., 2019a; Yang et al., 2024).

In urban areas, where human activities are more concentrated, pollution sources are dominated by centralized anthropogenic emissions (industry, traffic, and centralized heating), with temporal trends driven by both industrial activity and the heating cycle. During the NHP, industrial contributions increase, especially in towns such as GEM and DLX, where surrounding mining operations constitute a major anthropogenic source (Zhu et al., 2016). Traffic-related contributions follow a trend similar to that in rural NHP, also peaking in July and August, reflecting the common influence of increased vehicle flow during the tourism season. Meanwhile, precursors such as SO_2 and NO_x from industrial and traffic emissions undergo photochemical and gas-to-particle conversion under summer sunlight, enhancing the formation of secondary inorganic aerosols (e.g., SO_4^{2-}, NO_3^-) (Zhang et al., 2013; Ma et al., 2017). During the HP, trends in salt-lake emissions, soil dust, and coal/biomass combustion are generally similar to those in rural areas. However, unlike in rural regions, traffic contributions in urban areas continue to increase during the HP, highlighting the higher intensity and less seasonally-variable nature of human activities in towns (Peng et al., 2021).

This study observed that the contribution of biomass burning to atmospheric dust deposition in rural areas of the QDB during the HP was higher than that of soil dust. Given that the collected dust samples had particle sizes >10 μm, while biomass burning typically emits aerosols in the submicron

range, we propose several potential explanations. Firstly, during the HP, factors such as increased soil moisture and snow cover significantly suppress soil dust emission, resulting in a lower intensity than in other seasons (An et al., 2018; Yang et al., 2019). Simultaneously, biomass and coal burning for heating increases substantially, leading to intense, short-term emissions of fine particles. Although it was fine initially, these high-concentration ultrafine particles can undergo coagulation or coalescence, aggregating with each other or onto pre-existing coarse particles, thereby increasing their size (Butler and Mulholland, 2004; Kulmala et al., 2004; Li et al., 2020). Furthermore, fine particles from biomass burning (e.g., carbonaceous materials) may mix internally with coarse particles like soil dust or salt dust from the QDB, forming internally mixed particles (Li et al., 2003; Hand et al., 2010). During source apportionment, such coarse particles are more likely to be attributed to the biomass burning source. Additionally, the QDB is a significant source of salt dust (Zhu et al., 2025). Salt dust particles (e.g., halite, gypsum) provide excellent condensation nuclei for soluble substances emitted from biomass burning, greatly promoting hygroscopic growth (Li et al., 2003; Kumar, 2010; Wang, 2013). The basin's topography also favors stable inversion layers, inhibiting pollutant dispersion and allowing particles more time to grow, mix, and age in the atmosphere. Prevailing wintertime winds may also transport pollutants from surrounding regions into the basin.

Moreover, the dust in this study was collected using a passive sampler. Over 90% of the collected dust particles were smaller than 100 μm, with approximately 25% less than 10 μm (Figure S2), indicating the presence of fine particles (<10 μm), albeit in a relatively small proportion. The particle size distribution of atmospheric dust deposition is similar to that of TSP, with both primarily consisting of particles smaller than 100 μm. Using PMF source apportionment, this study identified a notably high contribution from biomass burning in rural areas, particularly during the heating period. Similarly, studies on atmospheric TSP in Iran (Ashrafi et al., 2018), the Qinghai-Xizang Plateau (Lulang) (Zhao et al., 2013), Northeast China (Jia et al., 2024), and Qingdao (Liu et al., 2022) have also reported significant contributions from biomass and coal combustion. This suggests that contributions from biomass and coal combustion can indeed be observed in particles larger than 10 μm. Finally, the PMF model may have uncertainties in resolving sources with similar chemical profiles. If the chemical compositions of local soil dust and biomass burning particles overlap after long-range transport and complex atmospheric reactions, the model might not fully separate them

(Cesari et al., 2016).

3.5 Influence of meteorological factors

Meteorological parameters critically influence the accumulation and dispersion of airborne pollutants. To examine their impact on the chemical composition of dust deposition in this study, monthly data for temperature, humidity, precipitation, wind speed and direction, sunshine duration, frequency of visibility ≤10 km were obtained from four monitoring stations (XZH, GEM, NMH, DLX) during the study period (Table 1 and Figure S17). Data from the LTC and BLX sites were excluded from this analysis due to a lack of on-site meteorological instrumentation.

Based on ANOVA results, wind speed and temperature were significantly higher during the NHP ($P < 0.05$), promoting dust emission and consequently increasing atmospheric dust flux (Jiménez et al., 2018; Li et al., 2019b; Yang et al., 2024). However, at DLX, wind speed showed no significant seasonal difference, while precipitation and relative humidity were significantly higher in the NHP ($P < 0.05$). Since precipitation is a known suppressor of dust deposition (Li et al., 2019b), meteorological factors were not the primary driver of the elevated dust flux at DLX during the NHP (Figure S18); other factors, such as increased tourism, likely played a more important role. Additionally, correlation analysis revealed that dust and salt dust flux were positively correlated with the frequency of visibility ≤10 km and wind speed ($P < 0.05$) and negatively correlated with relative humidity ($P < 0.05$), consistent with previous findings (Wei et al., 2023).

Pearson correlation analysis further elucidated the relationships between dust chemical components and meteorological parameters. Ions derived from salt lake (ps ions) and crustal elements (e.g., Al) showed significant correlations with wind direction and speed ($P < 0.05$, Figure S19), indicating the dominant control of wind on the emission and transport of both playa salt and soil dust. During the HP in both rural and urban areas, relative humidity was negatively correlated with ps ions and crustal elements ($P < 0.05$), suggesting that higher humidity promotes the removal of airborne particles (Deshmukh et al., 2011). Playa salt ions are hygroscopic, so higher relative humidity facilitates their deliquescence (Rörig-Dalgaard, 2021). The mean annual relative humidity in the QDB (about 34 %RH) is notably lower than Wuhan (75.4 %RH) (Zang et al., 2021), Handan (63.35 %RH) (Meng et al., 2016), Pearl River Delta region (67.05 %RH) (Yue et al., 2015), Jorhat,

India (80.29 %RH) (Rabha et al., 2021) and other regions, providing favorable conditions for the long-range transport of salt-lake-derived particles.

Table 1 Meteorological conditions at the monitoring stations during the sampling periods.

Site	Period	Temperature (°C)	Relative humidity (%)	Precipitation ($mm \cdot d^{-1}$)	Wind speed ($m \cdot s^{-1}$)	Wind direction	Sunshine duration ($h \cdot m^{-1}$)	Frequency of visibility ≤10 km
XZH	Whole	4.27 ± 10.58	33.56 ± 5.74	2.63 ± 3.64	3.29 ± 0.63	WNW	237.73 ± 26.86	36.52 ± 15.27
	HP	-3.16 ± 6.30	33.57 ± 6.36	1.47 ± 2.95	2.93 ± 0.49	WNW	240.58 ± 25.04	41.19 ± 16.61
	NHP	15.65 ± 2.86	33.54 ± 4.63	4.41 ± 3.97	3.83 ± 0.40	W	233.55 ± 28.82	29.36 ± 9.10
GEM	Whole	5.76 ± 9.80	31.22 ± 5.39	3.07 ± 4.03	1.94 ± 0.43	W	237.44 ± 30.10	65.89 ± 26.78
	HP	-0.91 ± 5.94	30.38 ± 5.58	1.78 ± 1.69	1.75 ± 0.35	W	233.03 ± 30.95	83.54 ± 14.18
	NHP	16.45 ± 2.83	32.57 ± 4.38	6.1 ± 4.77	2.24 ± 0.36	W	244.21 ± 27.40	37.65 ± 15.71
NMH	Whole	5.42 ± 10.01	33.03 ± 6.99	2.93 ± 4.05	1.53 ± 0.39	WNW	221.93 ± 23.79	78.58 ± 16.21
	HP	-1.57 ± 6.08	30.23 ± 6.09	0.54 ± 1.14	1.70 ± 0.37	W	218.58 ± 20.84	88.75 ± 8.04
	NHP	16.13 ± 2.77	37.32 ± 6.04	6.59 ± 4.17	1.29 ± 0.30	WNW	226.84 ± 26.79	62.99 ± 12.84
DLX	Whole	3.32 ± 9.07	38.24 ± 8.48	18.45 ± 23.90	2.40 ± 0.35	WNW	254.63 ± 56.83	33.69 ± 12.3
	HP	-3.08 ± 5.39	34.04 ± 6.26	7.12 ± 8.89	2.42 ± 0.33	WNW	260.09 ± 49.28	34.5 ± 11.45
	NHP	13.12 ± 2.78	44.68 ± 5.95	35.83 ± 29.34	2.38 ± 0.35	W	230.33 ± 32.44	32.45 ± 11.98

Note: HP = heating period; NHP = non-heating period.

For nps ions, nps-Cl^- (a tracer of coal combustion) during the HP was positively correlated with wind direction ($P < 0.05$, Figure S19), indicating downwind transport. In contrast, nps-K^+ (a tracer of biomass burning) was negatively correlated with wind speed and solar radiation ($P < 0.05$), consistent with the observed peak in combustion contributions during winter when solar radiation is the lowest. Increased wind speed may suppress biomass-burning intensity (Zhang, 2024), while higher solar radiation can reduce smoldering-phase nps-K^+ emissions and enhance photochemical removal of aerosols (Kuang et al., 2020; Huang and Gao, 2021).

For carbonaceous components (Figure S20), in the rural NHP, soot-EC, OC, and the char/soot ratio were positively correlated with precipitation and negatively with the frequency of visibility

≤10 km ($P < 0.05$). In the rural HP, the char/soot ratio was positively correlated with precipitation. In the urban NHP, char-EC, soot-EC, and their ratios showed positive correlations with humidity and precipitation ($P < 0.01$), while POC was positively correlated with the frequency of visibility ≤10 km ($P < 0.01$). Total carbon, OC, and SOC were negatively correlated with wind speed ($P < 0.05$). In the urban HP, EC and char-EC were negatively correlated with the frequency of visibility ≤10 km; POC was positively correlated with wind speed and solar radiation but negatively with the frequency of visibility ≤10 km ($P < 0.05$); and the char/soot ratio was positively correlated with wind speed and negatively with the frequency of visibility ≤10 km ($P < 0.01$).

Overall, wind speed was negatively correlated with TC, OC, SOC, POC, and the char/soot ratio ($P < 0.05$). This pattern aligns with findings from Xi'an (Cao et al., 2009), New Delhi (Tiwari et al., 2015), and Karachi (Bibi et al., 2017), although it was not observed in rural areas (Sharma et al., 2002). Increased wind speed enhances ventilation, promoting aerosol dispersion and reducing carbonaceous aerosol concentrations (Saha and Despiau, 2009). The low mean wind speed (about 2.30 m/s) may also indicate that carbon emissions are predominantly local rather than distant sources (Shen et al., 2021). Positive correlations of precipitation and relative humidity with soot-EC, char-EC, OC, and the char/soot ratio likely reflect the hygroscopic nature of carbonaceous aerosols: under humid conditions, particles absorb water, increase in mass, and undergo accelerated dry deposition. For instance, $PM_{2.5}$ deposition rates can increase 2-3 fold as relative humidity rises from 30% to 70%, thereby shortening their atmospheric residence time (Wu et al., 2016). The negative correlations of EC, char-EC, and POC with the frequency of visibility ≤10 km align with findings in cities such as Suzhou (Zhang et al., 2010) and Beijing (Kong et al., 2021), attributable to the strong light-absorbing (EC) and light-scattering (OC) properties of these components, which jointly enhance atmospheric extinction and reduce visibility (Zhang et al., 2010; Kong et al., 2021). Weaker correlations in rural areas indicate that carbonaceous components are less influenced by local meteorological dilution and removal processes, pointing to more complex source influences.

In summary, seasonal variations in atmospheric dust deposition result from the combined effects of meteorological conditions and anthropogenic emissions. In urban areas, meteorological factors, especially wind speed and relative humidity, play a dominant role in the dispersion and removal of local anthropogenic pollutants. In rural areas, dust composition is more strongly influenced by large-scale wind-driven dust transport and regional combustion activities.

Wind direction analysis provides a valuable supplement to the PMF model (Watson et al., 2008). Studies have shown that emissions from sources such as traffic, biomass burning, coal combustion, and salt lakes exhibit significant directional patterns, which can be effectively visualized using polar plots (Saraga et al., 2021). In this study, bivariate polar plots were generated for the concentrations of POC, SOC, char-EC, soot-EC, and selected ions/elements (Figure S21-S24), as well as for the source contributions resolved by the PMF model (Figure S25 and S26), to illustrate their directional provenance. Notably, the directional patterns of the various aerosol components exhibited distinct differences and did not fully align with the local prevailing wind directions.

The polar plots of key tracer ions and elements (Figure S21-S24) showed patterns consistent with the corresponding PMF-resolved source factors (Figure S25 and S26), supporting the validity of the source apportionment. Specifically, the following pairs showed similar directional distributions: Total-ps and the salt-lake source; nps-K^+, POC, and char-EC with biomass burning; nps-Ca^{2+} and Si with soil dust; nps-SO_4^{2-}, nps-Cl^-, and SOC with coal combustion; Fe and Cd with industry; Ni and Zn with vehicle combustion; Fe and Al with industry; and nps-SO_4^{2-} together with NO_3^- with secondary formation. These spatial consistencies reinforce that the selected chemical tracers are associated with their respective PMF-identified sources.

Wind speeds in rural areas are generally higher than those in urban environments, with predominant wind directions between west (W) and north (N), indicating the potential long-distance transport of pollutants. Specifically, emissions from saline lakes primarily originate from the west-northwest (WNW) direction. Back trajectory modeling suggests that HP levels are mainly attributed to salt lake emissions from the northwestern QDB, while in NHP, sources include long-distance transport from salt lakes in the Tarim Basin (Figure S6, S7). For industrial emissions during the HP, sources primarily originate from medium- to short-distance transport in the W direction, with NHP periods showing contributions from both the west (W) and northwest-northwest (NNW) directions. In contrast, soil dust in HP is predominantly sourced from long-distance transport from the W direction, whereas in NHP, it is largely of local origin. Biomass burning during HP mainly arises from local sources, while its contribution during NHP is minimal and comes from both local and long-distance transport, consistent with PMF results.

In urban areas, lower wind speeds and varying dominant wind directions between the HP and NHP lead to different sources of pollutants. During HP, emissions from saline lakes and coal

combustion primarily originate from short-distance transport in the WNW direction, while biomass burning is sourced from the W. In contrast, industrial emissions, soil dust, and vehicle combustion are predominantly attributed to local sources. In NHP, pollutants demonstrate a dispersion trend from their sources to urban areas. Industrial emissions primarily come from the WNW to north-northeast (NNE) directions, soil dust from the southwest (SW), and vehicle emissions from the W direction. Notably, both salt lake and secondary formation sources exhibit similar directional patterns, indicating a common origin from the SW direction. The presence of processing enterprises near salt lakes suggests that secondary aerosols may form from emitted SO_2 and NO_2 (Hewitt, 2001). These findings align with previous analyses showing that urban industrial and traffic emissions mainly derive from local sources, while coal and biomass burning predominantly originate from surrounding rural areas, corroborating the role of nearby soil sources in salt lake and soil emissions.

3.6 Environmental implication

The source apportionment analysis using the PMF model indicates that in the QDB, rural dust-fall predominantly originates from the combustion of solid fuels, including firewood, yak dung, and coal, accounting for approximately 72.05% of the total contribution. This proportion significantly exceeds contributions reported for rural areas in Beijing (41%) (Hua et al., 2018), Agra (54.3%) (Agarwal et al., 2020), and Beihai, Guangxi Province (66.7%) (Zhang et al., 2019).

The higher contribution in this study likely reflects the local energy profile, as the sampling site in Haixi Mongol and Xizang Autonomous Prefecture, Qinghai Province, primarily relies on coal, yak dung, and firewood, constituting 58%, 23.5%, and 13% of rural energy consumption, respectively (Jiang et al., 2020; Shen et al., 2021). In contrast, solid biomass fuels, including wood and yak dung, account for over 70% of rural household energy consumption in Xizang, with yak dung alone representing 53% (Liu et al., 2008; Xiao et al., 2015). Similar patterns emerge in South and Central Asia, where biomass fuels dominate residential heating (firewood: 39%; dung: 29%) (Amacher et al., 1999; Heltberg et al., 2000; Hoeck et al., 2007; Foysal et al., 2012; Behera et al., 2015; Kerimray et al., 2018). In northern China, rural domestic heating primarily relies on coal (46%), firewood (23.8%), and electricity (15.1%) (Tao et al., 2018), further highlighting the unique energy composition of QDB.

Recent studies have shown that South Asia, Central Asia, and Xizang contribute significantly to high concentrations of atmospheric PM, particularly BC, which accelerates glacier melting in the QXP (Ming et al., 2010; Xia et al., 2011; Chen et al., 2015). The QDB is recognized as a significant dust source affecting the glacier surfaces on the QXP, although it is often overlooked (Dong et al., 2014; Wei et al., 2017; Zheng et al., 2021). Compared to other dust sources, the QDB exhibits higher emissions from coal combustion, giving it a unique influence on the QXP. The organic matter and pollutants, such as PAHs, released from household solid fuel combustion, particularly coal (98%) and dung (94%), are substantially higher than those from firewood (Leavey et al., 2017; Secrest et al., 2017; Ye et al., 2020). Consequently, the impact of PM from coal combustion in the QDB on the QXP is significant. Specifically, the presence of BC in PM increases glacier albedo, accelerating the melting of glaciers and snow in the region (Kang et al., 2020) and impacting global freshwater resources (Huss and Hock, 2018). Additionally, BC enhances cloud condensation nuclei, ice number concentration, and cloud cover (Zhou et al., 2025), thereby influencing global climate change. Furthermore, coal combustion releases harmful emissions, including CO_2, NO_x, CO, SO_2 and sulfur trioxide (Munawer, 2018), adversely affecting local human health and exacerbating climate warming on the QXP (Liu et al., 2006; Li et al., 2023), with broader implications for global climate. Therefore, the atmospheric pollutants emission of the QDB deserves considerable attention. However, this study focuses primarily on larger particles, indicating a need for further research on the environmental impacts of carbonaceous aerosols in atmospheric PM within the QDB.

In addition to the distinctive energy consumption structure in the rural QDB, which leads to significant contributions from coal and biomass burning during HP, atmospheric dust deposition in the QDB during the NHP primarily originates from traffic and industrial emissions. The contribution from traffic emissions during the NHP was twice that during the HP. Considering the larger particle size of the dust samples collected in this study, the traffic-related dust is likely derived mainly from vehicle non-exhaust emissions, such as road dust (Gondwal and Mandal, 2021). This indicates that NHP atmospheric conditions are significantly influenced by resource development and tourism. Sampling sites, such as Qarhan Salt Lake, along with GEM and LTC stations within approximately 100 km (Figure 1), suggest that salt lake resource extraction has a lower impact on regional aerosols than traffic emissions, despite salt lakes being the primary resource. This is likely because salt lake development mainly involves solar evaporation and chemical processes like extraction and

adsorption, which emit fewer pollutants compared to other mining methods (Zhen, 2010). Consequently, salt lake resource exploitation exerts a relatively minor effect on local atmospheric carbonaceous aerosol.

Similar salt lakes with comparable environments to QDB, such as Salar de Uyuni in Bolivia, the Atacama Salt Lake in Chile, and Ombre Muerto in Argentina, are rich in lithium resources (Li et al., 2014), making them focal points for resource development. Additionally, Salar de Uyuni, the Atacama Salt Lake, Junggar Basin, and the Great Salt Lake are renowned tourist destinations. This suggests that, in arid basin salt lakes with similar climates and intensive human activity, atmospheric carbonaceous aerosols are likely influenced by resource exploitation and tourism, especially tourism. The study's findings can inform policy decisions regarding unexploited salt lakes in South America, such as Ombre Muerto and Salar de Uyuni. However, while QDB also hosts mineral resources such as copper, iron, and tin, this research focused on larger particles (>100 μm), which are more indicative of local sources. Given that sampling was conducted around the salt lakes, potential impacts from other mineral resource developments may have been underestimated. Further research is necessary to fully assess the environmental effects of carbonaceous aerosols in QDB atmospheric particles.

In conclusion, the localized source profiles developed in this study, such as OC/EC ratios for heating-period emissions and key heavy-metal tracers, provide critical input parameters for regional climate and air quality models. This enables more accurate simulation of the emission, transport, and deposition of carbonaceous aerosols in the QDB and supports quantitative assessment of their contribution to glacier retreat on the QXP. Based on that rural areas in arid and semi-arid basins such as QDB rely heavily on solid fuels for heating, with associated environmental impacts, we recommend that local governments implement targeted clean-energy transition policies. These should prioritize promoting clean heating alternatives (e.g., solar and electric heating) in rural regions, supported by financial subsidies and technical assistance. Simultaneously, the adoption of clean stoves and upgraded biomass fuels should be encouraged to reduce emissions from traditional biomass and coal combustion (Dickinson et al., 2015; Li et al., 2011; Shen et al., 2017). Given the close climatic and environmental linkage between QDB and QXP, we further propose integrating air pollution control in QDB into the broader environmental protection framework of QXP. Establishing a joint regional air pollution prevention and control mechanism for the "QDB–QXP"

system would help align emission reduction in the basin with glacier protection goals on the plateau. To strengthen the scientific foundation of such a mechanism, future studies should focus on clarifying the key transport pathways, transformation processes, and quantitative impacts of air pollutants from QDB to QXP glaciers.

4. Conclusion

This study analyzed the composition of dust deposition at six sampling sites in the southern QDB from January 2020 to March 2023 and examined DF, soluble ions, trace and carbonaceous element content in urban and rural samples during both domestic heating and non-domestic heating periods. Through integrated application of backward trajectory modeling, PMF, and carbon speciation indices, we identified dominant dust sources and evaluated domestic heating impacts on atmospheric processes in remote regions.

The findings revealed that DF and carbon emissions were significantly higher in rural than in urban areas. Among carbon indicators, urban areas exhibited elevated EC levels (1.46 ± 1.60 mg/g), while OC levels were higher in rural (3.73 ± 2.61 mg/g). Economic development increase OC/EC ratios, but they are driven by different intrinsic factors. Char-EC was the dominant contributor to EC (80.44%), with urban char-EC levels (85.00%) showing a notable increase compared to rural levels (75.88%). Secondary organic carbon was the principal contributor to OC (72.61%), with rural SOC levels surpassing (78.44%) those in urban areas (67.78%). The OC/EC and char-EC/soot-EC ratios, along with PMF results, indicated that during HP, dust deposition in the QDB was primarily derived from coal combustion (26.85%) and biomass burning (24.04%), while traffic emissions accounted for 34.64% of dust during NHP. Coal and biomass burning were the main contributors to rural dust, strongly influenced by domestic heating, whereas urban dust predominantly originated from traffic (46.83%) and industrial emissions (16.12%). Compared to other dust sources in the QXP, coal consumption in the QDB is higher during the domestic heating period. The resulting emissions of BC and greenhouse gases may exacerbate glacier melting in the region, warranting increased attention. Given the distinctive carbonaceous aerosol signatures identified in the QDB, we recommend prioritizing their radiative forcing effects in regional environmental policymaking and climate modeling frameworks. Furthermore, findings of this study offer a valuable scientific

basis for understanding atmospheric carbonaceous aerosols in arid basins and salt lake regions with climates similar to QDB. They can particularly inform policy decisions regarding unexploited salt lakes in South America, such as Ombre Muerto and Salar de Uyuni.

However, this study primarily focused on larger-scale PM and examined the effects of heating on carbonaceous aerosols in the QDB. It lacks an investigation of aerosols with smaller particle sizes (e.g., PM_{10}, $PM_{2.5}$, PM_1), which is essential for a comprehensive understanding of carbonaceous aerosol characteristics in this unique region. Furthermore, in addition to offline observations, future research should incorporate online observations with high spatiotemporal resolution and utilize numerical air quality models such as CMAQ, CAMx, WRF-CHEM, and NAQPMS to analyze the spatiotemporal distribution and future trends of carbon aerosols in the QDB.

Author contribution

HZ: Conceptualization, data curation, formal analysis, funding acquisition, investigation, methodology, project administration, validation, writing – original draft.

LZ: Data curation, formal analysis, methodology.

SZ: funding acquisition, validation, writing – review & edited.

XZ: Supervision, conceptualization, funding acquisition, writing – review & edited.

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Declaration of competing interest

The authors declare that there is no conflict of interest.

Data availability

Datasets for this research has been uploaded in Zenodo and is available at https://doi.org/10.5281/zenodo.14382853 (Zhu, 2024).

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