



Full length article

Fine particulate matter emissions from electric vehicle fast charging stations

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ABSTRACT

The global shift to electric vehicles necessitates the expansion of Direct Current Fast Charging (DCFC) stations, yet the related environmental and public health impacts remain unclear. Here, we report that the power cabinet at DCFC stations emit fine particulate matter (PM_{2.5}). We collected integrated filter samples from 50 DCFC stations across 47 cities in Los Angeles County, California. The daily PM_{2.5} concentrations were between 7.3 and 39.0 $\mu\text{g m}^{-3}$, significantly higher than urban background sites ($p = 0.02$) and the nearest U.S. EPA monitoring stations ($p < 0.0001$). To understand the emission mechanism of these particles, we measured real-time PM_{2.5} mass concentration, particle size distribution, and other pollutants, as well as characterized particle chemical compositions on integrated filter samples. Our results indicate that these particles, primarily in the sub-micrometer range (0.5–1.0 μm), are likely due to particle resuspension from the power cabinets. PM_{2.5} samples from power cabinets showed higher levels of brake and tire wear tracers (Ba, Cu, Zn) and dust tracers (Ca, Al, Fe) compared to samples from nearby chargers and background sites. With no current emission standards for DCFC, managing particle resuspension is crucial for improving air quality and protecting public health as transportation electrification advances.

1. Introduction

Mitigating greenhouse gas (GHG) emissions from the transportation sector is crucial for achieving climate goals in the United States and globally. In response, policymakers worldwide have committed to accelerating the transition from internal combustion engine vehicles to electric vehicles (EVs) (Greene et al., 2014; Lieven, 2015; Ou et al., 2018). Diverse regional policies supporting EV transition have been adopted in the United States (Greene et al., 2014). California, which is the largest market for new cars in the United States, has set a goal of putting 5 million EVs on the road and achieving 100 % EV sales for new passenger cars and light trucks by 2035 (Office of Governor Gavin Newsom, 2020). This unprecedented large-scale transportation electrification will necessitate the establishment of a vast network of EV chargers globally. Public EV chargers mitigate the constraints of home charging (Kaufmann et al., 2021), thereby alleviating drivers' concerns

about limited driving range (commonly known as “range anxiety”), and provide crucial support for the widespread adoption of EVs in multi-dwelling housing that may lack home charging capability. With the massive investments through the Inflation Reduction Act and the Infrastructure Investment and Jobs Act, the U.S. government has ambitious plans to allocate \$7.5 billion toward the advancement of EVs and charging infrastructure (The White House, 2023). This commitment includes the establishment of a widespread network of 500,000 EV charging stations by 2030 (The White House, 2023). Among different types of public EV chargers, Level 3 Direct Current Fast Charging (DCFC) is especially critical for mitigating range anxiety compared to low-power Level 1 and medium-power Level 2 chargers, particularly for unplanned trips or emergencies. DCFCs greatly reduce charging time, thereby eliminating one of the major barriers toward wider adoption of EVs (Brown et al., 2022; Saha et al., 2022).

The dramatic expansion of DCFCs accelerates the transition to clean

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transportation, which is expected to mitigate climate change and improve air quality. However, their environmental and public health impacts have been poorly understood. Previous studies have primarily focused on optimization of power sources and clean energy grid integration to reduce the carbon footprint of EV lifecycles (Jayachandran et al., 2021; Powell et al., 2022). Studies have also evaluated their impacts on reducing GHG emissions and improving urban air quality (Tang et al., 2021), and on residential property values (Liang et al., 2023). Conversely, environmental and public health concerns have been raised about the deployment of DCFCs. For example, studies have indicated that EV chargers produce electromagnetic radiation during the charging process (Bae and Park, 2022), but reported minimal health risks overall (Gryz et al., 2022). Additionally, the operation of DCFC stations may release heavy metals, such as nickel, lead, and lithium (National blueprint for lithium batteries, 2021). Recently, some claims have been made on social media about DCFC stations using diesel engines for power (Marquis, 2023), raising questions about energy cleanness and potential air pollutant emissions, but these claims lack empirical data support. To determine whether DCFCs emit air pollutants and to understand potential emission mechanisms, we collected fine particulate matter (PM_{2.5}) filter samples at DCFC stations across 47 different cities in Los Angeles County, conducted real-time sampling, and investigated the chemical composition of PM_{2.5}.

2. Methods

2.1. DCFC stations for sample collection

We sourced the data for DCFC locations from the Alternative Fuel Data Center, U.S. Department of Energy (U.S. Department of Energy, Alternative Fueling Station Locator), and cross-referenced with the Tesla Supercharger network in Los Angeles County (Supercharger, 2023) to obtain the most up-to-date information on station locations as of November 14, 2023. This cross-referencing was necessary because the Alternative Fuel Data Center did not provide real-time data for Tesla locations. For the projected 2028 data, we obtained projections from a policy memo estimating the additional DCFCs under the Infrastructure Investment and Jobs Act (Laska and Hughes-Cromwick, 2022).

The methodology for selecting DCFC stations for sample collections involved a systematic approach to ensure a broad representation of charging locations within Los Angeles County. We focused on outdoor DCFC locations within the geographical boundary of Los Angeles County. Among these 313 outdoor stations, we further narrowed down our selection to those stations that possessed an output exceeding 100 kW for each fast charger and were equipped with at least one stand-alone power cabinet. A total of 113 DCFC locations were identified in Los Angeles County. To ensure a diverse and representative sample, we used a random selection process to include a wide range of cities within the county to avoid clustering in certain areas. As a result of this process, we selected a total of 50 DCFC stations spreading across 47 cities for sampling. The charging output powers ranged from 100 to 350 kW, with 250 kW being the dominant category, covering 28 out of the 50 DCFC stations sampled. The number of EV fast chargers at each site ranged from 3 to 44, and the number of power cabinets varied from 2 to 11. Currently, the Tesla Supercharger network dominates the DCFC market in the United States, with a 61 % share of all operational EV fast chargers, followed by Electrify America at 10 % and EVgo at 7 %. Thus, we sampled 30 Tesla Supercharger stations, which accounted for 60 % of all 50 locations. Out of the 50 DCFC stations, we selected five for an additional two-week integrated PM_{2.5} sampling to analyze chemical composition.

We chose eight traditional gas stations for comparative analysis, each situated near one of the DCFC stations, and strategically distributed across various locations in Los Angeles County for representativeness. Additionally, we selected four urban background sampling areas. These were primarily located in park areas and were chosen from the north,

west, south, and east regions of Los Angeles County. The purpose of these locations was to serve as reference points for air pollutant concentration comparisons.

2.2. Field measurements

2.2.1. Integrated 24-h PM_{2.5} sample collection and analysis

Between February and May 2024, we collected 24-h integrated PM_{2.5} samples at the power cabinet (with a 3 cm distance between the sampler inlet and the power cabinet cover) across 50 DCFC stations and 12 comparison sites (i.e., eight gas stations and four urban background sites) in Los Angeles County. To achieve this, we used 37 mm polytetrafluoroethylene (PTFE) filters (Cytiva R2PJ037, 2.0 µm pore size) coupled with Ultrasonic Personal Aerosol Samplers (UPAS v2.1 PLUS, Access Sensor Technologies, Fort Collins, CO, USA). The UPAS monitors collected PM_{2.5} filter samples using a cyclone inlet with a 2.5 µm cutoff size and provided real-time PM_{2.5} concentration data through the on-board sensor (Sensirion SPS30). The performance of UPAS monitors has been validated in both laboratory and field settings, showing comparable results to gold standards such as the Harvard Impactor and the Mesa Labs/BGI Triplex Cyclone (Pillarissetti et al., 2019; Ripley et al., 2022; Volckens et al., 2017). Air sampling consistently started at midnight (00:00 AM) with the flow rate set at 1 L/min using the “AST UPAS” mobile application (Version 1.0.4, Build 1.0.1). Following air sampling, we retrieved the sampling log from the micro-SD card within the UPAS monitor, which contained information on flow rate, sampled air volume, and meteorological parameters. The mass of the collected PM_{2.5} was measured gravimetrically using a microbalance (Model MX5, Mettler Toledo Inc., Columbus, OH, USA) with 1-µg resolution. The average 24-h PM_{2.5} mass concentration was determined by dividing the mass of sampled PM_{2.5} by the sampled air volume.

Outdoor 24-h PM_{2.5} data from the nearest air monitoring stations during the sampling period were downloaded from the U.S. EPA (U.S. Environmental Protection Agency, Outdoor Air Quality Data). The monitoring site IDs were 06-037-1103, 06-037-1201, 06-037-1302, 06-037-1602, 06-037-2005, 06-037-4008, 06-037-9035, 06-059-0007, 06-071-0027, 06-111-0009, and 06-111-2002.

2.2.2. Real-time measurements

Real-time measurements were conducted at the power cabinet of the DCFC station, with all instruments powered by a portable battery power station (Explorer 3000 Pro, Jackery Inc., Fremont, CA, USA). We used conductive silicone tubing to connect the sampler inlets directly to the power cabinet, maintaining a 3 cm distance between them. PM_{2.5} mass concentrations were measured with a DustTrak DRX aerosol monitor (Model 8534, TSI, Inc., St. Paul, MN, USA). The DustTrak covers a concentration range of 0.001–150 mg/m³, and the data logging interval for the DustTrak was set at 1 s. The DustTrak is sensitive to the composition and size distribution of airborne particles, leading to a notable upward bias in PM_{2.5} during air sampling (Zhao et al., 2017). Based on published studies, a correction factor of 2.0 was applied for DustTrak measurements (Javed and Guo, 2021). CO and CO₂ concentrations were monitored by a Q-Trak Air Quality monitor (Model 7575-X, TSI Inc., St. Paul, MN, USA) at 1-second intervals. A dual-beam ozone monitor (OM; Model 205, 2B Technologies Inc., CO, USA) was used to measure ozone concentration and ambient temperature at 10-second intervals. The size distributions of particles in the size range of 0.54 to 20 µm were measured using an Aerodynamic Particle Sizer (APS, Model 3321, TSI, Inc., St. Paul, MN, USA) at 1-minute intervals. All real-time instruments were synchronized to a master clock to ensure accuracy and consistency in data collection. We also recorded the number of EVs being charged at the DCFC station at 5-minute intervals.

2.2.3. Characterization of particle volatility using holographic microscopy

We employed a holographic particle monitoring device termed c-Air to quantify the volatility of each particle (Luo et al., 2021). The device

was placed at the power cabinet and comparison sites to sample the air with a miniaturized vacuum pump (M00198, GTEK Automation, Inc., Lake Forest, CA, USA) at a rate of 2.9 L/min, and the airborne particles were captured by impacting them on a sticky transparent coverslip (Zefon, cassette air-o-cell). A vertical-cavity surface-emitting laser (VCSEL) diode (OPV300, TT Electronics, plc., Plano, TX, USA) was used to illuminate the coverslip from above and created inline holograms of the collected particles. These holograms, capturing particles in real-time, were recorded as time-lapsed images at 2 Hz (0.5 s per frame) by a complementary metal–oxide–semiconductor (CMOS) image sensor chip (IMX219PQ, Sony, Corp., Japan) for 2 min.

The captured holograms were processed with a shading correction algorithm and digitally back-propagated by 750 μm to roughly reach the top of the coverslip surface using the angular spectrum method (Ozcan and McLeod, 2016). The twin image noise and defocus artifacts in the back-propagated holograms were removed by a trained convolutional neural network (CNN) model (Luo et al., 2021), which was trained to transform randomly defocused, back-propagated hologram images into phase-recovered, in-focus images. By utilizing the acquired holographic image sequence of each detected particle, we derived the phase distribution of particles. Subsequently, we calculated the contact angle of each particle at each time frame, labeling those with decaying contact angle and reasonable lifetime (2–30 s) as volatile particles according to our developed methods (Luo et al., 2021).

2.3. $\text{PM}_{2.5}$ element analysis

Two-week integrated $\text{PM}_{2.5}$ samples were collected on 37 mm PTFE filters using UPAS monitors at five DCFC stations across Los Angeles County for element analysis. After weighing the sampled $\text{PM}_{2.5}$ mass, the PTFE filters were removed from the support rings, and the particulate matter was dissolved using a mixed-acid microwave-assisted bomb digestion. Digestions were conducted in a Milestone Ethos Easy microwave with an SK15 pressurized rotor and microvials, using a solution of 1.5 mL 16 M Nitric Acid, 0.5 mL 12 M Hydrochloric Acid, and 0.1 mL 28 M Hydrofluoric Acid. The microwave program involved a 25-minute ramp to 205 $^{\circ}\text{C}$, followed by a 25-minute hold at 205 $^{\circ}\text{C}$. Digests were diluted to 15 mL with ASTM Type 1 reagent water and further diluted (1:2) for Sector Field Inductively Coupled Plasma Mass Spectrometry (SF-ICP-MS) analysis (Thermo-Finnigan Element XR). The limits of detection (LOD) for $\text{PM}_{2.5}$ elements on sampling filters are listed in Table S1.

2.4. Quality assurance and quality control

The UPAS components (i.e., pump, $\text{PM}_{2.5}$ inlet system, and filter cartridge) were cleaned before sampling, and flow rates were checked both before and after deployment. After sampling, the PTFE filters were immediately stored at -20°C until analysis. During the study period, six blank filters were also collected for quality assurance and quality control purposes. The mean $\pm$ SD of the field blanks in this study was 0.5 ± 0.4 μg . According to the U.S. EPA quality assurance handbook for air pollution measurement systems, the acceptance criteria for laboratory filter blanks and field blanks are $< \pm 15.1$ μg and $< \pm 30.1$ μg , respectively (U.S. Environmental Protection Agency, 2017; Buonanno et al., 2011). Before weighing, all PTFE filters were balanced at $20 \pm 1^{\circ}\text{C}$ and 35 ± 1 % relative humidity in a controlled environmental chamber for at least 24 h. Each filter was weighed three times before and after sampling, and the mass increase of the filter was obtained from average weight. The collected $\text{PM}_{2.5}$ mass was further adjusted by subtracting the mass increase of the blank filters. Due to the sensitivity of PTFE filters to static electricity, the PTFE filters were discharged on a polonium-210 strip for at least 20 s before each weighing.

2.5. Statistical analyses

Statistical summaries, including mean, standard deviation (SD), median, interquartile range (IQR), coefficient of variation, and range were calculated for air pollutant levels across various sampling locations. Time series plots, box plots, contour plots, and pie charts were created to visualize the distribution of air pollutant levels. To identify significant emission peaks from real-time monitoring data, we computed the rolling averages for each pollutant and set 10 % of these averages as the threshold for breakpoint determination (Hurtado et al., 2020; Topál et al., 2016). The selection of 10 % as the threshold aims to balance sensitivity to actual environmental changes with the risk of false positives caused by minor fluctuations. Statistically, variations exceeding 10 % of normal fluctuation ranges are more likely to indicate genuine environmental changes rather than random noise. Specifically, we first computed the rolling averages for the raw data of each pollutant using a window size of five time points. Subsequently, we calculated 10 % of each pollutant's rolling average to establish the threshold. When the difference between the rolling averages of two consecutive time points surpassed this threshold, the latter time point was marked as a breakpoint. As the $\text{PM}_{2.5}$ data are not normally distributed, the Mann-Whitney U test was performed to compare $\text{PM}_{2.5}$ levels between DCFC stations and other sampling locations. All statistical analyses were conducted using R software (version 4.1, Vienna, Austria). Statistical significance was set at a two-tailed alpha level of 0.05 ($p < 0.05$).

3. Results

3.1. Distribution of EV chargers

Fig. 1a illustrates the distribution of operational EV fast chargers across the United States as of November 2023, totaling 36,000 units (US Department of Energy, 2023). While some stations were located along interstate highways, the majority were clustered in major metropolitan areas, especially along the West and East Coasts, corresponding with major population centers. As shown in Fig. 1b, California led with 9,900 EV fast chargers, followed by Texas with 2,200 and Florida with 2,100, respectively. The DCFC network is expected to undergo a rapid expansion from 2023 to 2028, as facilitated by the Infrastructure Investment and Jobs Act. This is illustrated by the darker orange bars for each state in Fig. 1b.

Among all counties in the United States, Los Angeles County had the highest number of EV fast chargers in operation, totaling 1,938 units. On average, Los Angeles County had about six individual chargers per DCFC station. Out of a total of 313 operational DCFC stations, we identified 113 that meet our sampling criteria, which included specific requirements for charging output power and the presence of power cabinets. We then randomly selected 50 DCFC stations across 47 cities in Los Angeles County to perform field measurements (Fig. 1c). For comparison purposes, we also sampled eight gas stations and four urban background sites across Los Angeles County during the same sampling period.

3.2. $\text{PM}_{2.5}$ concentrations at DCFC stations

Fig. 2 shows a typical DCFC station in Los Angeles County, alongside $\text{PM}_{2.5}$ sample collections at various locations. This DCFC station was located at a business plaza, surrounded by residential neighborhoods (Fig. 2a). It was equipped with four power cabinets and 16 fast charging stalls connected by electric wires. The prevailing wind direction was from the south. We used a real-time instrument to monitor $\text{PM}_{2.5}$ and found high concentrations near the power cabinets (see supplementary materials, video). To measure $\text{PM}_{2.5}$ concentration at the DCFC station, we set up an UPAS on top of one of the power cabinets, approximately 2 m above the ground (Fig. 2b). For comparison, we also set up a UPAS monitor at a nearby upwind EV charger, 5 m away (Fig. 2c), and another UPAS at a downwind background site, 25 m away at the same height.

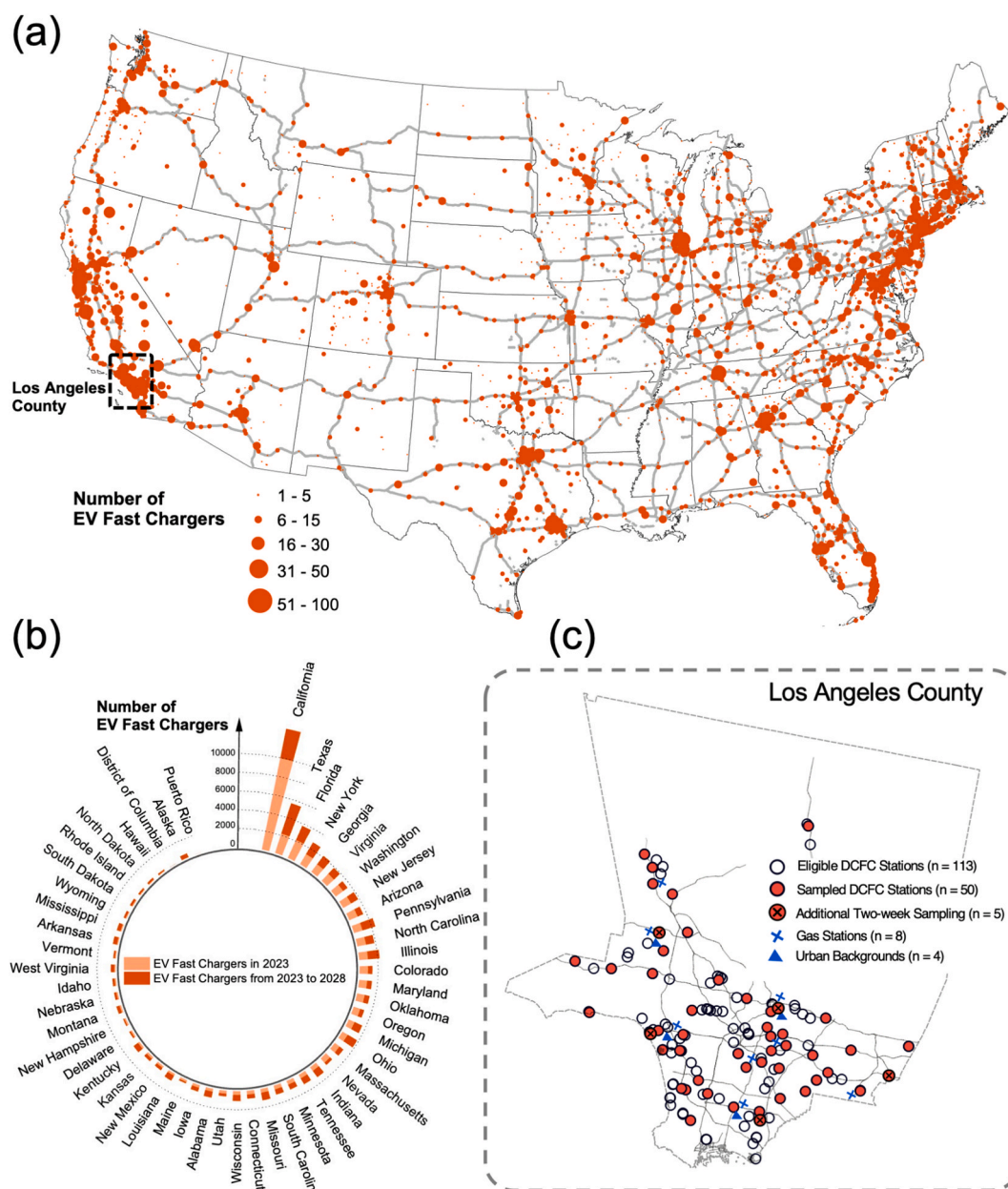


Fig. 1. Distribution of electric vehicle (EV) fast chargers across the United States and in Los Angeles County. (a) A map showing the distribution of EV fast chargers across the United States as of November 2023. (b) A radial histogram depicting the number of EV fast chargers in each state in 2023, along with projections from 2023 to 2028. (c) Field measurement locations selected for this study within Los Angeles County.

The coefficients of variation of $PM_{2.5}$ concentrations during 24-h measurements at 30-second resolution at the power cabinet, the nearby upwind EV charger, and the downwind background site were 137 %, 34 %, and 24 %, respectively (Fig. 2d–2f). We observed frequent $PM_{2.5}$ peaks at the power cabinet, especially between 7:00 am and 9:00 pm, with the highest concentration exceeding $300 \mu g m^{-3}$ (Fig. 2e). The nearby upwind EV charger also showed some $PM_{2.5}$ peaks, which were typically below $30 \mu g m^{-3}$ (Fig. 2f). In contrast, the downwind background site had more stable $PM_{2.5}$ concentrations (Fig. 2d). This indicated that the power cabinet at the DCFC station could be a potential $PM_{2.5}$ emission source.

To investigate whether high $PM_{2.5}$ concentrations at power cabinets are a common issue, we collected integrated 24-h filter samples for $PM_{2.5}$ at the power cabinet of each of the 50 randomly selected stations using UPAS monitors. Detailed information about the number and output power of EV fast chargers at each station can be found in

Table S2. Fig. 3a presents the 24-h mean $PM_{2.5}$ concentrations at each of the 50 sampled DCFC stations, ranging from $7.3 \mu g m^{-3}$ to $39.0 \mu g m^{-3}$, with 46 % of the sites exceeding the 24-h World Health Organization (WHO) Global Air Quality Guidelines (AQGs), set at $15 \mu g m^{-3}$. As shown in Fig. 3b, the mean $\pm$ SD and median (IQR) $PM_{2.5}$ concentrations were $15.2 \pm 6.3 \mu g m^{-3}$ and 14.1 (6.7) $\mu g m^{-3}$ for DCFC sites, $11.9 \pm 4.7 \mu g m^{-3}$ and 11.8 (7.3) $\mu g m^{-3}$ for gas stations, and $8.6 \pm 3.8 \mu g m^{-3}$ and 9.3 (7.1) $\mu g m^{-3}$ for urban background sites. The Mann-Whitney U test shows significantly higher $PM_{2.5}$ concentrations at DCFC stations compared to those at the urban background sites ($p = 0.02$). In comparison, we also identified the nearest U.S. Environmental Protection Agency (EPA) outdoor air monitoring station for each DCFC station. The mean $\pm$ SD and median (IQR) $PM_{2.5}$ concentrations at the U.S. EPA outdoor air monitoring stations during the same sampling period were $9.6 \pm 4.1 \mu g m^{-3}$ and 9.9 (6.8) $\mu g m^{-3}$, respectively, which were significantly lower than those at the DCFC stations ($p < 0.0001$).

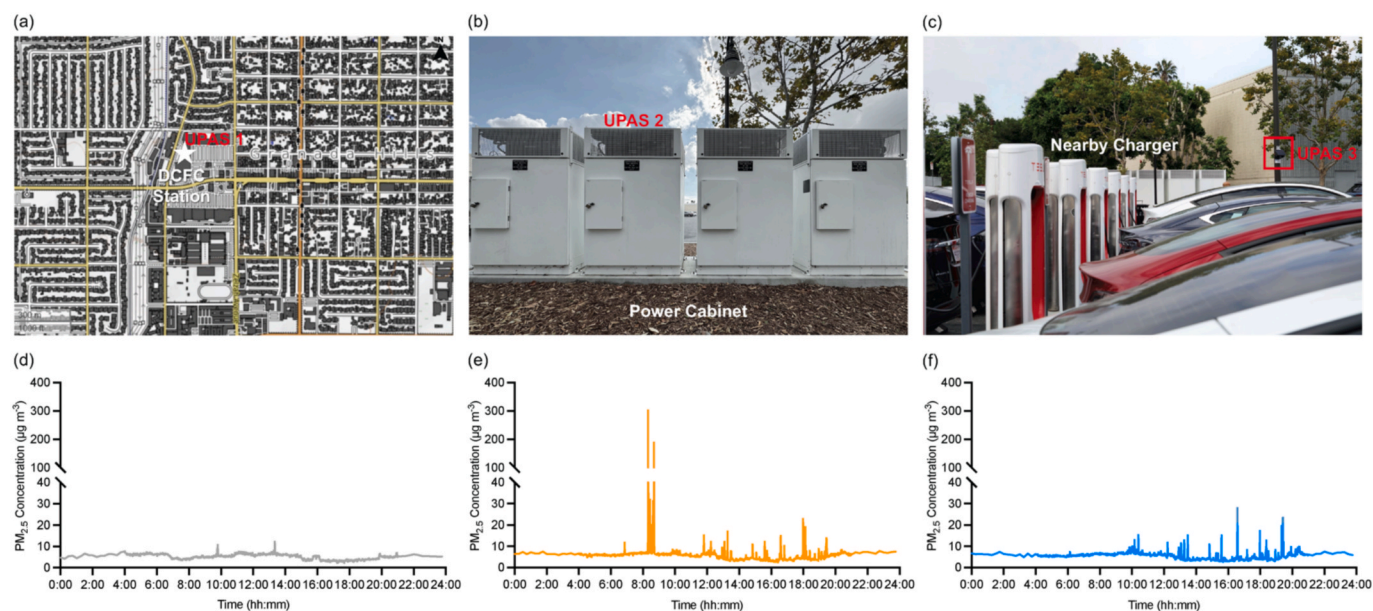


Fig. 2. Fine particulate matter ($PM_{2.5}$) concentrations measured at a downwind background site, a power cabinet, and a nearby upwind EV charger. (a) A map showing a typical Direct Current Fast Charging (DCFC) station located near residential areas in Los Angeles County along with the background sampling site for Ultrasonic Personal Aerosol Sampler (UPAS) No. 1. (b) A photograph of the power cabinet at the DCFC station and the sampling location for UPAS No. 2. (c) A photograph of the nearby EV chargers where $PM_{2.5}$ samples were collected by UPAS No. 3. (d) $PM_{2.5}$ concentrations at the background site. (e) $PM_{2.5}$ concentrations at the power cabinet. (f) $PM_{2.5}$ concentrations at the nearby EV charger.

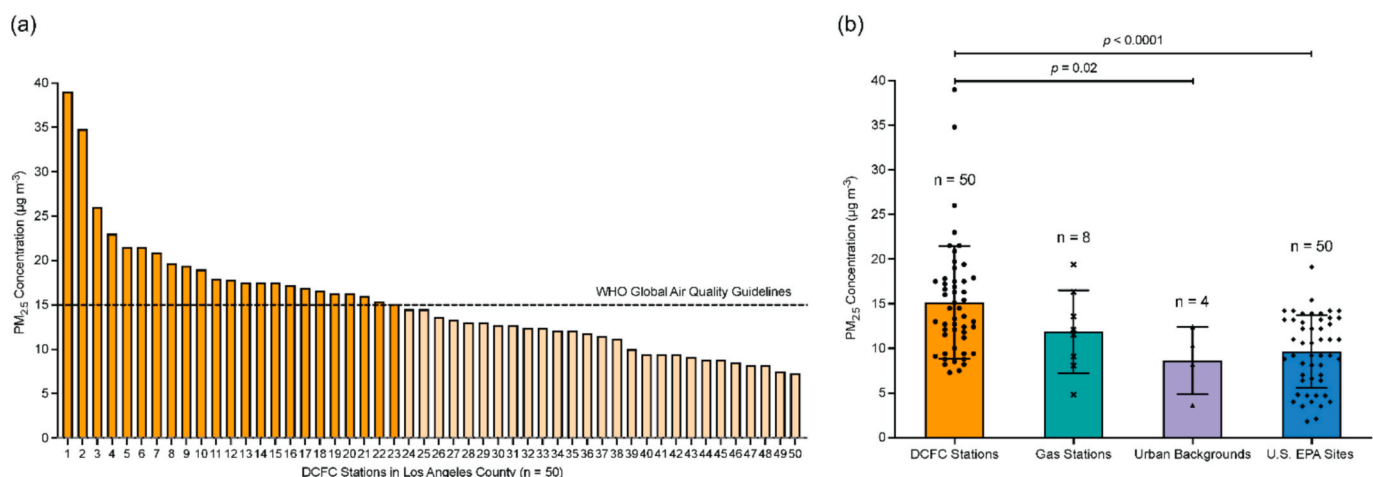


Fig. 3. Comparisons of 24-h mean fine particulate matter ($PM_{2.5}$) concentrations across different sampling locations. (a) Average 24-h $PM_{2.5}$ concentrations at 50 Direct Current Fast Charging (DCFC) stations in Los Angeles County. The black dotted line represents the World Health Organization (WHO) Global Air Quality Guidelines (AQGs) for a 24-h average of $PM_{2.5}$ at $15 \mu g m^{-3}$. (b) Average 24-h $PM_{2.5}$ concentrations at DCFC stations ($n = 50$), gas stations ($n = 8$), urban background sites ($n = 4$), and the nearest U.S. Environmental Protection Agency (EPA) outdoor air monitoring stations (11 stations total) ($n = 50$).

(Fig. 3b).

3.3. Potential mechanisms for particle emissions at DCFC stations

Given the unprecedented nature of these findings and the absence of previous literature, we investigated four potential particle emission and formation mechanisms associated with the specific characteristics of DCFC stations. These stations use converters inside power cabinets to switch Alternating Current (AC) to Direct Current (DC) and deliver power to EV batteries. This process results in the production of excessive heat (Rajendran et al., 2020), requiring thermal management of either air cooling or liquid cooling technologies to keep the temperature of power cabinet components within an operational range.

The first potential mechanism pertains to combustion, motivated by

the lack of empirical data on whether some DCFC stations use diesel engines for power. The second possible mechanism involves nucleation due to corona discharge, given the potential presence of high voltage and electromagnetic field at DCFC stations. The third mechanism is condensation, since coolants used in power cabinets may vaporize due to high temperatures and undergo a condensation process to generate aerosols. The final possible mechanism is resuspension, which occurs when particles previously deposited on surfaces inside power cabinets are reintroduced into the atmosphere through mechanical disturbances.

To test each of the potential particle emission mechanisms, we conducted real-time measurements of $PM_{2.5}$ mass concentrations, particle size distributions, and other air pollutants at the power cabinet of the specific DCFC station illustrated in Fig. 2. Ambient temperature and EV charging activities at each fast charger were also monitored

simultaneously. The results from this specific site are presented in Fig. 4, which shows time series plots of pollutant concentrations over a 5-hr period, along with ambient temperature and charging load—defined as the proportion of EVs actively charging relative to the total number of fast chargers at the station. As shown in Fig. 4a, we observed frequent $\text{PM}_{2.5}$ peaks, similar to those observed at the power cabinet in Fig. 2d, with peaks reaching approximately $40 \mu\text{g m}^{-3}$. EV charging activity data are also presented in Fig. 4a. Surprisingly, there appears to be no direct correlation between the number of EVs being charged at the station and $\text{PM}_{2.5}$ concentrations. A lag effect, however, seems to exist; for example, $\text{PM}_{2.5}$ levels increased around 9:00 am following a period of intense EV charging activity that started at 8:30 am. As discussed later in this paper, we hypothesize that EV charging activity is only one of several factors influencing the observed particle emissions.

To test the first mechanism (i.e., combustion), we measured carbon monoxide (CO) and carbon dioxide (CO_2) concentrations (Fig. 4b), indicators of combustion, which were found to be stable and comparable to urban background levels (Verhulst et al., 2017). We conducted a breakpoint detection analysis to identify moments when changes in air pollutant concentrations exceeded expected variability during the 5-hr period. A total of 52 breakpoints were observed in $\text{PM}_{2.5}$ concentrations (10.4 ± 5.0 per hour), while no breakpoints were detected for CO or CO_2 . This suggests the absence of combustion emissions at the power cabinet, indicating that DCFC stations are less likely to rely on diesel-powered generators. However, since our CO and CO_2 measurements were limited to the immediate vicinity of the DCFC station, we cannot rule out the possibility that a diesel generator was located downwind of the sampling inlet or farther away from the station.

To test the second hypothesized mechanism (i.e., nucleation due to corona discharge), we measured ozone (O_3) concentrations along with ambient temperature (Fig. 4d). This is because corona discharges can create reactive species such as O_3 , which can then participate in chemical reactions in the air, leading to particle formation (Chen and Davidson, 2002). As shown in Fig. 4d, O_3 concentrations gradually increase over time from below 10 ppb in the early morning to over 30 ppb

by noon, coinciding with the rise in ambient temperature. This is typical for a good air quality day in Los Angeles (Parrish et al., 2016). Like CO and CO_2 , there are no breakpoints for O_3 , which suggests that the corona discharge mechanism is unlikely.

To test the other two potential mechanisms (i.e., condensation and resuspension), we measured the particle size distribution in the range of 0.54 to $20 \mu\text{m}$ using the Aerodynamic Particle Sizer (APS). Fig. 4c shows time-resolved particle size distributions as a contour plot, where the x-axis represents the same period when other pollutant data were collected. The y-axis is the particle size, and the color intensity indicates the normalized particle number concentration (dN/dLogDp) for a given size at a given time. Particle sizes up to $1 \mu\text{m}$ are presented in Fig. 4c, as larger particles were rarely detected. As shown in Fig. 4c, the particle concentration and size distribution vary greatly over time, with the dominant sizes in the submicron size range (from 0.5 to $1 \mu\text{m}$). Additionally, there is an interesting pattern of decreasing particle mode diameter and generally increasing particle concentrations over time. Due to the size range of the APS, particles smaller than $0.54 \mu\text{m}$ cannot be detected. Although smaller particles contribute less to the $\text{PM}_{2.5}$ mass compared to larger particles (Saha et al., 2020), future studies are warranted to investigate their size distribution, particularly ultrafine particles (UFPs).

To test if these particles originate from the condensation of glycols (i.e., mechanism no. 3), which are most commonly used in DCFC cooling systems due to their excellent heat transfer properties (Chou, 2019), we measured particle volatility using our previously developed c-Air monitor (Luo et al., 2021; Çetintaş et al., 2022). This device employs computational microscopy and holography and has been used in our previous studies on electronic cigarette (e-cig) aerosols (Luo et al., 2021; Çetintaş et al., 2022). Fig. 5 illustrates the particle size distributions and their volatility for samples collected from three different sites: DCFC station (Fig. 5a), gas station (Fig. 5b), and urban background site (Fig. 5c). Similar to the data presented in Fig. 4c, particles detected at the DCFC station are predominantly in the submicron size range. Fig. 5d-f show the volatility of the particles determined by analyzing the size

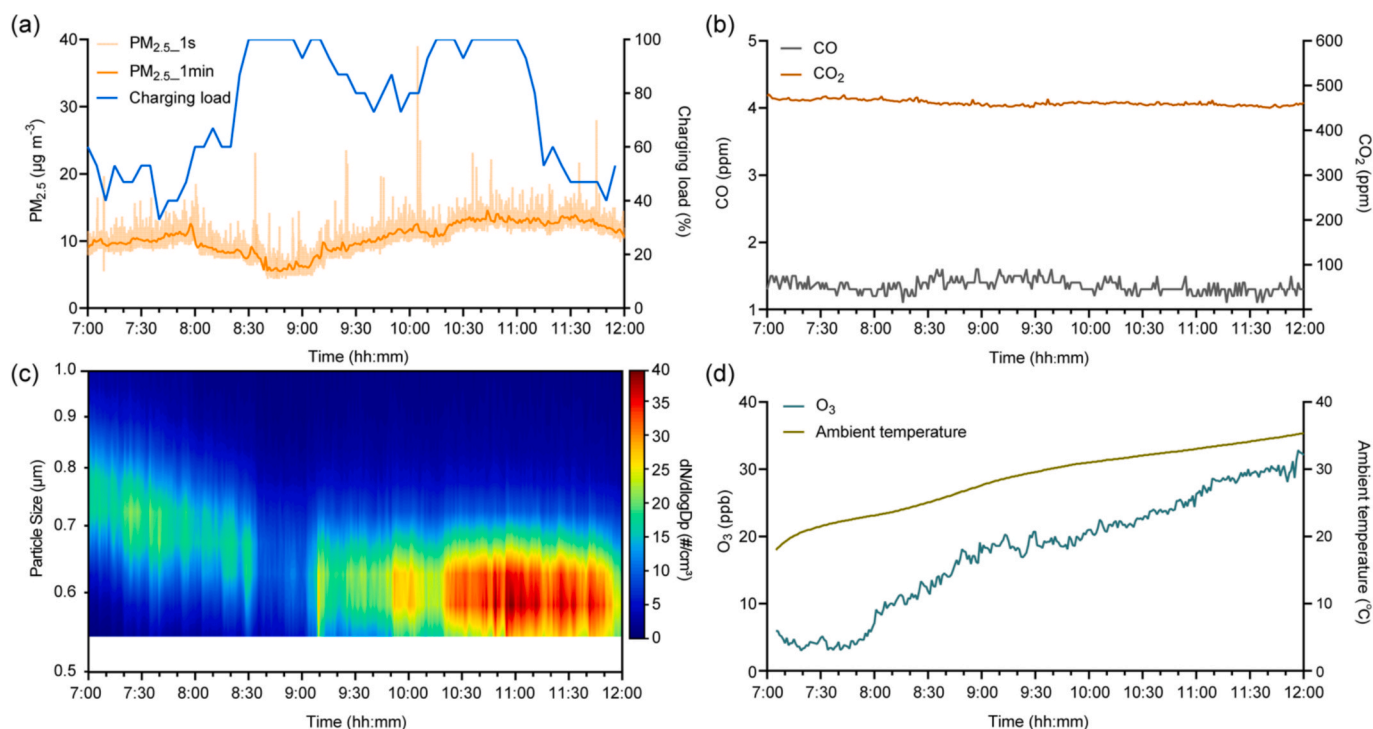


Fig. 4. Real-time measurements of air pollutants, electric vehicle (EV) charging load, and ambient temperature at the Direct Current Fast Charging (DCFC) station over a 5-hr period. Time series plots of (a) Fine particulate matter ($\text{PM}_{2.5}$) and charging load, (b) Carbon monoxide (CO) and carbon dioxide (CO_2), (c) Particle size distributions, and (d) Ozone (O_3) and ambient temperature.

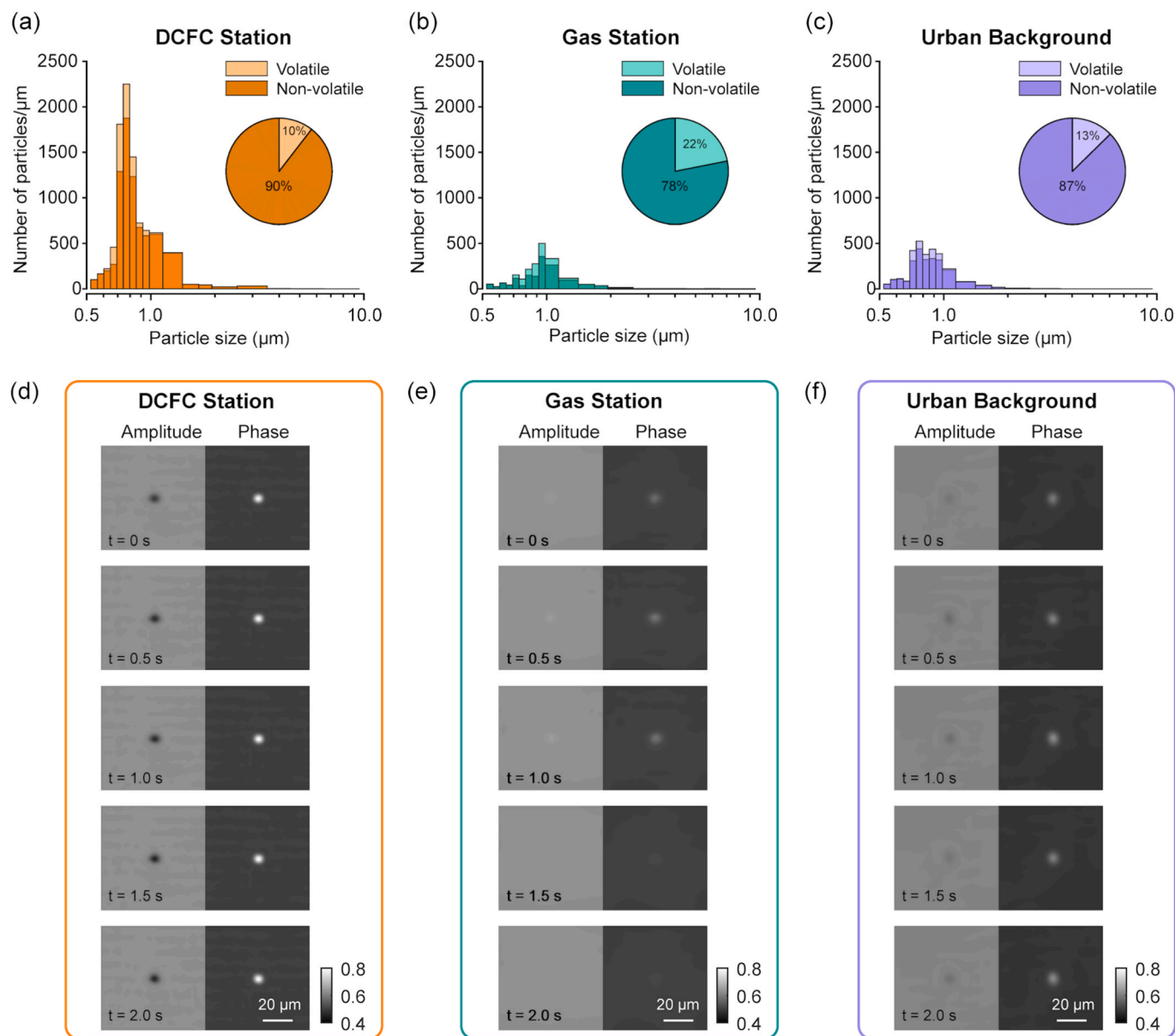


Fig. 5. Particle size distributions and volatility at the Direct Current Fast Charging (DCFC) station and comparison sites. Size distribution of volatile and nonvolatile particles from (a) DCFC station, (b) Gas station, and (c) Urban background site. The reconstructed holographic images (amplitude and phase) of particles collected from (d) DCFC station, (e) Gas station, and (f) Urban background site at different time points. The amplitude indicates the intensity of light scattered by the particles. Higher amplitudes result in brighter images, while lower amplitudes lead to dimmer images. The phase details the optical path length as light travels through the particles, with higher phases indicating larger particle sizes. The color bar reflects the reconstruction values of both amplitude and phase.

changes over time, as demonstrated in the images of the typical captured particles. For instance, Fig. 5e shows the volatilization process of a detected volatile particle at a gas station, where the phase of the particle vanished at $t = 2$ s, indicating the evaporation of the sample. Overall, among all collected particles from the DCFC station, gas station, and urban background site, those from the DCFC station have the lowest percentage of volatile particles (i.e., 10%). This contrasts sharply with e-cig particles. It is well established that liquid-based e-cig particles that are formed through the condensation of e-liquids primarily consisting of propylene glycol (PG) and vegetable glycerin (VG) show extremely high volatility (Çetintaş et al., 2022). Thus, condensation is unlikely to be the mechanism for these observed submicron particles.

3.4. $\text{PM}_{2.5}$ chemical composition at DCFC stations

This leads us to the last mechanism, resuspension. To test the

hypothesis of particle resuspension, we characterized the chemical composition of $\text{PM}_{2.5}$ at DCFC stations. From the 50 sampled DCFC stations, we selected five (shown in Fig. 1c) to collect two-week integrated $\text{PM}_{2.5}$ filter samples concurrently using UPAS devices powered by 20-watt solar panels (Voltaic Systems). At each station, three UPAS units were deployed simultaneously: one on top of the power cabinet, one near the EV charger, and one at a nearby background site. Table 1 presents $\text{PM}_{2.5}$ mass and element concentrations, with a focus on brake and tire wear and dust tracers at the sampling locations. The average two-week $\text{PM}_{2.5}$ concentration at power cabinets was $12.0 \pm 3.0 \mu\text{g m}^{-3}$, which was 1.22 and 1.56 times the $\text{PM}_{2.5}$ levels at nearby chargers and background sites, respectively. $\text{PM}_{2.5}$ elemental concentrations of brake and tire wear tracers (Ba, Cu, Zn) at the power cabinet were 1.35–1.70 times those at the nearby EV charger, and 2.40–2.96 times those at the background site.

Similarly, dust tracers such as Ca, Al, and Fe were 1.65–1.95 times as

Table 1PM_{2.5} mass and element concentrations at power cabinets, nearby chargers, and background sites at five DCFC stations.

DCFC Station	Sampling location	PM _{2.5} Mass ($\mu\text{g m}^{-3}$)	Brake and Tire Wear Tracer (ng m^{-3})			Dust Tracer (ng m^{-3})		
			Ba	Cu	Zn	Ca	Al	Fe
1	Power cabinet	9.9	6.2	3.5	3.9	64.7	33.3	67.5
	Nearby charger	8.9	3.4	1.3	2.0	38.3	11.6	35.5
	Background site	5.7	2.5	0.9	1.6	22.4	7.6	31.8
2	Power cabinet	8.4	3.5	4.3	2.2	54.2	15.8	38.3
	Nearby charger	8.2	2.7	1.1	2.0	40.2	10.9	30.5
	Background site	5.7	1.7	0.7	1.3	30.9	12.8	21.1
3	Power cabinet	11.6	7.8	2.9	7.5	95.4	35.4	125.2
	Nearby charger	10.2	7.2	2.5	5.5	69.5	48.2	96.1
	Background site	8.8	6.3	2.2	3.5	66.9	38.8	75.7
4	Power cabinet	15.2	13.6	5.2	11.9	153.7	110.2	190.7
	Nearby charger	10.2	8.9	3.4	5.4	66.2	41.0	91.5
	Background site	9.8	7.4	2.5	4.2	56.7	24.4	78.9
5	Power cabinet	14.7	23.1	6.9	11.3	161.2	128.5	265.7
	Nearby charger	11.3	17.9	5.2	7.3	83.7	54.5	161.7
	Background site	8.6	4.4	1.5	2.0	57.8	20.2	55.8
Average level at power cabinets		12.0 $\pm$ 3.0	10.8 $\pm$ 7.8	4.6 $\pm$ 1.6	7.4 $\pm$ 4.3	105.8 $\pm$ 49.6	64.6 $\pm$ 50.9	137.5 $\pm$ 92.4
Average level at nearby chargers		9.8 $\pm$ 1.2	8.0 $\pm$ 6.1	2.7 $\pm$ 1.7	4.4 $\pm$ 2.4	59.6 $\pm$ 19.7	33.2 $\pm$ 20.6	83.1 $\pm$ 53.5
Average level at background sites		7.7 $\pm$ 1.9	4.5 $\pm$ 2.4	1.6 $\pm$ 0.8	2.5 $\pm$ 1.3	46.9 $\pm$ 19.2	20.8 $\pm$ 12.0	52.7 $\pm$ 25.8
Power cabinets/nearby chargers		1.22	1.35	1.70	1.68	1.78	1.95	1.65
Power cabinets/background sites		1.56	2.40	2.88	2.96	2.26	3.11	2.61
Nearby chargers/background sites		1.27	1.78	1.69	1.76	1.27	1.60	1.58

high at the power cabinet as at the nearby EV charger, and 2.26–3.11 times as high as at the nearby background site. It is reasonable to expect that samples collected near EV chargers would exhibit higher levels of non-exhaust emissions, including brake wear, tire wear, road surface wear, and resuspended road dust (Fussell et al., 2022), compared to background sites due to the frequent braking and charging activities associated with EVs. Notably, non-exhaust particle levels were even more concentrated near power cabinets. These results indicate that these particles are enriched by brake and tire wear, as well as dust, further supporting the hypothesis of particle resuspension at these stations.

4. Discussion

In this study, we found that power cabinets at DCFC stations may emit PM_{2.5}, revealing a previously overlooked source of air pollution within EV infrastructure. Although DCFC stations are intended to support EV adoption and reduce GHG emissions, our results suggest they may also pose localized air quality risks. Despite the rapid expansion of EV infrastructure, its potential environmental and public health impacts remain underexplored.

Our findings suggest that particle resuspension is the most likely mechanism behind the observed particle emissions at the power cabinet, for the following reasons. First, a large variety and amount of PM from both natural and anthropogenic sources, such as pollen and vehicular exhaust, along with non-exhaust emissions (e.g., brake, tires, and road wear) (Fussell et al., 2022) are continuously released into the atmosphere. These PM can deposit on various surfaces, providing an “infinite” source of particles to be resuspended (Henry et al., 2023). Thus, particle accumulation on the internal surfaces of power cabinets is highly likely. In fact, some power cabinets use filters to prevent ambient PM from entering the system (Lovrak, 2023; DC 360kW Power Cabinet, 2024). Second, power cabinets employ either air cooling or liquid cooling technologies to reduce system heat, both utilizing mechanical fans to dissipate the heat into the ambient air (Power converter cooling, 2024). This could create forced convection and generate turbulence, detaching the settled particles and causing them to resuspend in the atmosphere (Theron et al., 2020). Finally, an intriguing observation from Fig. 4c is the mode diameter of detected particles, which decreases from about 0.75 μm to approximately 0.6 μm over the 5-hr sampling period. This pattern aligns with the particle resuspension theory, which indicates that smaller particles are more difficult to detach from surfaces due to the size-dependent nature of the forces involved (Hinds and Zhu,

2022). Specifically, the particle resuspension process is governed by adhesive forces, which are proportional to the particle diameter (d), and removal forces, which, in the case of DCFC, are primarily driven by air current and are proportional to d^2 . This observed shift in particle mode diameter suggests that larger particles are more easily resuspended first as the DCFC station begins operation in the morning, likely because a substantial amount of dust has accumulated overnight. As the charging activity intensifies and becomes more continuous, smaller particles are also resuspended, but at a later stage.

It should be noted that dust resuspension is a complex process that depends on a range of variables, including surface type, dust loading, particle size, air velocity, relative humidity, and electrostatic fields (Tian et al., 2014; Qian and Ferro, 2008). In the case of DCFC stations, this process is likely influenced by intertwined factors such as the amount and size of particles accumulated on the internal surface of power cabinets, charging activities, fan speed, ambient temperature, and relative humidity among others. While our data support the notion that particle resuspension is a potential mechanism, the unprecedented nature of these observations warrants future studies to investigate the influencing factors.

While no study we are aware of has specifically documented particle resuspension due to DCFC stations, extensive literature exists on road dust resuspension (Henry et al., 2023), particularly in recent studies focusing on non-exhaust emissions from mobile sources (Fussell et al., 2022). Exposure to these resuspended particles has been linked to various health issues, including respiratory and cardiovascular diseases (Khan and Strand, 2018). With the shift toward EVs, which do not produce exhaust emissions, the role of non-exhaust emissions in urban air pollution is expected to become more important (Fussell et al., 2022; Wang et al., 2023). This highlights the need for addressing particle resuspension in air quality management to protect public health, as society moves rapidly toward vehicle electrification (Kupiainen, 2016; Casotti Rienda and Alves, 2021). Our research suggests that DCFC stations could be a notable source of particle resuspension, thus expanding the scope of sources considered in urban PM pollution studies.

With the U.S. government's target to establish 500,000 EV chargers by 2030, and with the rapid ongoing EV adoption worldwide, the anticipated increase in DCFC stations raises concerns about potential environmental and health impacts of this unexpected particle emission on a population scale. Implementing regulations and engineering controls in the design and construction of future DCFC stations is crucial to effectively mitigate particle emission from power cabinets. For example,

while some DCFC stations have filters in the air inlet to prevent PM from entering the unit, manufacturers could also add filters to prevent resuspended particles from re-entering the atmosphere. Moreover, strategic site selection should emphasize placing DCFC stations at a distance from schools, nursing homes, and densely populated residential areas to minimize potential risks. In addition, future studies should investigate the toxicity of these particles to better understand their potential health risks. More broadly, these findings emphasize the need for continuous monitoring of major energy transitions that may result in unintended risks to public health before major investments are made (Yao et al., 2025).

5. Conclusions

In summary, we found that daily $\text{PM}_{2.5}$ concentrations ranged from 7.3 to 39.0 $\mu\text{g m}^{-3}$ across 50 DCFC stations in 47 cities throughout Los Angeles County, California, and our extensive real-time field measurements suggest that particle resuspension at power cabinets is the most likely mechanism responsible for the observed particle emissions at these stations. Additionally, we observed elevated levels of brake and tire wear tracers (Ba, Cu, Zn) and dust tracers (Ca, Al, Fe) in $\text{PM}_{2.5}$ samples collected near power cabinets compared to those from nearby chargers and background sites. These results highlight an overlooked air quality concern in electric vehicle infrastructure. By identifying power cabinets as a previously unexamined emission source, our study provides critical insights into particle levels near DCFC stations, informing future regulations and interventions to support the transition to sustainable energy. As these newly identified particles may differ from other ambient pollutants and have largely unknown toxicity and health effects, future studies are warranted to confirm these findings and investigate the health implications of elevated $\text{PM}_{2.5}$ levels near DCFC stations.

Author contributions

Y.Y., M.N., M.J., and Y.Z. designed the research. Y.Y., M.N., H.C., Y. L., and Y.Z. performed field sample collection and analysis. Q.Y. and Q. W. conducted data analysis on electric vehicle fast chargers. Y.Z. supervised the research. Y.Y., M.N., and Y.Z. wrote the manuscript. A.O., M.J., and Y.Z. reviewed and edited the manuscript.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary material

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.envint.2025.109581>.

Data availability

Data will be made available on request.

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