Micro-morphological analysis of foliar uptake and retention of airborne particulate matter (PM)-bound toxic metals: implications for their phytoremediation

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\textbf{ABSTRACT}

While airborne particulate matter (PM) pollution is a serious problem for urban environments, it can be reduced through uptake by plant leaves. In this study, we investigated and compared the uptake of PM-bound toxic metals by different plant species. Enrichment factor (EF) and correlation analyses across different sample types indicated anthropogenic origins of these toxic metals with airborne source signatures. Scanning electron microscopy (SEM) analyses of both leaf surfaces (adaxial and abaxial) suggested that the micro-morphological properties of the leaf surface (e.g. stomata, trichomes, epicuticular wax, and epidermal appendages) control the accumulation of PM and associated metals in plant leaves. \textit{Senna siamea} leaves showed the most micro-morphological variation as well as the maximum concentration of toxic metals. It was found that foliar uptake of PM-bound toxic metals is affected by leaf surface morphological characteristics. Our results imply that plant speciation strategies can be used to help decrease PM pollution.

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\textbf{Introduction}

Air pollution is the most pressing problem in urban environments. Air pollutants are categorised into various components, of which PM is one of the most significant. PM is further categorised based on particle diameter as suspended/large PM (10–100 \textmu m), respirable/coarse PM (2.5–10 \textmu m), fine PM (0.1–2.5 \textmu m), and ultrafine particles (UFPs) (<0.1 \textmu m) \cite{1}. PM\textsubscript{2.5} (fine PM) was estimated to cause approximately 4.2 million premature deaths worldwide in 2016 and Most of these deaths occurred in middle- and low-income countries \cite{2}. Particles ranging in size from respirable to UFP can enter the human body and cause...
several chronic issues such as respiratory symptoms and cardiovascular disease. PM can also contain a number of toxic substances such as airborne toxic metals [3]. Transportation and industrialisation activities are responsible for the emission of airborne toxic metals in association with PM of different sizes into the air.

According to an estimate, 91% of the global population lives in areas where air quality is below world health organisation (WHO) guidelines. WHO air quality guidelines state that the annual average concentration of PM$_{2.5}$ in the atmosphere should not exceed 10 µg·m$^{-3}$ [4]. The concentrations of PM around 35 µg·m$^{-3}$ are often seen in most urban areas [5]. In 2021, air quality assessments were conducted in the region of Europe by 42 countries covering a total of 1588 cities. Compared to 2020, air quality improved in 14 countries and declined in 25 countries. However, only 55 cities were able to meet the recommended annual 2021 air quality guideline of WHO for PM$_{2.5}$ (5 µg·m$^{-3}$) [6]. Moreover, only 3.9% of cities in United States and Canada meet WHO’s annual PM$_{2.5}$ level guidelines [6,7]. 70% of worldwide wildfires were caused by the burning of African grasslands, which emit large amounts of ambient PM$_{2.5}$ [8]. It has been calculated that air pollution cause about 780,000 premature deaths per year in Africa [9]. Moreover, most polluted African city did not comply with the guideline values. For instance, the annual average PM$_{2.5}$ concentration in N’Djamena (one of the largest cities in Chad, Africa) was 77.6 µg·m$^{-3}$ which was more than 10 times the WHO air quality guideline value [8]. The countries of Oceania showed the world’s cleanest overall air quality. In this region, 1 city in New Zealand, 1 city in New Caledonia, and 46 cities in Australia complied the air quality guideline [8].

Air pollution has become a serious health concern in developing Asian countries [10–13]. According to a comprehensive study conducted in 230 Asian cities, about 80% of the studied cities had annual average PM$_{10}$ level higher than the acceptable level as defined by the WHO guideline (Interim Target-2, 50 µg·m$^{-3}$). Moreover, only two of these cities [Pekanbaru and Palangkaraya (Indonesia)] had an annual average PM$_{10}$ level within the WHO air quality guideline (20 µg·m$^{-3}$) [14,15]. In recent years, the annual mean PM$_{10}$ values measured in northern India were within the WHO and National Ambient Air Quality Standards (NAAQS), while most cities are critically polluted [16]. Moreover, Greenpeace India (2017) reported that 1.2 million people are dying annually due to air pollution in India. In India, the average respirable suspended particulate matter (RSPM) concentration is 160 µg·m$^{-3}$, which is higher than the NAAQS for many different categories (i.e. 120 µg·m$^{-3}$ for industrial, 60 µg·m$^{-3}$ for residential, and 50 µg·m$^{-3}$ for sensitive areas) [17].

This increase in airborne PM has been suggested to be due to various sources including vehicular emission, road dust resuspension, wear and tear of brakes and tires, industrialisation, and dust from construction [18,19]. Since toxic/heavy metals (e.g. Pb, Cd, Cu, Zn, Cr, and Ni) bind to PM, their presence in PM can increase exposure toxicity [20–22]. Hence, there is a pressing need to control the levels of PM and its associated toxic metals in urban environments.

In the urban environment, road side vegetation plays an effective role in reducing PM pollution [23,24]. Aerial plant parts such as branches, bark, stems, and especially leaves are able to reduce PM pollution via adsorption [25,26]. Moreover, the large surface areas of leaves enable them to accommodate increased metal deposition on their surfaces [27]. However, the ability to accumulate metals can vary across different plant species [28,29]. This capacity can be determined by qualitative (epidermal cells, cuticular
composition and structures, texture of both surfaces (adaxial and abaxial), and stomata type) and quantitative (numbers of stomata and trichomes) micro-morphological leaf features [30–32]. Hence, micro-morphological studies of leaf surfaces can be employed to assess the PM pollution load in urban environments [29,32].

Although the accumulation of these toxic metals in plant roots via soil has been extensively studied, relatively limited research has focused on the uptake pattern of toxic metals by plant leaves through the airborne route. The aim of the present study was to assess the accumulation potential of airborne toxic metals by plant leaves from different locations of an urban environment in India. Plants naturally growing in six different sites of Bilaspur city were selected for detailed study. To this end, Pb, Cd, and Cu were selected as target metals for analysis in different plant and environmental samples. This is because Pb, Cd, and Cu were commonly found in PM deposited at leaf surfaces of different plant species from various urban locations, especially from the study area [29, 33–37]. Furthermore, it has been reported that Pb and Cd are listed as hazardous air pollutants (HAPs) in the list of USEPA [38,39]. Traffic and industrial emissions were found responsible for Cu build-up in urban road side environments [40,41]. Vehicular activities including exhaust emissions, tyre wear, brake wear emission, metallic corrosion of engine wears of running vehicles, and coal combustion in industrial activities also release these metals in urban environment [42–45]. Therefore, these toxic metals can easily bind to suspended PM and further deposit leaf surfaces of roadside plants [25,46]. As such, the objectives of this study were (1) to assess the abundance of toxic metal content (Pb, Cd, and Cu) relative to Fe content in order to derive an enrichment factor in the leaves of the selected plants for foliar dust, road dust, and soil samples; (2) to explore the metal source processes; and (3) to assess the effect of micro-morphological leaf surface properties (adaxial and abaxial) on the accumulation of PM and associated toxic metals. Our findings offer valuable insights into the role of plant species in controlling PM pollution in urban environments.

Materials and methods

Study site

The study was conducted in Bilaspur (E = 82°08’28.32, N = 22°05’9.6), which is located in the state of Chhattisgarh in central India. The selected area included the city centre and covered both commercial and residential areas of the city. All sites are major squares of the city with strong traffic source activity (Figure S1). Site 1 is the traffic square located near Govt. Girls Higher Secondary School. This site was affected by transport activities of both light and heavy vehicles. Site 2 is a market square located near a bridge that connects the city. Site 3 and site 4 are located on National Highway (NH) 49, which has high traffic-related emission sources. Site 5 is located near Apollo Hospital Square, which connects the Rajkishor Nagar, Vasant Vihar, and Lingyadih residential areas. Site 6 is located on a city bypass road that connects NTPC (National Thermal Power Corporation) Sipat to NH130 Raipur Road.

Selection of plant species

Six angiospermic plant species with different leaf surface characteristics were selected for this study (Ficus religiosa, Mangifera indica, Butea monosperma, Alstonia scholaris,
Azadirachta indica, and Senna siamea). These species are widespread, remain green year-round, and are very common in subtropical areas.

*F. religiosa* (Family: Moraceae) is a species of large evergreen trees. Its leaves are generally 5–12 cm long and 5–8 cm wide, with alternate reticulate intercostae, a leathery appearance, an ovate-lanceolate (caudate) shape, and a green colour when they are mature.

*M. indica* (Family: Anacardiaceae) is considered to be the most delicious Indian fruit plant. Its evergreen trees are erect with broad canopies, and its leaves are alternate and clustered at the tip of the branch. Its leaves are 12–20 cm long and 6–8 cm wide, pinnate, and have reticulate/intercostae venation.

*B. monosperma* (Family: Fabaceae) is a medium-sized deciduous tree with a bent trunk and irregularly arranged branches. The leaves are compound, alternate/spirally arranged on the stem, trifoliate, 8–12 cm long, and 5–7 cm wide. They also have an acute apex with an ovate shape, reticulate venation, and a prominent midrib.

*A. scholaris* (Family: Apocynaceae) is a tall ornamental tree with rough grey bark. The branches emerge from the same place in the main stem and are arranged in a whorled pattern. The leaves are leathery, and one whorl contains 4–7 leaves. Leaf shape is similar to that of leaves on the mango tree.

*A. indica* (Family: Meliaceae) is an evergreen tree that grows vertically erect and has greyish brown bark. Its leaves are imparipinnate and alternate with 12–20 cm long rachis. The leaflets generally contain 10–14 leaves, which are arranged in opposite/sub-opposite positions. The leaves are 4–6 cm long, 2–3 cm wide, and have a serrate margin. One pair contains 12–20 leaves. The leaves also display intercostae reticulate venation with a prominent midrib.

*S. siamea* (Family: Fabaceae) is a deciduous tree of medium height, up to 15–20 m. It has a wide spreading canopy, and its leaves are pinnately compound, 25–35 cm long, and 10–15 cm wide. The leaflets are arranged in the opposite position in 10–14 pairs. The leaves are dark green and have an ovate shape, entire margins, and reticulate venation [47, 48].

In the present study, all plants showed similar phenological conditions. Therefore, at the time of collection, the leaves of the different plants were similarly matured. Special care was also taken at the time of sampling to select leaves equally in terms of morphological damage, bird faeces, dynamics such as chlorosis and necrosis, and uneven dust deposition.

**Sampling of leaves, foliar dust, road dust, and soil samples and sample preparation**

Weather conditions were relatively stable within 2 weeks before sampling. The average temperature was 23.3°C and average relative humidity was 49.8%. Rainfall was not recorded prior to two weeks of sampling (Source: [https://www.worldweatheronline.com/](https://www.worldweatheronline.com/)). Dusted leaves from each site were collected at a height of 2.5 m above the ground. Ten leaves were collected each study site from branches facing towards the road side. Steel trays and fine plastic hair brushes were used to collect road dust from
the edge of road. Soil samples from each site were collected from the surface at a depth of 40 cm by the scraper plate method [18]. Considering the average depth of the plant fine roots, the soil-surface was dug to the same depth. Four soil samples (10 g each) were collected from each site from the same profile. All four samples were mixed to create a single uniform sample that represented the soil of that particular site. All samples were kept in air-tight Ziploc polyethylene bags and brought to the laboratory. The sampled dusted leaves were well stirred in a 200 mL glass beaker with 100 mL micro-distilled water for 10 min. In this suspension, foliar dust particles sticking to the surfaces of the leaves were separated. To obtain dust, the suspension was vaporised at 50°C on a hot plate. The washed leaves were oven dried for 24 h at 50°C. After drying, the leaves were crushed with a mortar and pestle and passed through the sieve to obtain a fine powder. Road dust and soil samples were also kept at room temperature for 2 days, after which they were ground with a mortar and pestle and collected into separate beakers by weighing 0.5 g of each. All samples were digested separately into aqua-regia solution (3HNO₃: HCl); sample preparation for metal analysis was performed according to the method of [18].

Analysis of metals

After sample preparation, heavy metal concentrations were determined using flame atomic absorption spectroscopy (AA 7000, Shimadzu, Japan). Standard solutions of the target metals were obtained from Inorganic Ventures (USA) (Fe: AAFe-1, 1000 ± 10 µg·mL⁻¹, 2% (v/v) HNO₃ traceable to NIST-SRM 31269, Pb: AAPb-1, 1000 ± 10 µg·mL⁻¹, 0.5% HNO₃ traceable to NIST-SRM 3128, Cd: AACd-1, 1000 ± 10 µg·mL⁻¹, 3% (v/v) HNO₃ traceable to NIST-SRM 3108, and Cu: AACu-1, 1000 ± 10 µg·mL⁻¹, 3% (v/v) HNO₃ traceable to NIST-SRM 3114). Five-point calibration curves were obtained by serial dilution of metal standard solutions and used to quantify the concentrations of the four metals (Pb, Cd, Cu, and Fe) in different samples after correction for the blank value. The concentrations of each metal (µg·g⁻¹) were analyzed in triplicate. The detection limits (DL: µg·g⁻¹) of the four target metals were 0.04 µg·g⁻¹ (Fe), 0.08 µg·g⁻¹ (Pb), 0.006 µg·g⁻¹ (Cd), and 0.009 µg·g⁻¹ (Cu). The precision, when expressed in terms of relative standard deviation (RSD %), was below 5% for all metals. The average recovery of different metals in triplicate analyses from randomly selected fortified samples (soil) was as follows: Fe (88.8 ± 3.80%), Pb (104.0 ± 2.84%), Cd (98.5 ± 3.98%), and Cu (113.5 ± 4.30%).

Enrichment factor (EF)

The EF assesses the relative contribution of manmade sources to toxic metal levels in collected samples. The EF was calculated according to equation (1)

\[
EF = \frac{(S_{(E)}/S_{(R)})_{leaf}}{(C_{(E)}/C_{(R)})_{crust}}
\]

where \(S_{(E)}\) is the concentration of a target metal \((E)\) in the examined environmental sample, \(S_{(R)}\) is the concentration of the reference metal in the examined environmental sample, \(C_{(E)}\) is the concentration of a target metal \((E)\) in the crust, and \(C_{(R)}\) is the concentration
of the reference metal in the crust. The $C_{(E)}$ values for Pb, Cd, and Cu were 20, 0.098 (98 ppb), and 25 ppm, respectively. Fe was used as the reference metal ($C_{(R)}$) with a crustal value of 35,000 ppm (3.5 wt. %) [49]. Based on EF values, the following categories were used for pollution levels and source (natural or anthropogenic): $EF < 2 = \text{low enrichment}$, $2 \leq EF < 5 = \text{moderate enrichment}$, $5 \leq EF < 20 = \text{significant enrichment}$ (low anthropogenic emission), $20 \leq EF < 40 = \text{very high enrichment}$ (moderate anthropogenic emission), and $EF \geq 40 = \text{extremely high enrichment}$ (high anthropogenic emission) [50].

**Scanning electron microscopy**

Fresh leaves were thoroughly washed with ionised water, blotted dry, and then cut into square sections (about 0.5 cm wide x 0.5 cm long) using razor blades. The leaves were cut from the margins and the mid-rib area. The leaf samples were kept in a mixture of 2.5% glutaraldehyde solution overnight to enable pre-fixation, after which the samples were kept in 2% osmium tetroxide for post-fixation for one hour. The samples were washed twice with phosphate buffer solution for 15 min and then passed through a series of acetone solutions (30%, 50%, 70%, 95%, and 100%) for dehydration. Drying was performed in a critical point drier (CPD) using $CO_2$ as a carrier gas. An aluminum stub with an adhesive surface was used to mount the leaf samples. The stub was coated with 40–60 nm gold using a Denton Vacuum. SEM analysis of samples was carried out with an FEI NOVA NANO SEM-450 scanning electron microscope equipped with different detectors.

**Statistical analysis**

The statistical analyses were carried out using the statistical package for the social science, version 22 (SPSS-22, IBM, Chicago, USA). One-way Analysis of variance (ANOVA) was applied to analyze the significant differences in concentrations of target toxic metals measured in different sample types. The level of significance ($p$ value) was set at different levels (<0.05 and <0.001). The correlation coefficient was used to determine different significance level ($p < 0.05$ to $p < 0.01$) between leaf vs. foliar dust, road dust, and soil samples.

**Results and discussion**

**Concentrations of metals in different matrices**

The concentrations of the four target metals in the four sample types are shown in Figure 1. Site 6 (S. siamea) showed the highest concentrations of all metals in foliar dust (Table S1). The maximum concentration of Fe in foliar dust in the present study was more than 10 times higher than that observed in a previous study in Bilaspur, whereas it was nearly 2 times higher for Pb [20]. The maximum concentration of Cd in foliar dust was comparable with that observed in Miskolc (industrial city), Hungary ($4 \pm 1 \text{ mg kg}^{-1}$: Platanus × acerifolia) [27]. This comparative study was performed with samples from high traffic and active urban sites and showed traffic-related sources for these toxic metals.
The maximum concentration of Pb in *S. siamea* in this study was comparable with a previous study (16.0 ± 0.80 μg·g⁻¹: *S. siamea*) [18]. However, it was more than 3 times higher than the concentration reported in Mashhad, Iran (4.5 ± 1.9 μg·g⁻¹: Platanus orientalis) and 2 times higher than that reported in Córdoba, Argentina (6.89 μg·g⁻¹: Tillandsia capillaris) [51,52]. For Cd, the maximum concentration (0.75 ± 0.09 μg·g⁻¹: *A. scholaris*) was comparable with that reported in Okayama, Japan (0.82 ± 0.13 μg·g⁻¹: Rhododendron Pulchrum: S7) [53]. However, it was nearly 75 times higher than that reported in Shanghai, China (0.109 ± 0.060 μg·g⁻¹: Pittosporum tobira) [26].

In road dust, the maximum concentrations for Pb (79.8 ± 1.47 μg·g⁻¹) and Cu (171 ± 2.65 μg·g⁻¹) were found at site 4 (*A. scholaris*). However, the maximum concentrations were relatively lower and comparable to those for Pb (114.82 μg·g⁻¹) and Cu (184.42 μg·g⁻¹) in urban road dust of the Pearl River Delta, South China, respectively [54]. The maximum concentration of Cd (2.72 ± 0.72 μg·g⁻¹) was found at site 6 (*S. siamea*); this concentration was nearly 2 times higher to those reported in Pearl River Delta, South China (1.59 μg·g⁻¹) and Jiaozuo, China (1.25 μg·g⁻¹) [54,55]. Moreover, the maximum concentration was around 4 times higher than that reported in Beijing, China (0.72 ± 0.74 μg·g⁻¹) [56].

![Figure 1](image.png)

**Figure 1.** Concentrations of metals in the four environmental samples at the six study sites (μg·g⁻¹, *n* = 3). The differences in metal concentrations across different sample types were analyzed by a one-way ANOVA (the level of significance is shown by asterisks).
In soil, the maximum concentrations of Fe (36,942 ± 2.08 μg·g⁻¹), Pb (11.6 ± 0.97 μg·g⁻¹), and Cd (0.21 ± 0.04 μg·g⁻¹) were found at site 2 (M. indica). These concentrations were lower than those reported in Suva, Fiji (Fe = 39,525.5 μg·g⁻¹, Pb = 59.30 μg·g⁻¹, Cd = 3.10 μg·g⁻¹) [57,58]. However, the maximum concentration of Cu (25.4 ± 2.37 μg·g⁻¹, Site 6) was 10 times lower than those reported in Suva, Fiji (265.7 μg·g⁻¹) [57].

Source apportionment of metals and relationships between the different matrices

The results of ANOVA showed the significant differences (p < 0.05 and 0.001 level) among concentration of metals in road dust, leaf, and foliar dust sample within study sites (Figure 1). A correlation analysis was conducted between metal concentrations within different samples to determine their sources of origin. In foliar dust, significant correlations were found between Pb vs. Cd (r = 0.93, p < 0.01), Pb vs. Cu (r = 0.85, p < 0.05), and Cd vs. Cu (r = 0.89, p < 0.01) [Table S2a (ii)]. Similar results were found in leaf and road dust samples [Tables S2a (ii) and (iii)]. However, in soil, the correlations were not significant, implying negligible anthropogenic influence. A correlation analysis was also conducted between leaf and other samples (foliar dust, road dust, and soil) (Table S2b). All metals (Fe, Pb, Cd, and Cu) showed significant correlations (p < 0.05, 0.01) between leaf vs. foliar dust and leaf vs. road dust.

Enrichment factor (EF)

To identify the sources of toxic metals in the four samples (foliar dust, leaves, road dust, and soil) and thus determine whether the toxic metals are derived from anthropogenic activity or natural sources, the EFs were calculated (Table S3). In foliar dust, the EFs for Pb, Cd, and Cu showed moderate to very high enrichment, which suggests anthropogenic origin. In leaf samples, the EFs of Pb and Cu showed similar levels of enrichment (significant enrichment). This finding was comparable with the EF value of Tilia spp. and Aesculus hippocastanum leaves (EF > 10) isolated from an urban area in Belgrade, Serbia [59]. However, for Cd, the EFs ranged from 22.2 (B. monosperma) to 81.1 (A. scholaris). With the exception of site 3 and site 5, all other sites showed extremely high enrichment (EF ≥ 40), with maximum enrichment at site 4, followed by site 6, site 2, and site 1. A similar trend was observed for the EFs in leaves of different Ficus species at Dhaka city, Bangladesh (e.g. leaf of F. benghalensis: Pb = 18.54–94.83; leaf of F. aurea: Cd = 32.70–2167.29), and in the mosses in Zlatibor, Serbia (Pb = 25.2 and Cd = 22.5), which reflects anthropogenic origin [50,60]. The high EF value of Cd in the S. siamea leaf samples is comparable with a previous study [18]. In road dust, Pb and Cu showed moderate to significant EFs. The EFs of Cd ranged from 9.20 (B. monosperma) to 25.7 (S. siamea), indicating significant to very high enrichment. A similar pattern of Pb and Cu EFs in road dust was observed by [61], which was mainly associated with traffic emission and vehicular part corrosion. The road dust EFs of Pb and Cu were comparable with the road dust EFs from Delhi, India, which indicated an anthropogenic source of origin [62]. In soil samples, all selected metals had an EF of 2.
or below, suggesting that they originated from natural sources or were background concentrations.

**SEM-based micro-morphological study**

The micro-morphological parameters considered for each plant are listed in Table 1.

**F. religiosa**

In *F. religiosa*, emergence was visible at the adaxial surface of the mid-rib portion of the leaves, which can facilitate deposition of PM (Figure 2(a–c)). On the abaxial surface, epicuticular waxy platelets were found. Waxy platelets were arranged in rosette-like structures on which PM was deposited in a range of 3.244–4.169 µm (Figure 2(d,e)). Moreover, stomata were found on the abaxial surface. These stomata were covered by PM and had a mean opening of 1.719 µm × 904.6 nm (Figure 2(f)).

**M. indica**

The adaxial surface of *M. indica* leaves shows rough morphological variations due to sparse arrangement of the cuticular layer [62]. This roughness permits the deposition of PM of different sizes (929.8 nm to 14.08 µm, Figure 3(a,b)). For instance, fine PM (0.1–2.5 µm) was deposited on the surface (Figure 3(c)). The abaxial surface was found to be rougher, presumably due to the presence of the resin secretor gland, stomata, and epicuticular wax with deposited PM (Figure 3(d,e)). The abaxial surface of *M. indica* has been reported to have a granular form of epicuticular wax surrounding the stomata, a wrinkled cuticular surface, and reticulate-shaped glands, all of which encourage the retention and accumulation of PM [63]. As such, the stomata were able to uptake fine PM with a diameter below 1000 nm (e.g. 931.4 and 840.2 nm) (Figure 3(f)).

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**Figure 2.** SEM of *F. religiosa* leaf surfaces showing PM deposition and accumulation: (a), (b), and (c) adaxial leaf surface; (d), (e), and (f) abaxial leaf surface.
Table 1. Comparative micro-morphological traits of the leaves of six plant species.

| S. No. | Tree species | Family      | Leaf shape           | Epicuticular wax | Stomata       | Trichome |
|--------|--------------|-------------|----------------------|------------------|---------------|----------|
|        |              |             | Adaxial | Abaxial | Adaxial | Abaxial | Adaxial | Abaxial |
| 1      | *F. religiosa* | Moraceae    | Cordate            | Absent  | Present | Absent | Present | Absent | Absent |
| 2      | *M. indica*  | Anacardiaceae | Oval-lanceolate   | Absent  | Present | Absent | Present | Absent | Absent |
| 3      | *B. monosperma* | Fabaceae   | Obovate            | Absent  | Absent  | Absent | Absent  | Present | Present |
| 4      | *A. scholaris* | Apocynaceae | Oblong – lanceolate | Present | Present | Absent | Present | Present | Present |
| 5      | *A. indica*  | Meliaceae   | Lanceolate         | Absent  | Present | Absent | Present | Present | Absent |
| 6      | *S. siamea*  | Fabaceae    | Ovate              | Present | Present | Present | Present | Present | Present |
B. monosperma
Both surfaces of B. monosperma leaves were covered at high density with large trichomes with similar surface morphology types (513.4 and 355.5 µm) (Figure 4(a,c)). This morphology facilitated deposition of large particles up to 173.4 × 55.76 µm (Figure 4(b)) on the adaxial surface and below 17.32 µm on the abaxial surface (Figure 4(d)). The stomata were not seen clearly in the SEM images as they may have been covered due to the excessive density of the trichomes and deposited PM.

A. scholaris
The leaves of A. scholaris have projection-like structures on both surfaces near the milk secretor glands with epicuticular wax (Apocynaceae). The presence of trichomes and appendages facilitated high deposition and accumulation of PM in the fine (PM$_{2.5}$) to respirable (PM$_{10}$) range on the adaxial surface (1.41–4.40 µm) (Figure 5(a,b)). However, the significant deposition of PM damaged the leaf surface, leading to accumulation of PM inside the leaf (Figure 5(c)). In addition, trichomes were found in higher density on the abaxial surface than on the adaxial surface (Figure 5(d,e)), and fine PM deposits (0.54 µm to 3.21 µm) were observed on the stomata and rough surface of the groove in the mid-rib portion of the leaf (Figure 5(f)). The waxy layer covering the surface of the gland facilitated entrapment of PM (Figure 5(g,h)). The deposited PM aggregated and accumulated on the abaxial surface, thus forming a deposition layer over the surface (Figure 5(i)).

A. indica
The leaves of A. indica also showed different micro-morphological characteristics when compared to other plants. On the adaxial surface, trichomes were present only on the midrib portion (Figure 6(a,b)). The morphological features of the trichome surface facilitated the accumulation of PM on the surface (Figure 6(c)). The presence of stomata

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**Figure 3.** SEM of M. indica leaf surfaces showing PM deposition and accumulation. (a), (b), and (c) adaxial surface; (d) and (e) abaxial surface. (f) PM accumulation inside stomata on the abaxial surface.
and the raised surface of the epidermal cell wall make the abaxial surface rougher; this surface showed more PM deposition than the adaxial surface (Figure 6(d,e)). Moreover, the abaxial surface had stomata that facilitated PM accumulation through stomatal pores (Figure 6(e)). Granulated epicuticular wax and the raised epidermal cell wall structure also have been shown to enable PM accumulation on both surfaces [64].

*S. siamea*
On the adaxial surface of *S. siamea* leaves, dust-loaded epidermis, irregular patches of an epicuticular waxy layer, and dense trichomes were observed (Figure 7(a,b)). Grooves were found at the surface of the mid-rib area; these grooves were covered with a wax lining that facilitates the deposition of small PM (Figure 7(c)). The mean length of the observed trichomes was 60.76 µm, with deposition of PM with sizes up to the fine range (712.2 nm to 10.32 µm) (Figure 7(d)). Moreover, PM was deposited in and around stomatal openings on the adaxial surfaces (Figure 7(e)). Hence, PM-bound toxic metals can potentially accumulate in stomata on the leaf surface [20]. Similarly, the abaxial surfaces showed high trichome density, epicuticular wax, and stomata that facilitate deposition and increase the efficiency of PM accumulation (Figure 7(f–h)). Fine PM (e.g. 152.8 nm to 714.5 nm) was deposited over the stomata and readily accumulated in the internal stomatal pores (7.823 × 2.127 µm) (Figure 7(i,j)).

The concentration levels of target toxic metals varied considerably across the different sample types such as foliar dust, road dust, and leaves. Comparison of the data with those
of previous studies indicated that the samples were significantly enriched with the target metals. The lack of correlation between leaf vs. soil samples indicates that metals can accumulate in leaves from foliar dust or road dust resuspension, suggesting the

Figure 5. SEM of *A. scholaris* leaf surfaces showing PM deposition and accumulation. (a), (b), and (c) adaxial surface; (d), (e), (f), (g), (h), and (i) abaxial surface.

Figure 6. SEM of *A. indica* leaf surfaces showing PM deposition and accumulation. (a), (b), and (c) adaxial surface; (d), (e), and (f) abaxial surface.
significance of metals of airborne origin [20]. Likewise, the EF results demonstrated that the Pb, Cd, and Cu in the foliar dust, leaf, and road dust samples arose from anthropogenic activities. The EFs of the various samples can be ranked as follows: (leaf > foliar dust > road dust > soil). Thus, the metals in the leaves came mainly from airborne sources (e.g. traffic sources and other urban sources) and not from soil.

Leaf surface roughness is determined by grooves and bulges of arranged epidermis cells, the presence of epicuticular wax, secretor glands, veins, and number/density of stomata and trichomes on both surfaces [65,31,18]. *S. siamea* showed more diverse leaf surface micro-morphological features (epicuticular wax, stomata, and trichomes) followed by *A. scholaris*, *A. indica*, *M. indica*, *F. religiosa*, and *B. monosperma*, respectively (Figure 7(a–j), Table 1). *F. religiosa* and *M. indica* showed the similar characters, i.e. presence of epicuticular wax and stomata on abaxial surfaces. Whereas, *B. monosperma* has only trichomes on both surfaces with high density. In addition, leaf surfaces of *M. indica* did not show the

Figure 7. SEM of *S. siamea* leaf surfaces showing PM deposition and accumulation. (a), (b), (c), (d), and (e) adaxial surface; (f), (g), (h), (i), and (j) abaxial surface.
presence of trichomes as compared to\textit{ A. indica}. However, due to the more numbers and large opening of stomata, they show more accumulation of metals [66]. It has been previously observed that in comparison to the density and size of the stomata, the pattern and dynamics of the stomatal opening is rather more important factor for foliar PM accumulation [67]. This suggests that the high complexity in external morphology is helpful in the greater deposition of PM (Figure 4). On the other hand, the foliar accumulation of the metals from PM depends on a number of factors. For instance, the dynamics of stomatal opening and its arrangement is the most important factor in accumulation of metals. In addition, external supporting morphology is also helpful. The deposited PM entered directly into the stomata according to its ultra fine size ranges and accumulated inside (e.g. Figure 3 and 7). Moreover, it can also convert toxic metals in ionic state that enters into the leaves through the aqueous pores which are present on cuticular edge of stomatal guard cells, and with the help of the epidermal cuticular anticlinal wall [68,69].

Conclusions

Based on the results of concentration levels of target metals in different plant and environmental samples, EFs, and correlation analyses in this study, it was evident that the target metals bound with PM have anthropogenic source signatures. Moreover, there was a line of confirmation that these metals were partitioned through airborne route which is thought to be less prevalent than their accumulation from soil via roots of the plants. The results of this study further revealed that leaves of different plants have unique micro-morphological traits that facilitate the retention and accumulation of PM in different size ranges. Based on these micro-morphological features of the leaves and the accumulation potential of PM (or PM-bound toxic metals), the six plants are ranked as follows: \textit{S. siamea} > \textit{A. scholaris} > \textit{M. indica} > \textit{F. religiosa} > \textit{A. indica} > \textit{B. monosperma}. Hence, it was found that the particle type, particle diameter class, and assessment scale should all be considered in order to accurately evaluate the comprehensive particle retention abilities of an urban tree species. Based on our findings, these plants can be used for phytomonitoring and management of PM (PM-bound toxic metals) in urban environments. The study also recommends more research covering diverse sampling locations and plant species with variable source characteristics and environmental conditions. Before choosing plant species for urban forestry and greening programmes, it is also important to consider socioeconomic and aesthetic factors.

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