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Urban-gradient variation in soil Cd and its uptake, translocation, and organ allocation in Ulmus *pumila L.*

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Abstract

Cadmium (Cd) contamination in arid urban and peri-urban environments may be influenced by both soil-derived inputs and atmospheric particulate deposition. This study evaluated the spatial distribution of soil Cd and its accumulation in different organs of Ulmus pumila L. across the urban center, suburban area, and outer suburban area of Toksun, Xinjiang, China. Soil and plant samples were collected from the three zones and analyzed for Cd concentrations, organ allocation, and accumulation indices. Soil Cd varied considerably among sites, but differences among the three urban zones were not statistically significant (Kruskal–Wallis test, $H = 1.987$, $p = 0.3702$). The relationship between soil Cd and distance from the urban center was weakly positive ($r = 0.287$, $p = 0.0103$), whereas a significant negative relationship occurred within the urban-center subgroup ($r = -0.347$, $p = 0.0011$). This contrast indicates that Cd distribution was driven by localized sources rather than a single uniform urban–suburban gradient. Cd accumulation was strongly organ-specific: bark had substantially higher Cd concentrations and the largest share of accumulated Cd, with bioaccumulation factors generally above 1, whereas root, branch, and leaf accumulation was limited. The results suggest that bark integrates internal Cd transport with retention of atmospheric particles and may serve as a biomonitoring tissue for Cd exposure in arid

urban environments. Further studies using bioavailable soil Cd and washed and unwashed bark fractions are needed to separate soil uptake from atmospheric deposition.

Key words: cadmium; *Ulmus pumila L.*; soil contamination; organ-specific accumulation; bark; bioaccumulation factor; urban – peri-urban gradient; Toksun; Xinjiang

1 Introduction

Cadmium (Cd) is a non-essential and highly toxic trace metal that poses a persistent threat to terrestrial ecosystems and human health. Unlike essential micronutrients, Cd has no known biological function in plants, yet it can be readily taken up from soil and transported to metabolically active tissues. Chronic exposure to Cd is associated with renal dysfunction, skeletal damage, reproductive toxicity, and carcinogenic effects in humans (Tchounwou et al., 2012; Jaishankar et al., 2014). Because Cd is relatively mobile in many soils and may remain environmentally available for decades, its accumulation in urban and peri-urban environments has become an important concern in environmental risk assessment and land-management planning (Alloway, 2013; Kabata-Pendias, 2011).

Urbanization can modify the spatial distribution, chemical speciation, and biological availability of Cd in soils. Atmospheric deposition, vehicle emissions, coal and industrial combustion, wastewater irrigation, sewage-derived materials, construction activities, and the long-term application of phosphate fertilizers are among the major anthropogenic sources of Cd in urban and peri-urban soils (Manta et al., 2002; Imperato et al., 2003; Charlesworth et al., 2003; Wuana and Okieimen, 2011). In addition, urban development changes soil properties, including pH, organic matter content, cation-exchange capacity, texture, and redox conditions, all of which regulate Cd retention and mobility (McLaughlin et al., 2000; Smolders and Mertens, 2013). Consequently, soil Cd concentrations may exhibit strong spatial heterogeneity along urban–suburban gradients rather than following a simple monotonic decline with distance from the city

center. Previous studies have documented elevated metal concentrations in urban soils and substantial differences among land-use types, cities, and surrounding rural areas (Madrid et al., 2002; Wei and Yang, 2010; Lu et al., 2012). However, the spatial pattern of Cd along an urbanization gradient remains insufficiently characterized in arid and semi-arid regions, where low precipitation, limited vegetation cover, strong atmospheric dust transport, and intense human disturbance may jointly affect contaminant redistribution.

Soil contamination does not necessarily translate directly into plant contamination. The transfer of Cd from soil to plants is controlled by a combination of total soil concentration, soil physicochemical properties, Cd bioavailability, root uptake capacity, and plant-specific regulatory mechanisms (McLaughlin et al., 2000; Peralta-Videa et al., 2009; Rizwan et al., 2016). At the root–soil interface, Cd may enter plant cells through transport systems originally involved in the uptake of essential elements such as zinc, iron, calcium, and manganese. Once absorbed, Cd can be chelated by thiol-rich compounds, sequestered in vacuoles, or transported through the xylem and phloem to aboveground organs (Clemens, 2006; Uraguchi and Fujiwara, 2012; Clemens et al., 2013). Plant responses to Cd include restriction of root-to-shoot transport, intracellular immobilization, antioxidant activation, and changes in nutrient homeostasis (Seregin and Ivanov, 2001; Yadav, 2010; Rascio and Navari-Izzo, 2011). These processes may produce marked differences in Cd concentration, bioconcentration factor (BCF), translocation factor (TF), and bioaccumulation factor (BAF) among roots, bark, branches, and leaves. Therefore, measuring only total plant Cd concentration is insufficient for understanding the mechanisms of Cd uptake, internal transport, and organ-level allocation.

Trees are particularly useful for investigating soil–plant metal transfer because of their extensive root systems, long lifespan, large biomass, and ability to integrate long-term environmental exposure. Woody species may also retain substantial amounts of metals in perennial tissues such as bark and branches, thereby influencing the apparent translocation and allocation patterns of contaminants within the plant (Pulford and Watson, 2003; Vamerali et al., 2009; Dickinson et al., 2009). Bark is additionally exposed to atmospheric particles and stem deposition, and its high surface area and structural composition may make it an important compartment for metal retention. The distribution of Cd among tree organs may therefore reflect not only root uptake from soil but also internal redistribution and organ-specific storage. Such

organ-specific patterns are particularly relevant for evaluating the suitability of trees as biomonitors and for identifying the tissues most sensitive to urban metal exposure (Ali et al., 2013; Haider et al., 2021). Nevertheless, compared with agricultural crops and hyperaccumulator species, relatively limited attention has been given to Cd transfer and allocation among the organs of widely planted urban trees.

Ulmus pumila L. is a drought-tolerant and ecologically important tree species that is widely distributed in northern China and commonly used in shelterbelts, urban greening, and windbreak systems. Its tolerance of water limitation and poor soils makes it particularly important in the arid and semi-arid landscapes of Xinjiang. The species is also suitable for assessing spatially variable contaminant exposure because its perennial organs can record cumulative environmental conditions, while its leaves and young tissues may respond more rapidly to recent exposure. However, the extent to which urbanization affects soil Cd concentration and its subsequent uptake, translocation, and allocation in U. pumila remains unclear. In particular, it is not known whether higher soil Cd concentrations near urban areas are accompanied by enhanced root uptake, greater transfer to aboveground tissues, or preferential retention in specific organs such as bark.

Toksun County, Xinjiang, provides an appropriate natural setting for examining these questions. The area contains a clear urban–suburban gradient and is characterized by an arid continental climate, strong dust activity, limited precipitation, and substantial spatial differences in human disturbance. These conditions may enhance the accumulation and redistribution of particulate-bound Cd while simultaneously modifying its mobility in soil. A gradient-based analysis in this region can therefore provide insight into how urbanization influences Cd behavior under environmental conditions that differ from those in humid and densely industrialized regions.

In this study, we investigated the variation in soil Cd and its uptake, translocation, and organ allocation in *U. pumila* along an urban–suburban gradient in Toksun County, Xinjiang, China. Specifically, we aimed to: (1) quantify the differences in soil Cd concentration among the urban center, suburban area, and outer suburban area; (2) characterize the spatial distribution and pollution classification of soil Cd; (3) determine the organ-specific and spatial variation in Cd concentrations in U. pumila; (4) evaluate root uptake, root-to-shoot translocation, and relative Cd allocation using BCFs, TFs, and organ allocation ratios; and (5) compare organ- and zone-specific BAFs and identify the tissues in which BAF exceeded 1. We hypothesized that: (i) soil Cd

concentrations would decrease with increasing distance from the urban center; (ii) Cd accumulation would differ significantly among plant organs because of contrasting transport and storage functions; and (iii) bark would show greater Cd accumulation and a higher frequency of BAF > 1 than leaves, branches, and roots. By integrating spatial soil assessment with organ-level plant analysis, this study seeks to clarify the mechanisms governing Cd transfer within an urban tree system and to provide a scientific basis for urban ecological risk assessment and biomonitoring in arid regions.

2 Materials and methods

The study was conducted in Toksun County, Xinjiang, northwestern China, along an urban–suburban gradient extending from the urban center through the suburban area to the outer suburban area. The location of the study area and the distribution of the sampling sites along this gradient are shown in Fig. 1.

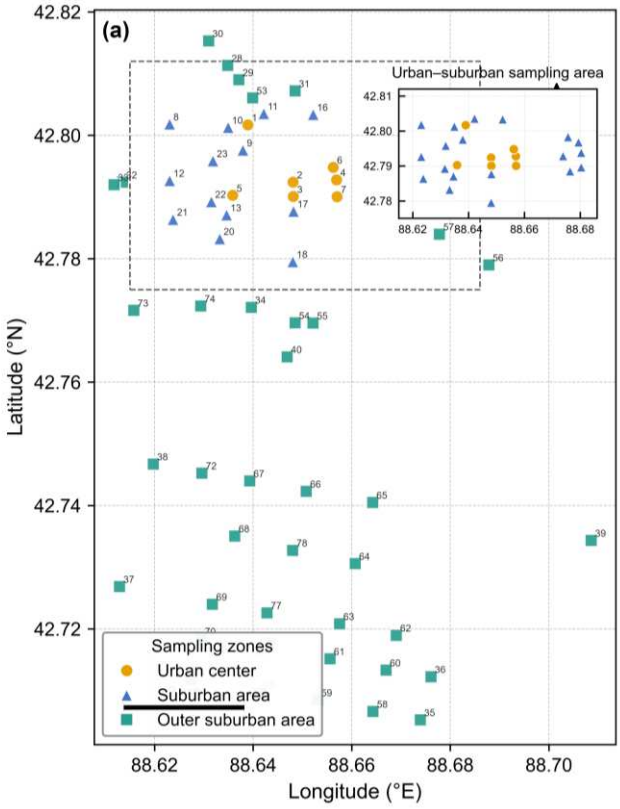


Fig. 1. Study area and distribution of sampling sites along the urban–suburban gradient in Toksun County, Xinjiang, China. Different symbols and colors represent the urban center, suburban area, and outer suburban area, respectively. Numbers denote sampling-site IDs.

2.1. Soil and plant sampling and preparation

Field sampling was conducted in July 2025 during comparable growing-season periods. At each site, surface soil and plant tissues were collected from the same Ulmus pumila L. individual. After removing litter and visible debris, five soil subsamples were collected from the 0–20 cm layer using a five-point sampling design and combined into one composite sample. Soil samples were air-dried, cleaned of stones and plant residues, ground, and passed through a 0.149-mm sieve.

Roots, bark, branches, and leaves were collected from the corresponding tree. Roots were carefully separated from adhering soil. Plant tissues were washed with deionized and distilled water, oven-dried at 65–75 °C to constant weight, ground, passed through a 0.5-mm sieve, and stored in sealed containers until analysis.

Because plant tissues were washed before analysis, the measured concentrations mainly represent tissue-associated PTEs after removal of loosely attached particles. However, strongly retained atmospheric particles, particularly on bark, may not have been completely removed.

2.2. Determination of soil and plant PTE concentrations

Soil and plant samples were analyzed according to relevant Chinese national or sectoral standards. Soil Cd, Cr, Pb, and Zn were determined by ICP-MS (NexION 1000G, PerkinElmer, USA) after microwave digestion following HJ 832–2017. Soil total As and Hg were determined by atomic fluorescence spectrometry (AFS-3000, Beijing Haiguang Instrument Co., China) according to GB/T 22105.2–2008 and GB/T 22105.1–2008, respectively.

Plant samples were digested according to GB 5009.268–2016. Cr, Cd, Pb, Zn, and As were measured by ICP-MS, whereas Hg was determined by atomic fluorescence spectrometry

(AFS-933, Beijing Haiguang Instrument Co., China). The method detection limits for soil and plant elements are listed in Table 1.

Rhodium was used as the internal standard, with recoveries ranging from 86.1% to 106.1%. Reagent blanks, duplicate samples, and quality-control materials were included in each analytical batch. All concentrations are reported on a dry-weight basis.

Table 1. Analytical methods, instruments, and method detection limits for PTEs in soil and plant samples.

Matrix	Element	Standard method	Instrument	MDL (mg kg^{-1})
Soil	Cd	HJ 832–2017	ICP-MS (NexION 1000G, PerkinElmer)	0.07
Plant	Cd	GB 5009.268–2016	ICP-MS (NexION 1000G, PerkinElmer)	0.002

2.3. Soil contamination screening and potential ecological risk assessment

Because soil pH values were above 7.5, the pH-specific screening values for agricultural soils in GB 15618–2018 were used as comparative reference values for Zn, Cd, Hg, As, and Cr. As the study area included peri-urban and industrially influenced land, these values were used for conservative screening rather than direct regulatory assessment of non-agricultural land.

The single-factor pollution index was calculated as:

$$P_i = \frac{S_i}{C_i}$$

where $C_i$is the measured concentration of element ii in a soil sample and S_i is the corresponding pH-specific screening value in GB 15618–2018. A value of P_i> 1 indicates that the concentration exceeded the applicable screening reference value.

The Nemerow comprehensive pollution index was calculated at the site level as:

$$P_N = \sqrt{\frac{2}{P^2{}_{i,max} + P^2{}_{i,avg}}}$$

where $P_{i,max}$ is the maximum single-factor pollution index among the five investigated PTEs

at a given site, and $P_{i,avg}$ is the arithmetic mean of the corresponding single-factor indices. Site-level P_i and $P_N$values were subsequently summarized and compared between survey years.

Potential ecological risk was assessed using the approach proposed by Håkanson (1980). Xinjiang soil environmental background values were used as reference concentrations (Håkanson 1980). The contamination factor (f_i^C), individual potential ecological risk factor (r_i^E), and integrated potential ecological risk index (RI) were calculated as follows:

$$f_i^C = \frac{n_i^C}{C_i}$$

$$r_i^E = r_i^T \times f_i^C$$

$$RI = \sum_{r_i^E}^{i=1} n$$

where C_i is the measured concentration of element i, n_i^C is its corresponding Xinjiang background concentration, r_i^T is the toxic-response factor, and nn is the number of investigated elements. The toxic-response factors for Zn, Cr, Cd, Hg, and As were 1, 2, 30, 40, and 10, respectively [23]. The classification criteria for P_i, P_N, r_i^E, and RI, as well as the screening and background values used in this study, are summarized in Table S1.

2.4. Tissue-specific accumulation and translocation indices

The soil-to-tree accumulation potential of Zn, Cd, Hg, As, and Cr in *Ulmus pumila L.* was assessed using the bioconcentration factor (BCF):

$$BCF_{tissue} = \frac{C_{soil}}{C_{tissue}}$$

where C_{tissue} is the concentration of a given PTE in roots, bark, branches, or leaves, and C_{soil} is the concentration of the same element in the corresponding soil sample collected beneath the same tree. BCFs were calculated separately for each tissue, element, and sampling site.

Internal translocation was assessed using tissue-to-root translocation factors (TFs):

$$TF_{root-bark} = \frac{C_{root}}{C_{bark}}$$

$$TF_{root-branch} = \frac{C_{root}}{C_{branch}}$$

$$TF_{root-leaf} = \frac{C_{root}}{C_{leaf}}$$

where C_{bark}, C_{branch}, C_{leaf}, and C_{root} denote PTE concentrations in the corresponding tissues. A BCF > 1 indicates that the concentration in a given tissue was higher than that in the associated soil, whereas a TF > 1 indicates that the concentration in the receiving aboveground tissue exceeded that in roots.

2.5. Analysis of five-year changes in soil PTE concentrations and risks

The same sites were surveyed in 2020 and 2025, allowing paired comparisons of soil Zn, Cr, Cd, Hg, and As concentrations. Missing, non-numeric, and negative values were excluded. The relative change was calculated as:

$$\triangle C(\%) = \frac{Median_{2020}}{Median_{2025} - Median_{2020}} \times 100$$

Uncertainty in the estimated median change was evaluated using nonparametric paired bootstrap resampling. The 2.5th and 97.5th percentiles of the bootstrap distribution were used as the bootstrap 95% confidence interval (CI).

The same paired analytical framework was used to compare site-level P_N, r_i^E, and *RI* values between years, where applicable. Statistical significance was set at $p < 0.05$. Forest plots were used to display median percentage changes and their bootstrap 95% CIs.

2.6. Spatial interpolation and distance-gradient analysis

Spatial distributions of PTEs in soil, roots, bark, branches, and leaves were visualized using inverse distance weighting interpolation in Python. Coordinates were approximately converted to metres using latitude-specific conversion factors. IDW interpolation was performed with a power parameter of 2 on a 250 × 250 grid. The interpolation was used only to visualize local spatial variation within the sampled area.

For each matrix–element combination, interpolation surfaces were generated for Cd, with sampling sites overlaid on the maps. Because the analysis did not account for anisotropy, wind direction, terrain, land use, external validation, or source apportionment, the interpolated surfaces were not interpreted as precise contamination boundaries or direct evidence of emission sources.

Predictions near the edges of the study area and far from sampling sites were interpreted cautiously.

The distance from each site to the industrial core was calculated in metres. Sites were classified into near (<1.5 km), middle (1.5–3.0 km), and far (≥3.0 km) groups. Spearman rank correlation was used to evaluate associations between PTE concentrations and distance, and the Kruskal–Wallis test was used to compare distance groups. When significant, pairwise comparisons were conducted using Dunn's test with Holm correction. Distance-gradient analyses were performed separately for soil, roots, bark, branches, and leaves in the 2025 survey.

These analyses were interpreted as spatial associations with industrial proximity rather than definitive evidence of industrial-source contributions, because other factors, including wind, traffic, land use, terrain, and soil properties, may also affect PTE distributions.

2.7. Statistical analysis and data visualization

Statistical analyses and visualizations were conducted using Python 3.12 with the pandas, numpy, scipy, and Matplotlib packages. R and RStudio were also used where applicable. All tests were two-sided, with significance defined as $p < 0.05$. Wilcoxon signed-rank tests and paired bootstrap resampling were used for comparisons between 2020 and 2025. Spearman correlations were used for distance-gradient relationships, and Kruskal–Wallis tests followed by Dunn's test with Holm correction were used for comparisons among distance groups. Maps were generated using Python-based IDW interpolation, and figures were exported at 300 dpi. All PTE concentrations are expressed as mg kg^{-1} on a dry-weight basis.

3 Results

3.1 Soil Cd concentration in different urban zones

Soil Cd concentrations showed a significant spatial variation along the urban-suburban gradient (Fig. 2). Across 48 soil sampling sites, the mean Cd concentration was 0.1856 ± 0.0542 mg/kg, with a range of 0.0987–0.3124 mg/kg.The large coefficient of variation (CV = 29.2%) indicated marked variability in soil Cd distribution.

Soil Cd concentrations differed numerically among the three urban zones, but the overall

difference was not statistically significant according to the Kruskal–Wallis test (H = 1.987, p = 0.3702). (Figure 2a). In the urban center, soil Cd concentration ranged from 0.144 to 0.291 mg/kg with a mean of 0.2121 ± 0.0526 mg/kg (mean ± SD). In the suburban area, soil Cd levels ranged from 0.093 to 0.291 mg/kg with a mean of0.1877 ± 0.0585 mg/kg . The outer suburban area showed soil Cd concentrations ranging from 0.093 to 0.31 mg/kg with a mean of 0.2059 ± 0.0675mg/kg .

The Kruskal-Wallis test did not reveal a statistically significant difference in soil Cd concentrations among the three urban zones (H = 1.987, p = 0.3702). The mean soil Cd concentrations in all three zones exceeded the Xinjiang regional background level of 0.1 mg kg^{-1} , although some individual samples were below this value.

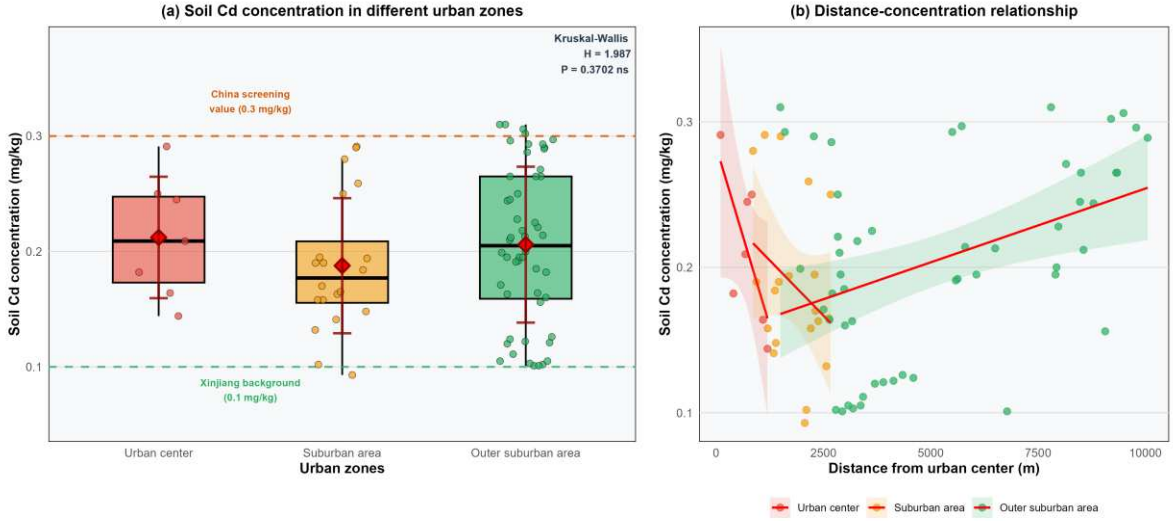


Figure 2. Soil Cd concentration in different urban zones and its relationship with distance from the urban center. (a) Box plot showing soil Cd concentration across three urban zones. The box represents the interquartile range (IQR), the line inside the box indicates the median, individual points represent individual samples (n = 79), red diamonds represent the mean values with error bars indicating ±SD (standard deviation). Dashed lines indicate the China screening value (0.3 mg/kg) and the Xinjiang background level (0.1 mg/kg). Letters above the boxes denote significant differences ($p < 0.05$, Dunn's post hoc test). (b) Scatter plot showing the relationship between distance from urban center and soil Cd concentration. Each point represents a soil sample colored by zone. The red regression line represents the linear relationship across all samples.

A weak negative correlation was observed between distance from the urban center and soil

Cd concentration (Figure 2b, r = 0.287, p = 0.0103, r^2 = 0.0825). However, the R^2 value (0.0825) indicates that distance alone explains only 8.2% of the variance in soil Cd concentration. This suggests that while urban centers tend to have slightly higher Cd levels, other factors such as local industrial activities, traffic emissions, and historical land use patterns play important roles in determining soil Cd levels.

3.1.2 Spatial distribution of soil Cd

Spatial interpolation using linear and cubic spline methods revealed pronounced spatial heterogeneity in soil Cd concentrations across the study area (Figure 3). Continuous Cd concentration surfaces were generated from the discrete sampling data using three related approaches. Linear interpolation produced piecewise planar surfaces between sampling points (Figure 3b). Although computationally efficient and capable of preserving local variation, this method generated visible faceting effects and abrupt changes in slope at the boundaries between adjacent interpolation areas.

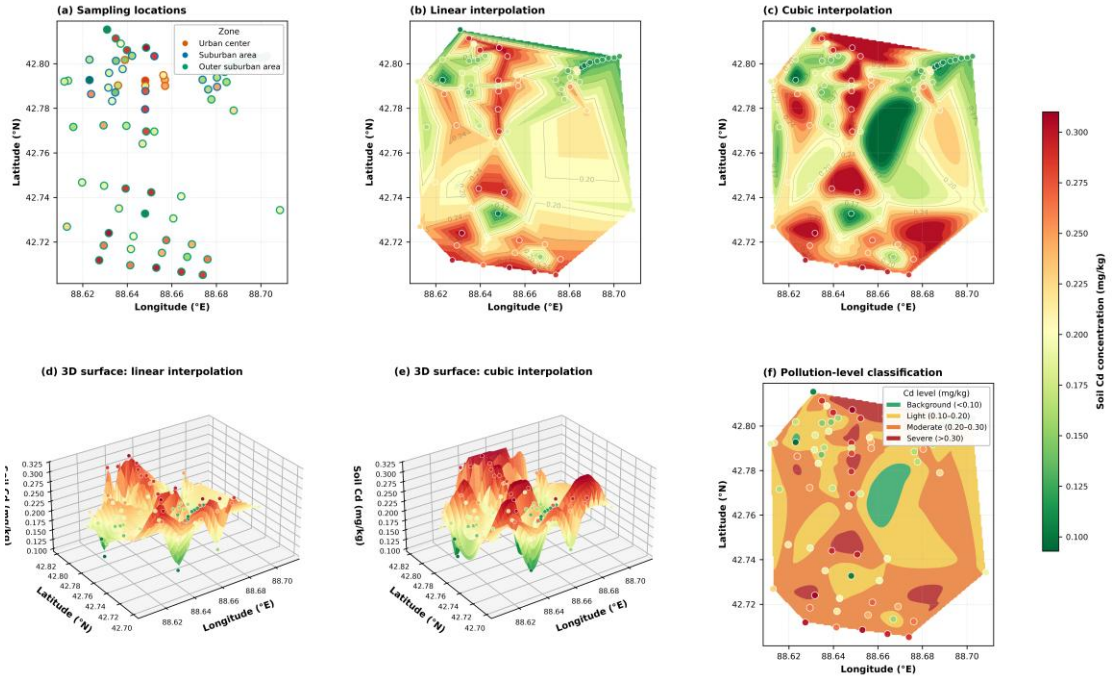


Figure 3. Spatial distribution and interpolated pollution classification of soil Cd. (a) Sampling locations and measured soil Cd concentrations. (b) Spatial surface generated using linear interpolation. (c) Spatial surface generated using cubic interpolation. (d) Three-dimensional surface based on linear interpolation. (e) Three-dimensional surface based on cubic interpolation. (f) Interpolated soil Cd pollution-level classification based

on the predefined concentration thresholds. Continuous colour scales represent measured or interpolated soil Cd concentrations, whereas the discrete colours in panel (f) represent pollution classes.The pollution-level categories were defined as background level (<0.1 mg/kg), light pollution (0.1-0.2 mg/kg), moderate pollution (0.2-0.3 mg/kg), and severe pollution (>0.3 mg/kg), according to the classification thresholds selected in this study.

Cubic spline interpolation generated smoother concentration gradients by fitting third-order polynomial functions between sampling points (Figure 3c). Compared with linear interpolation, the cubic spline method provided improved visual continuity and better represented the regional spatial pattern. To further reduce local fluctuations caused by uneven sampling density, Gaussian smoothing was applied to the cubic spline surface using a standard deviation of 1.5. The resulting smoothed surface was used to improve the visualization of regional trends and potential concentration hotspots. However, the smoothed surface should be interpreted as a visualization of the spatial pattern rather than as an independent increase in analytical accuracy.

The three-dimensional surface plots further illustrate the spatial variation in soil Cd concentrations (Figures 3d and 3e). The interpolated surfaces show several concentration peaks in the urban and adjacent suburban areas, followed by a general decrease toward the outer suburban region. These results indicate that Cd concentrations were spatially heterogeneous and that high-value areas were mainly concentrated within or near the urbanized part of the study area.

3.1.3 Spatial distribution patterns

The smoothed cubic spline interpolation map (Figure 3c) revealed several distinct spatial features. The highest interpolated Cd concentrations, generally exceeding 0.25 mg/kg, were mainly concentrated in the northwestern part of the study area, approximately between 88.62 and 88.65°E and 42.80 and 42.82°N. This area corresponded to the urban center and adjacent transitional suburban regions and represented the primary high-concentration cluster identified in the interpolated surface. The highest measured Cd concentration among the sampling sites was 0.31 mg/kg.

Moderate concentration zones, with interpolated values of approximately 0.20-0.25 mg/kg, extended southward and eastward from the main high-concentration area into the suburban region. This pattern suggests a gradual decrease in Cd concentrations with increasing distance from the

urban center. The observed spatial gradient may reflect the combined effects of urban activities, atmospheric deposition, traffic-related emissions, and other diffuse sources. Nevertheless, the spatial distribution alone cannot identify the specific pollution sources. In contrast, the eastern and southern margins of the outer suburban region generally exhibited lower Cd concentrations, mostly below 0.15 mg/kg. These areas were closer to the reported Xinjiang regional background value of 0.1 mg/kg and showed comparatively limited spatial influence from the urban area. A distance-decay trend was also observed between the urban center and the surrounding areas. The correlation between distance from the urban center and soil Cd concentration was negative and statistically significant ($r = -0.347$, $p = 0.0011$), indicating that Cd concentrations tended to decrease with increasing distance from the urban center. However, the relatively moderate correlation coefficient also suggests considerable local spatial variation.

3.1.4 Pollution-level classification

Based on the selected reference thresholds, the interpolated Cd concentrations were classified into four concentration categories in Figure 3f. These categories were defined as follows: background level (<0.1 mg/kg), light pollution (0.1-0.2 mg/kg), moderate pollution (0.2-0.3 mg/kg), and severe pollution (>0.3 mg/kg). Areas with concentrations below 0.1 mg/kg were classified as background-level areas and were mainly distributed along the outer suburban margins and the eastern periphery of the study area. These areas were generally consistent with the regional background concentration and showed relatively limited apparent anthropogenic influence. However, the term “background level” should be understood as a concentration-based classification and should not automatically be interpreted as “safe for all land uses” without reference to the applicable soil environmental quality standard and land-use type. The light-pollution category, corresponding to concentrations of 0.1-0.2 mg/kg, was distributed across parts of the suburban and outer suburban areas. These areas indicated concentrations above the selected background value but below the moderate-pollution threshold, suggesting a relatively low to moderate degree of anthropogenic influence. Moderate-pollution areas, with concentrations of 0.2-0.3 mg/kg, were mainly distributed around the urban center and in parts of the suburban region. These areas represented relatively high Cd concentrations and should be considered priority areas for continued monitoring and further investigation. A limited number of samples exceeded the

selected screening threshold, indicating localized Cd enrichment rather than widespread severe contamination. This threshold indicates that the measured or interpolated concentration exceeded the selected Cd screening value of 0.3 mg/kg for the specified land-use scenario. The severe-pollution classification should therefore be interpreted strictly in relation to the applicable standard, soil pH, land-use category, sampling depth, and regulatory framework. It should not be generalized to all land-use types without additional supporting information.

Overall, the classification map showed that the higher-concentration zones were mainly concentrated in the urban and adjacent suburban areas, whereas lower-concentration zones were more common toward the outer suburban margins. The spatial pattern was consistent with the negative relationship between distance from the urban center and soil Cd concentration.

3.2 Cd accumulation in *Ulmus pumila L.* organs

Cd concentrations showed clear organ-specific variation in U. pumila (Fig. 4A). Bark exhibited the highest mean Cd concentration and the greatest variation among the examined organs. In comparison, Cd concentrations in roots, branches, and leaves were considerably lower and showed relatively narrow distributions. These results indicate that Cd was preferentially retained in bark tissue rather than uniformly distributed among plant organs.

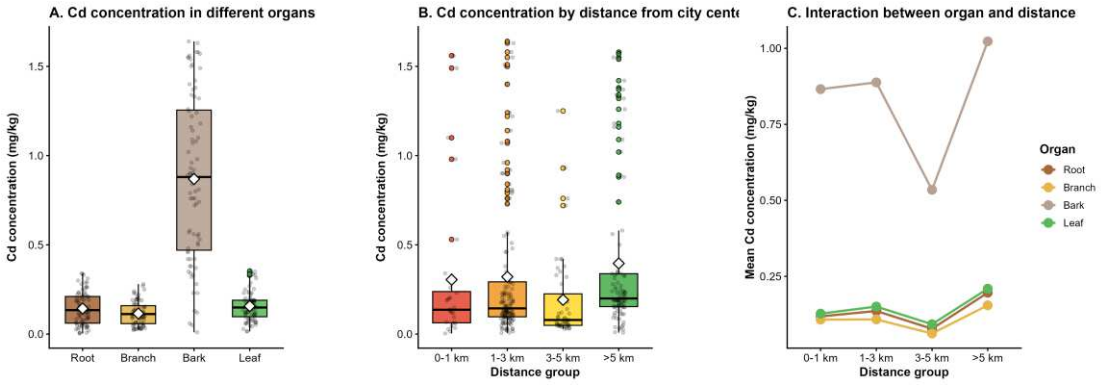


Figure 4. Organ-specific and spatial variation in Cd concentrations in *Ulmus pumila L.* (A) Cd concentrations in different plant organs. (B) Cd concentrations across distance groups from the city center. (C) Interaction between plant organ and distance group. Boxplots show the median, interquartile range, and data distribution; individual points represent observations, and diamonds indicate mean values. The lines in panel C represent mean Cd concentrations for each organ across the distance groups.

Cd concentrations also varied across the distance gradient from the city center (Fig. 4B). The highest overall Cd concentrations were observed in the samples collected more than 5 km from the city center, whereas the 3–5 km group generally showed lower concentrations. The 0–1 km and 1–3 km groups displayed intermediate Cd levels, although substantial variation and several high-value observations were present within these groups.

The organ-by-distance interaction pattern further demonstrated that the spatial variation in Cd concentration differed among plant organs (Fig. 4C). Bark consistently had much higher mean Cd concentrations than roots, branches, and leaves across all distance groups. Bark Cd concentration decreased in the 3–5 km group and increased markedly in the group located more than 5 km from the city center. In contrast, the other organs showed relatively low Cd concentrations and only modest changes along the distance gradient. Overall, Cd accumulation was primarily characterized by strong organ-specific partitioning, with a comparatively pronounced spatial response in bark tissue.

3.3 Soil-to-plant Cd transfer and allocation pattern

The Cd transfer indices revealed distinct patterns of uptake, internal translocation, and organ-level allocation in Ulmus pumila L. Root BCFs varied among the three urban zones, with the lowest overall values observed in the urban center and generally higher values in the suburban and outer suburban areas (Fig. 5A). The median root BCF increased from approximately 0.4 in the urban center to approximately 0.7–0.8 in the suburban and outer suburban areas, indicating stronger root-level Cd enrichment relative to soil Cd in the less urbanized zones. However, considerable within-zone variation was observed, particularly in the outer suburban area.

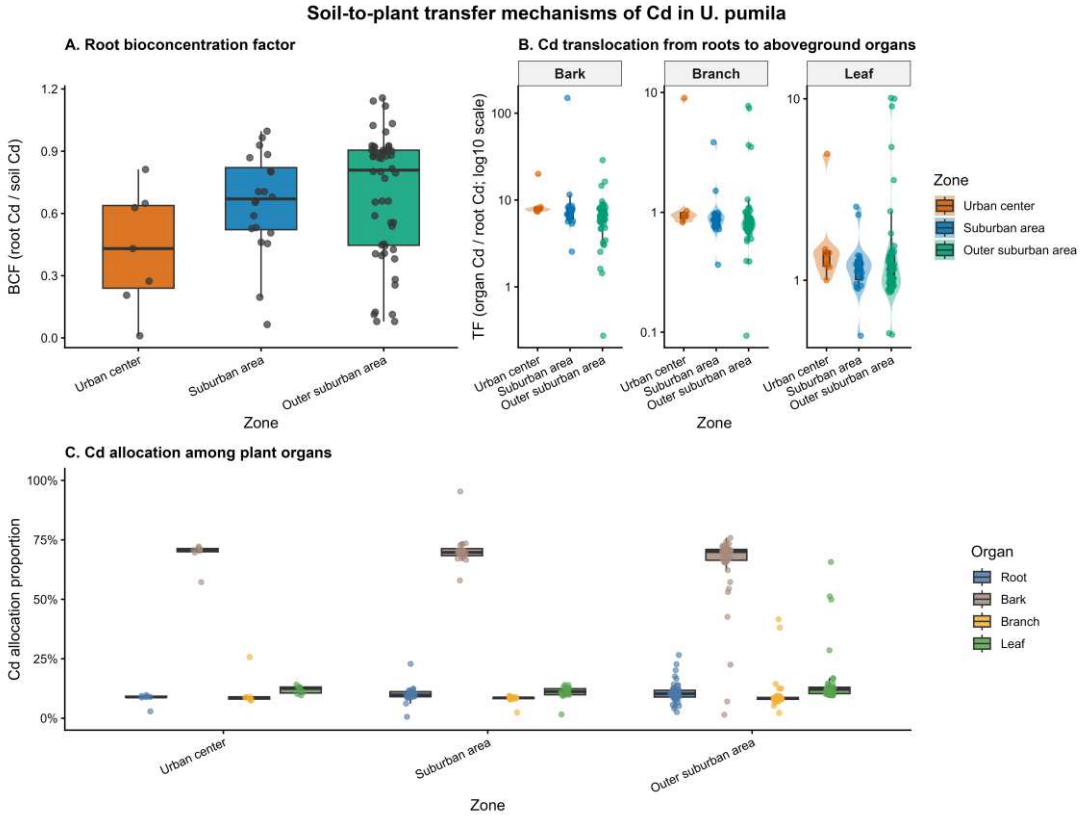


Figure 5. Uptake, translocation, and allocation of Cd in Ulmus pumila across urban zones. (A) Root bioconcentration factors (BCFs), calculated as the ratio of root Cd concentration to soil Cd concentration. (B) Translocation factors (TFs) from roots to bark, branches, and leaves. TF values are shown on a logarithmic scale because of the large difference in transfer magnitude among organs. (C) Relative allocation of Cd among roots, bark, branches, and leaves.

Cd translocation was strongly organ-dependent (Fig. 5B). Bark exhibited substantially higher TF values than branches and leaves across all zones, with most bark TF values exceeding 1 and several observations reaching more than 10-fold relative to root Cd. In contrast, branch and leaf TFs were generally close to or below 1, indicating limited transfer from roots to these organs. The logarithmic presentation also showed that the variation in bark TFs was much larger than that in branch and leaf TFs. A small number of very high bark TF values occurred in the suburban and outer suburban areas, suggesting substantial individual-level heterogeneity in Cd redistribution.

The organ allocation pattern further indicated that Cd was primarily retained in bark (Fig. 5C). Bark accounted for approximately 65–75% of the total Cd represented by the four analyzed organs in most samples, whereas roots, branches, and leaves generally contributed smaller

proportions, each commonly remaining below 15%. This bark-dominated allocation pattern was consistent across the three zones. Although some individual trees showed relatively high Cd allocation to roots or leaves, these observations did not alter the overall pattern of preferential Cd retention in bark.

3.4 Tissue-specific Cd accumulation factors in *Ulmus pumila L.*

The Cd bioaccumulation factor varied substantially among plant organs and urbanization zones (Figure 6). Bark showed the highest BAF values across all zones, with median values of 3.19 in the urban center, 4.70 in the suburban area, and 4.52 in the outer suburban area. In contrast, the median BAF values were relatively low in leaves, branches, and roots, ranging from 0.36 to 0.81 across the three zones.

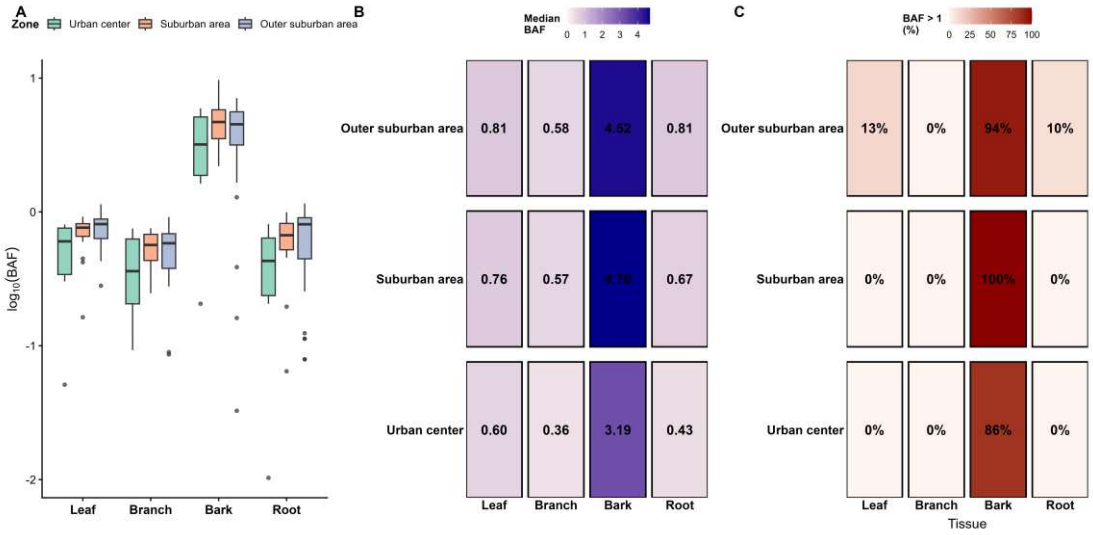


Figure 6. Cadmium bioaccumulation factors (BAFs) in leaves, branches, bark, and roots across three urbanization zones. (A) Boxplots of log10-transformed BAF values. (B) Heatmap showing the median BAF for each zone–tissue combination. (C) Percentage of samples with BAF > 1 in each zone and tissue. BAF was calculated as the Cd concentration in plant tissue divided by the corresponding Cd concentration in soil.

The median BAFs of leaves were 0.60, 0.76, and 0.81 in the urban center, suburban area, and outer suburban area, respectively. The corresponding values for branches were 0.36, 0.57, and 0.58, while those for roots were 0.43, 0.67, and 0.81, respectively. These results indicate that Cd accumulation was particularly pronounced in bark compared with the other plant organs.

The proportion of samples with BAF > 1 was also strongly tissue-dependent. Bark had the

highest proportion of samples exceeding unity, reaching 86% in the urban center, 100% in the suburban area, and 94% in the outer suburban area. The proportions for leaves were 0%, 0%, and 13%, respectively, whereas no samples with BAF > 1 were observed in branches or roots in the urban center and suburban area. In the outer suburban area, the proportion of samples with BAF > 1 was 10% for roots, while it remained 0% for branches.

Overall, bark exhibited the greatest Cd accumulation capacity, whereas leaves, branches, and roots generally showed BAF values below 1. The consistently high bark BAFs across all zones suggest that bark may be a more sensitive plant organ for indicating atmospheric or surface-associated Cd accumulation.

4 Discussion

4.1 Spatial variation of soil Cd among urban zones

Soil Cd concentrations showed clear variation among individual sampling sites, but the differences among the three urban zones were not statistically significant. The mean Cd concentration was slightly higher in the urban center than in the suburban area, while the outer suburban area also contained several relatively high-value samples. However, the Kruskal–Wallis test indicated no significant difference among the urban center, suburban area, and outer suburban area ($H=1.987$, $p=0.3702$)($H=1.987$, $p=0.3702$). Therefore, the present results do not support a simple pattern of progressively decreasing soil Cd concentrations from the urban center toward the outer suburbs.

Although the zone-based differences were not significant, the mean Cd concentrations in the three zones were generally higher than the Xinjiang regional background value of 0.1 mg kg−1. Several samples were also close to or above the Chinese screening value of 0.3 mg kg−1. This pattern suggests that Cd enrichment occurred at some locations, but the contamination was spatially heterogeneous rather than uniformly distributed throughout the study area. The presence of relatively high concentrations in the outer suburban area further indicates that local sources and particle redistribution may be more important than urban-zone classification alone.

Urban soil Cd may originate from a combination of traffic emissions, industrial activities, coal combustion, atmospheric deposition, construction, agricultural inputs, and the redistribution

of contaminated road dust. Previous studies have shown that heavy-metal concentrations in urban soils commonly display localized hotspots associated with traffic corridors, industrial land use, and historical human activities (Han et al., 2006; Caggiano et al., 2015). In arid regions, limited precipitation and frequent wind erosion may increase the resuspension and transport of contaminated particles. As a result, Cd-bearing dust generated in urban or industrial areas may be transported to suburban and outer-suburban locations, weakening a simple distance-related pattern.

The interpolation results also showed several local high-concentration areas rather than one continuous high-Cd zone restricted to the urban center. These localized patterns are consistent with multiple and spatially discontinuous sources. The results of previous studies in northwestern China similarly indicate that trace-element accumulation in urban vegetation, dust, and soil can be influenced by the interaction of local emissions, atmospheric particles, and soil properties (Baidourela et al., 2024a, 2024b). Thus, the spatial distribution observed in the present study is probably controlled by a combination of urban activity, local land use, wind-driven dust transport, and differences in soil properties.

4.2 Distance-dependent variation in soil Cd

The relationship between soil Cd concentration and distance from the urban center differed among the urban zones. When all sampling sites were analyzed together, soil Cd showed a weak but significant positive correlation with distance from the urban center ($r=0.287$, $p=0.0103$)($r=0.287$, $p=0.0103$). This result indicates that Cd concentrations tended to increase slightly with distance when the entire dataset was considered. However, the positive overall correlation should not be interpreted as evidence that increasing distance universally increases soil Cd concentrations.

In contrast, the urban-center subgroup showed a significant negative correlation between Cd concentration and distance ($r=-0.347$, $p=0.0011$). This indicates that, within the urban-center sampling range, Cd concentrations generally decreased as the distance from the city center increased. The negative trend is also consistent with the decline observed in the urban-center fitted line in Figure 2b. This pattern suggests that the most concentrated urban sources may be located near the central urban area or that Cd-containing particles are deposited more strongly near these sources.

The difference between the overall positive relationship and the negative relationship within the urban-center subgroup reflects the influence of spatial heterogeneity and data grouping. In particular, the overall correlation combines samples from the urban center, suburban area, and outer suburban area, whereas the subgroup correlation describes only the short-distance urban-center gradient. The green fitted line for the outer suburban area shows an opposite tendency, with Cd concentrations increasing at greater distances in some parts of the sampling range. This pattern may result from local sources, transported dust, agricultural activities, or the presence of isolated high-Cd sites. Therefore, the combined correlation is likely affected by differences among zones and should not be used to describe the behavior of any single zone.

The non-monotonic pattern is important for interpreting the spatial distribution of Cd in the study area. A simple urban-center source model would predict a consistent decline in Cd with increasing distance. The negative trend in the urban center agrees with this expectation, but the positive overall correlation and the increasing trend in some outer-suburban samples indicate that additional sources or transport processes are present. Similar deviations from simple distance-decay patterns have been reported in urban environments where industrial land use, road traffic, atmospheric deposition, and local soil disturbance produce multiple concentration peaks rather than a single central maximum (Akbar et al., 2006; Han et al., 2006; Caggiano et al., 2015). Accordingly, distance from the urban center should be treated as an indicator of general urban influence rather than as a direct measure of Cd source strength. More detailed sampling around industrial facilities, main roads, residential areas, irrigation channels, and agricultural land would be necessary to identify the specific sources responsible for the observed local enrichment.

4.3 Organ-specific Cd accumulation in *Ulmus pumila L.*

Cd concentrations differed substantially among the analyzed organs of Ulmus pumila. Bark contained the highest Cd concentrations, whereas roots, branches, and leaves generally showed lower values. Bark also contributed the largest proportion of total Cd accumulated in the four examined organs, accounting for approximately 65–75% in most samples. This organ-specific pattern was observed across the three urban zones, indicating that differences among tissues were more pronounced than differences among the broad geographic zones.

The relatively low Cd concentrations in roots may be associated with limited Cd bioavailability in the soil and with physiological exclusion at the root surface. Total soil Cd does

not necessarily represent the fraction available for plant uptake. Soil pH, carbonate content, organic matter, clay minerals, cation-exchange capacity, and interactions with Fe, Mn, Zn, Ca, and other ions can strongly affect Cd mobility and uptake (Adriano, 2001; Shahid et al., 2017). In addition, plants may reduce Cd transfer to the shoot by binding Cd to root cell-wall components or by storing it in root vacuoles. The cell wall can act as an important barrier because its functional groups are capable of adsorbing and immobilizing metal ions (Krzesłowska, 2011).

The lower Cd concentration in leaves may also indicate restricted translocation to photosynthetically active tissues. After entering the root, Cd can be transported through the xylem, but its movement may be limited by chelation, sequestration, and cellular compartmentalization. These processes help reduce the concentration of free Cd ions and protect physiologically sensitive tissues (Clemens, 2001; Hall, 2002). Metal transport and redistribution are also influenced by transpiration, xylem loading, phloem transport, and the relative sink strength of different tissues (Page and Feller, 2015). Therefore, low leaf Cd concentrations may reflect a physiological strategy that limits Cd exposure in leaves.

The high Cd concentration in bark probably reflects more than one pathway. Bark is a perennial tissue and can retain contaminants over a relatively long period. Its rough and fissured surface can intercept and retain airborne particles, including Cd-bearing dust. Consequently, bark concentrations may integrate internal transport from the roots with direct atmospheric deposition. Research on urban trees has shown that bark and leaves can be used to assess spatial differences in atmospheric trace-metal exposure (Catinon et al., 2009; Sawidis et al., 2011). The relationship between heavy-metal accumulation in U. pumila, rhizosphere soil, and total suspended particles reported by Baidourela et al. (2025) also supports the possibility that both soil and airborne particles contribute to metal accumulation in this species.

However, the present study did not distinguish between surface-deposited and internally absorbed Cd because bark samples were not separated into washed and unwashed fractions. Therefore, the high bark concentration should not be interpreted as evidence that Cd was transported primarily from soil through the roots. Instead, it most likely represents the combined influence of root uptake, internal transport, and long-term particle retention on the bark surface.

4.4 Interpretation of accumulation indices and environmental implications

The accumulation indices further emphasized the difference among organs. The generally

low root BAFs suggest that total soil Cd was not transferred efficiently into roots. In contrast, bark BAFs were relatively high, and most bark samples had BAF values greater than 1. A BAF greater than 1 indicates that the Cd concentration in the tissue exceeded the concentration measured in the associated soil. For bark, however, this ratio reflects total accumulation from both soil-derived and atmospheric pathways. It should therefore not be interpreted solely as an indicator of root uptake efficiency.

The high bark BAFs and the large contribution of bark to total organ Cd suggest that bark is a sensitive tissue for detecting Cd exposure in the study area. This is particularly relevant in the arid environment of Toksun, where dry conditions and wind erosion may enhance the deposition and retention of metal-containing particles. Nevertheless, the higher bark BAFs in the suburban and outer-suburban areas do not necessarily mean that these zones had higher Cd pollution. Because BAF is calculated as a concentration ratio, a lower soil Cd concentration can increase the ratio even when the tissue Cd concentration remains similar. BAF should therefore be evaluated together with absolute soil and plant concentrations.

The results also indicate that U. pumila has potential as a biomonitoring species, particularly when bark is used as the target tissue. Bark is perennial, easily collected, and capable of integrating exposure over an extended period. The strong bark accumulation observed in this study is consistent with previous evidence that trees can retain atmospheric metals on exposed tissues and reflect urban pollution gradients (Catinon et al., 2009; Sawidis et al., 2011). The findings from comparative studies of xerophytic plants also emphasize that species and organ selection are important when evaluating Cd accumulation and phytoremediation potential (Yusuyin et al., 2026). However, the results do not demonstrate that U. pumila is an efficient Cd phytoextractor. Phytoextraction requires effective soil uptake, substantial transport to harvestable tissues, high biomass production, and tolerance to metal toxicity. In the present study, Cd was concentrated mainly in bark, while root, branch, and leaf BAFs were generally low. Thus, U. pumila may be more suitable for long-term biomonitoring or phytostabilization than for the removal of large quantities of Cd from soil.

Several limitations should be considered. First, only total soil Cd was measured, and the bioavailable fraction was not determined. Second, soil pH, organic matter, carbonate content, and other factors affecting Cd mobility were not included. Third, the study was based on one sampling

period and did not consider seasonal changes in atmospheric deposition or plant growth. Fourth, the absence of washed and unwashed bark comparisons prevented a quantitative separation of external deposition from internal uptake. Future studies should combine total and bioavailable soil Cd with soil properties, suspended particulate matter, atmospheric deposition, and paired washed and unwashed bark samples. Despite these limitations, the present study demonstrates that Cd distribution in U. pumila is strongly organ-specific and that bark may provide a useful indicator of Cd exposure in arid urban and peri-urban environments.

5 Conclusions

Soil Cd showed considerable spatial heterogeneity across the study area, with concentrations generally exceeding the Xinjiang regional background level and higher values mainly occurring in the urban center and adjacent suburban areas. However, the absence of a significant difference among the three urban zones and the relatively weak relationship between distance and soil Cd concentration indicate that urban proximity alone cannot fully explain the observed distribution. Local anthropogenic sources and site-specific environmental conditions likely played important roles.

Cd accumulation in Ulmus pumila was strongly organ-specific. Bark contained the highest Cd concentrations, accounted for approximately 65–75% of the Cd accumulated in the four analyzed organs, and exhibited substantially higher BAFs than roots, branches, and leaves. Median bark BAFs ranged from 3.19 to 4.70, and 86–100% of bark samples had BAF values greater than 1, whereas the other organs generally had BAFs below 1. These results demonstrate preferential Cd retention in bark and suggest that bark is a potentially sensitive tissue for monitoring Cd accumulation in urban and peri-urban environments. Nevertheless, the contribution of atmospheric deposition to bark Cd accumulation should be further separated from root-derived Cd uptake in future studies.

Declarations

Author contributions

Nuerbiye Kurban: Investigation, Formal analysis, Visualization, Writing – original draft.

Huihui Wen: Investigation, Methodology, Data curation.

Shijie Dao: Methodology, Data curation.

Fuxiang Ma: Supervision, Writing – review & editing.

Aliya Baidourela: Conceptualization, Supervision, Funding acquisition, Writing – review & editing.

All authors read and approved the final manuscript.

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Competing interests

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.

Data availability

The datasets generated and analyzed during the current study are available from the corresponding author on reasonable request.

Ethics approval and consent to participate

Not applicable. This study did not involve human participants or animals.

Consent for publication

All authors have read and approved the final manuscript and consent to its publication in Environmental Monitoring and Assessment.

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