

PM_{2.5}-Bound Metals in a High-Tech Industrial Area of Northern Taiwan: Spatial Distribution, Sources, and Health Risks

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Abstract

Fine particulate matter (PM_{2.5}) is an imperative environmental health concern because its elemental composition and toxicity vary substantially with emission sources. However, the characteristics and health implications of PM_{2.5}-bound metals in high-tech industrial regions remain insufficiently characterized. This study investigated the concentrations, elemental composition, spatial distribution, source contributions, and potential health risks of PM_{2.5}-bound metals in a high-tech industrial region of northern Taiwan. Ambient PM_{2.5} samples were collected at 71 sites during two monitoring campaigns conducted in 2019 and 2020. The mean PM_{2.5} concentration was $19.33 \pm 8.97 \mu\text{g m}^{-3}$, with higher concentrations in autumn ($23.51 \pm 10.40 \mu\text{g m}^{-3}$) than in summer ($14.57 \pm 6.51 \mu\text{g m}^{-3}$), and localized hotspots were observed near industrial parks. Inductively coupled plasma mass spectrometry identified 30 particulate elements, with a mean total metal concentration of $736.85 \pm 751.20 \text{ ng m}^{-3}$. The metal profile was dominated by Zn, Ba, Cu, Pb, and Mn. Positive Matrix Factorization identified four major sources of particulate metals: fossil fuel combustion (33.1%), soil and road dust resuspension (29.3%), metal-related industrial emissions (22.3%), and tire and brake wear (22.3%). Probabilistic health-risk assessment using Monte Carlo simulation estimated a combined lifetime incremental cancer risk (ILCR) of 2.60×10^{-4} , exceeding the commonly applied acceptable risk level of 1×10^{-6} , with As and Cr as the principal contributors. Non-carcinogenic risks were substantially higher for children than adults, with hazard indices (HI) of 4.65 and 0.69, respectively, and were primarily associated with incidental ingestion of contaminated dust. Conclusively, the findings indicate that PM_{2.5} mass concentration alone may not adequately reflect the health risks of metal-rich particulate pollution in high-tech industrial environments. Source-specific controls targeting industrial metal emissions and traffic-related non-exhaust emissions, together with continued spatially resolved monitoring of toxic metals, are warranted to reduce potential health risks to nearby populations.

Introduction

Fine particulate matter with an aerodynamic diameter $\leq 2.5 \mu\text{m}$ (PM_{2.5}) has received considerable attention because of its significant impact on human health (Englert 2004; WHO 2005). Owing to their small size, PM_{2.5} can penetrate deeply into the human respiratory system, accumulate in the lungs, and even enter the bloodstream. Numerous epidemiological studies demonstrated that long-term exposure to PM_{2.5} is associated with increased risks of chronic obstructive pulmonary disease (COPD), cardiovascular diseases, stroke, and lung cancer (Brook et al. 2010; Dockery et al. 1993; Hystad et al. 2020; Krewski et al. 2009; Leppeule et al. 2012; Pope et al. 2002, 2004, 2006, 2009; Pun et al. 2017; Thangavel et al. 2022).

PM_{2.5} is composed of a complex mixture of chemical constituents, including elemental carbon (EC), organic carbon (OC), water-soluble ions (e.g., Na⁺, SO₄²⁻, K⁺, and NO₃⁻), trace metals (e.g., As, Cd, and Pb), and polycyclic aromatic hydrocarbons (PAHs) (Chen et al. 2021). The chemical characteristics of PM_{2.5} largely depends on its formation mechanisms and emission sources, which may originate from both natural and anthropogenic activities. Primary PM_{2.5} can be directly emitted from sources such as sea spray, volcanic activity, fossil fuel combustion, vehicle emissions, construction activities, biomass burning, and industrial processes. In contrast, secondary PM_{2.5} is formed in the atmosphere through chemical transformations of gaseous precursor pollutants (Zhang et al. 2015).

Because PM_{2.5} originates from diverse sources, its chemical composition can vary significantly across regions, resulting to differences in toxicity and health outcomes (Cao et al. 2012). Among the various chemical components of PM_{2.5}, heavy metals have attracted particular attention owing to their high toxicity, persistence in the environment, and potential for bioaccumulation (Ali et al. 2019; Duan et al. 2020; Moryani et al. 2020; Tsai et al. 2020). Exposure to toxic metals in PM_{2.5} may therefore pose substantial risks to human health.

According to the World Health Organization, ambient air pollution contributes to approximately seven million premature deaths worldwide each year, thus the ambient PM_{2.5} exposure is a major contributor to this global health burden (WHO 2021). Furthermore, more than half of the global population is exposed to a PM_{2.5} level higher than 35 µg m⁻³, which was previously defined as the interim target level by the WHO air quality guidelines (Health Effects Institute 2019, 2020).

In Taiwan, previous studies on PM_{2.5} and its chemical composition have primarily focused on areas surrounding traditional heavy industries (Hsu et al. 2016; Tsai et al. 2020). However, in recent decades, high-tech industries, particularly semiconductor manufacturing, have become major drivers of economic development. The rapid expansion of high-tech industrial parks has also resulted in increasing population in their surrounding areas. Despite this trend, relatively limited information is available regarding the characteristics and potential health risks of PM_{2.5} in high-tech industry regions.

Therefore, this study aimed to investigate the ambient PM_{2.5} concentrations and their elemental composition, with a particular focus on heavy metals, in a high-tech industrial area in northern Taiwan. In addition, potential emission sources of PM_{2.5}-associated metals were identified, and the associated human health risks were evaluated. The findings provide important insights into the characteristics and potential impacts of PM_{2.5} in high-tech industrial environments.

Material and Methods

Study site and Sample Collection

This study was carried out in the vicinity of the Hsinchu Science Park (24.78269° N, 121.00628° E), located in northern Taiwan. That included Hsinchu City (~ 100 km²), Hsinchu County (~ 1400 km²), and Miaoli County (~ 1800 km²), with a total population of approximately 1.5 million residents. A substantial proportion of the population resided near the industrial parks; for example, about 500,000 people lived in Hsinchu City. In addition to a major global hub for semiconductor manufacturing, thin-film transistor liquid-crystal display (TFT-LCD) production, and LED fabrication, the region also contained several traditional industries, including petrochemical and metal casting facilities.

A total of 71 sampling sites were included in this study, as illustrated in Fig. 1. Ambient PM_{2.5} samples were collected at a height of approximately 1.8 m above ground level, representing the typical human breathing zone. Sampling was conducted during two periods: August to October 2019 (n = 71) and May to July 2020 (n = 61).

PM_{2.5} samples were collected using a Personal Environmental Monitor (PEM[®], MSP Corporation, USA) equipped with Teflon filters, operating at a flow rate of 10 L min⁻¹ for 24-hour sampling periods. A previous field comparison between the PEM[®] and a dual sequential fine particle sampler (URG-2000-01K) demonstrated strong agreement in PM_{2.5} mass concentrations (R² = 0.90)(Li 2018). Field blank samples were collected at a frequency of at least one blank for every four field samples to estimate potential contamination.

Sample Pretreatment and Quantification

Gravimetric analysis of PTFE filters was carried out to determine PM_{2.5} mass concentrations. Prior to weighing, filters were conditioned for at least 24 hours in a temperature and humidity-controlled chamber at 23 ± 1°C and 40 ± 5% relative humidity. Filters were weighed using an electronic microbalance (Mettler-Toledo MX5; sensitivity ± 1 µg).

Each filter was weighed at least three times before and after sampling, and measurements were considered valid only when the difference among repeated weighings was less than 10 µg. The net particle mass was calculated as the difference between post-sampling and pre-sampling weights. After weighing, filters were stored at -20°C until chemical analysis.

For metal analysis, filters were digested using microwave-assisted acid digestion. Each sample was treated with an acid mixture containing 5 mL HNO₃ and 0.3 mL HF at 200°C for 60 min, followed by the addition of 2.8 mL of 5% H₃BO₃ and further digestion at 140°C for 50 min using a microwave digestion system (Anton Paar Multiwave 3000). After digestion, the solutions were cooled and diluted to 15 mL with 5% HNO₃.

All reagents used were of trace-metal analytical grade, and ultrapure water (Milli-Q®, 18.3 MΩ) was used throughout the analysis.

A total of 30 elements, including Mg, Al, Ca, Ti, V, Cr, Mn, Fe, Co, Ni, Cu, Zn, As, Sr, Mo, Cd, Sb, Ba, Pb, La, Ce, Pr, Nd, Sm, Yb, and Lu, were quantified using inductively coupled plasma mass spectrometry (ICP-MS; Agilent 7700x).

Method detection limits (MDLs) ranged from 0.001 to 3.14 µg L⁻¹ for the selected metals. Calibration curves were established using multi-element standard solutions (SPEX CertiPrep) prepared in 5% HNO₃, with correlation coefficients R² > 0.995.

Quality Assurance and Control

Quality assurance and quality control (QA/QC) procedures were implemented to ensure analytical accuracy and precision. A certified reference material (NIST SRM-2786, urban particulate matter) was analyzed to verify method performance. Acceptable analytical criteria were defined as accuracy within ± 20% and precision better than 10%.

Both laboratory blanks and field blanks were analyzed to evaluate potential contamination during sampling and analysis. Analytical batches were considered acceptable when laboratory blank concentrations were less than twice the method detection limit (MDL).

In addition, at least 10% of samples were subjected to spike recovery and quality control analyses. Acceptable recovery ranges were 75–125% for spiked samples and 80–120% for quality control samples.

Source Apportionment

Pearson correlation analysis and principal component analysis (PCA) were performed to investigate possible emission sources (IBM SPSS Statistics Version 22).

To apportion sources of PM_{2.5}-bound metals, Positive Matrix Factorization (PMF version 5.0) developed by the United States Environmental Protection Agency (USEPA) was applied. Data were pre-treated following USEPA guidelines. Concentrations below the method detection limit (MDL) were replaced with MDL/2, with uncertainties set to 5/6 × MDL. Species were classified by signal-to-noise ratio (S/N): variables with S/N < 0.5 were excluded, 0.5 ≤ S/N < 2 were down-weighted, and S/N ≥ 2 were retained. Uncertainties were calculated as:

$$\text{Uncertainty} = \sqrt{[(\text{error fraction} \times \text{concentration})^2 + (\text{MDL})^2]}$$

The optimal number of factors (3–6 tested) was selected based on the Q value, residuals, and interpretability, with a four-factor solution chosen. Model stability was evaluated using bootstrap (BS) and displacement (DISP) analyses, indicating stable and robust results. Identified sources included fossil fuel combustion, road dust resuspension, tire and brake wear, and metal-related industry.

Health Risk Assessment

Health risk assessment was conducted following the methodology recommended by the U.S. Environmental Protection Agency (USEPA) and the Integrated Risk Information System (IRIS) (USEPA, 1989, 2004, 2005, 2009). Human exposure to particulate-bound metals was evaluated through inhalation, incidental ingestion, and dermal contact.

The time-weighted exposure concentration (EC) for inhalation was calculated as:

$$EC = C \times ET \times EF \times ED \times AT^{-1}$$

where C is the ambient metal concentration ($\mu\text{g m}^{-3}$), ET the exposure time (h day^{-1}), EF the exposure frequency (days year^{-1}), ED the exposure duration (years), and AT the averaging time.

The average daily dose (ADD) from incidental ingestion of deposited particulate matter, primarily resulting from hand-to-mouth and object-to-mouth behaviors in young children, was estimated as:

$$ADD_{\text{ing}} = C \times \text{IngR} \times EF \times ED \times CF \times BW^{-1} \times AT^{-1}$$

where IngR is the age-specific particle ingestion rate, BW body weight, AT the averaging time, and CF the unit conversion factor ($1 \times 10^{-6} \text{ kg mg}^{-1}$). When oral bioavailability was considered, the estimated dose was adjusted using the gastrointestinal absorption factor (GIABS) or an experimentally determined bioaccessible fraction.

Dermal exposure was estimated as:

$$ADD_{\text{derm}} = C \times SA \times AF \times ABS \times EV \times EF \times ED \times CF \times BW^{-1} \times AT^{-1}$$

where SA is the exposed skin surface area, AF the skin adherence factor, ABS the chemical-specific dermal absorption fraction, and EV the exposure event frequency (1 event day^{-1}). A higher skin adherence factor ($0.2 \text{ mg cm}^{-2} \text{ event}^{-1}$) was assigned to children to reflect their greater contact with dust-contaminated surfaces.

Carcinogenic risk was evaluated by estimating the incremental lifetime cancer risk (ILCR) for each exposure pathway. For ingestion and dermal contact, ILCR was calculated as:

$$ILCR = ADD \times SF$$

where SF is the chemical-specific cancer slope factor. For inhalation exposure, cancer risk was estimated as:

$$ILCR_{\text{inh}} = EC \times UR$$

where UR is the inhalation unit risk. The total lifetime cancer risk was obtained by summing ILCR values across all exposure pathways and carcinogenic metals. Cancer risks exceeding 1×10^{-6} were regarded as indicative of an unacceptable lifetime cancer risk. To characterize uncertainty and inter-individual variability, probabilistic risk assessment was performed using Monte Carlo simulation with 200,000 iterations (R4.3.3).

Non-carcinogenic risk was assessed using the hazard quotient (HQ), calculated by dividing the estimated exposure (EC for inhalation or ADD for ingestion and dermal contact) by the corresponding reference concentration (RfC) or reference dose (RfD). The hazard index (HI) was calculated as the sum of HQs across all exposure pathways. A HI greater than 1 indicates the potential for adverse non-carcinogenic health effects. Because summation of HQs assumes dose additivity, organ-specific hazard indices were additionally calculated for metals with different primary target organs (e.g., Mn for neurological toxicity and Cd for renal toxicity) to provide a more biologically relevant evaluation for both children and adults.

The exposure parameters, age-specific assumptions, and toxicological reference values used in the risk assessment are summarized in Tables S1 and S2.

Results and Discussion

Ambient PM_{2.5} Concentrations

The average ambient PM_{2.5} concentration in the study area was $19.33 \mu\text{g m}^{-3}$, ranging from 3.77 to $48.99 \mu\text{g m}^{-3}$. Seasonal variation was observed, with significantly higher concentrations in autumn ($23.51 \pm 10.40 \mu\text{g m}^{-3}$) compared with those in summer ($14.57 \pm 6.51 \mu\text{g m}^{-3}$). This seasonal pattern is consistent with historical monitoring records from the Atmospheric Air Quality Monitoring Station network operated by the Taiwan Ministry of Environment.

Although the summer sampling campaign (May–July 2020) coincided with the COVID-19 pandemic, the relatively lower PM_{2.5} concentrations are unlikely to be directly related to pandemic-related emission reductions. Unlike many countries, Taiwan did not implement extensive lockdown measures during this period. Data from the Hsinchu Atmospheric Air Quality Monitoring Station (24.8056356°N , $120.97236752^\circ\text{E}$) indicate that average PM_{2.5} concentrations during the summers of 2019 and 2020 were $10.7 \pm 5.8 \mu\text{g m}^{-3}$ and $9.7 \pm 5.0 \mu\text{g m}^{-3}$, respectively, that suggests pandemic-related reduction was neglected. In fact, increased use of private vehicles instead of public transportation during the pandemic may have offset potential emission reductions by increasing traffic-related emissions (Chang et al. 2021).

The spatial distribution of PM_{2.5} concentrations (Fig. 1) implies that traffic and industrial emissions are major determinants of local air quality. Coastal areas of Hsinchu City and mountainous regions in eastern Hsinchu County generally exhibited PM_{2.5} concentrations below the annual guideline value of $15 \mu\text{g m}^{-3}$. In contrast, sampling sites near Hsinchu Science Park and Hsinchu Fengshan Industrial Park (24.8825°N , 121.0369°E) showed concentrations ranging from 15 – $35 \mu\text{g m}^{-3}$, while the highest levels ($> 35 \mu\text{g m}^{-3}$) were observed near Hsinchu Biomedical Science Park (24.80575°N , 121.04831°E) and Zhunan Industrial Park (24.7111°N , 120.9125°E) emphasizing the influence of industrial activities.

Overall, PM_{2.5} concentrations in the studied high-tech region were below the World Health Organization Interim Target-II ($25 \mu\text{g m}^{-3}$) but slightly above Interim Target-III ($15 \mu\text{g m}^{-3}$). Compared with global observations (Table 1), the measured concentrations are similar to those reported in Warsaw, Poland, but substantially lower than those reported in many cities in China and India. Within Taiwan, PM_{2.5} levels in the study area are higher than those measured in Taipei and remote regions, but lower than those reported in industrialized or mixed-use areas such as Changhua and Kaohsiung.

Concentrations of Particulate Metals

The total concentration of particulate metals in PM_{2.5} averaged $736.9 \pm 751.2 \text{ ng m}^{-3}$ (Table 2). Among these, light metals accounted for $579.74 \pm 649.09 \text{ ng m}^{-3}$, while heavy metals contributed $156.79 \pm 161.50 \text{ ng m}^{-3}$. Lanthanides and uranium were present at trace levels ($0.32 \pm 0.48 \text{ ng m}^{-3}$).

The overall abundance of toxic metals followed the order:

$\text{Zn} > \text{Ba} > \text{Cu} > \text{Pb} > \text{Mn} > \text{Cr} > \text{Ni} > \text{V} > \text{Se} > \text{As} > \text{Sb} > \text{Mo} > \text{Cd}$

Table 1
The selected reported ambient PM_{2.5} level (µg/m³)

Region	Type	PM _{2.5}	Reference
Great Hsinchu, Taiwan	High-tech	19.3	This study
Taipei, Taiwan	Urban	15.6	(Hsu et al. 2021)
Chiayi, Taiwan	Suburban	25.6	(Hsu et al. 2021)
Hualien, Taiwan	Agricultural	10.2	(Hsu et al. 2021)
Kaohsiung, Taiwan	Industrial	25.3	(Hsu et al. 2021)
Shula, Taiwan	Mixed	39.2	(Tsai et al. 2020)
Changhua, Taiwan	Mixed	34.0	(Hsu et al. 2016)
Kaohsiung, Taiwan	Harbor	24.1	(Tseng et al. 2021)
Ningbo, China	Urban	54.3	(Xu et al. 2021)
Ningbo, China	Suburban	51.4	(Xu et al. 2021)
Hangzhou, China	Urban	80	(Xu et al., 2021)
Hangzhou, China	Background	66.3	(Xu et al. 2021)
Chengdu, China	Urban	67.44	(Kong et al. 2020)
Luoyang, China	Industrial	91.1	(Xu et al. 2022)
Pukou, China	Mixed	68.6	(Xie et al. 2020)
Xuanwu, China	Urban	61.1	(Xie et al. 2020)
Lishui, China	Rural	47.9	(Xie et al. 2020)
Tianhe, China	Urban	50.0	(Xie et al. 2020)
Conghua, China	Suburban	46.7	(Xie et al. 2020)
Heshan, China	Mixed	69.0	(Xie et al. 2020)
Beijing, China	Residential	133.97	(Cui et al. 2020)
Manila, Philippines	Harbor	35.3	(Tseng et al. 2021)
Kuala Lumpur, Malaysia	Urban	11.4	(Othman et al. 2019)
Dhaka, Bangladesh	Urban	32.7	(Rahman et al. 2021)
Greater Kolkata, India	Urban	313	(Das et al. 2015)
Los Angeles, USA	Urban	13.75	(Farahani et al. 2021)
Cordoba, Argentina	Urban	48.1	(Lanzaco et al. 2017)
Sao Paulo, Brazil	Urban	22.1	(de Miranda et al. 2018)
Milan, Italy	Industrial	87.32	(Zoran et al. 2020)
Madrid, Spain	Urban	34.14	(Artiñano et al. 2003)
Warsaw, Poland	Urban	18.8	(Juda-Rezler et al. 2020)

Region	Type	PM _{2.5}	Reference
Kosetice, Czech	Agricultural	13.6	(Pokorná et al. 2018)

Many of these metals are commonly associated with vehicular emissions. For example, Zn, Ba, Cu, and Pb are typical markers of traffic-related emissions, including tailpipe exhaust and non-exhaust sources such as brake and tire wear (Johansson et al. 2009;

Table 2
The ambient particulate metal level (ng/m³).

	August (2019)		Summer (2020)		All	
	mean	SD	mean	SD	mean	SD
Na	316.97	284.43	272.60	343.67	296.29	314.22
Mg	55.09	51.85	44.35	79.72	50.08	66.53
Al	142.71	256.68	50.20	194.07	99.58	234.22
K	107.80	180.54	124.09	272.27	115.39	228.09
Ti	13.10	45.81	5.39	8.71	9.50	34.21
Se	1.39	1.32	1.25	2.57	1.33	2.00
Sr	1.10	1.78	0.66	1.37	0.90	1.62
Ba	10.93	59.79	1.77	1.61	6.66	43.94
light-metal	649.09	562.94	500.33	727.38	579.74	649.09
V	3.32	2.10	0.62	0.46	2.06	2.07
Cr	2.32	3.48	2.11	5.16	2.22	4.35
Mn	4.73	3.38	2.45	2.35	3.67	3.16
Fe	87.72	71.40	64.76	117.72	77.02	96.50
Co	0.10	0.12	0.08	0.10	0.09	0.11
Ni	3.32	4.98	0.70	2.58	2.10	4.25
Cu	6.43	8.84	5.11	22.93	5.81	16.95
Zn	48.16	59.25	66.25	67.72	56.59	63.98
As	1.54	1.40	0.51	0.55	1.06	1.21
Mo	0.40	1.17	0.42	0.38	0.41	0.89
Cd	0.29	0.33	0.13	0.14	0.21	0.27
Sb	1.09	0.78	0.56	0.55	0.84	0.73
Pt	0.00	0.00	0.00	0.00	0.00	0.00
Pb	7.20	6.10	1.85	2.60	4.70	5.49
heavy-metal	164.71	147.78	145.55	176.45	155.78	162.06
Lu	0.00	0.00	0.00	0.00	0.00	0.00
La	0.09	0.06	0.08	0.19	0.09	0.14
Ce	0.13	0.09	0.16	0.37	0.14	0.27
Pr	0.01	0.01	0.02	0.08	0.02	0.05
Nd	0.03	0.03	0.08	0.28	0.06	0.19
Sm	0.01	0.00	0.01	0.01	0.01	0.01

	August (2019)		Summer (2020)		All	
	mean	SD	mean	SD	mean	SD
Yb	0.00	0.00	0.00	0.00	0.00	0.00
U	0.01	0.01	0.00	0.01	0.01	0.01
Lanthanide and U	0.29	0.18	0.36	0.68	0.32	0.48
overall	815.98	656.44	646.24	837.66	736.85	751.20

Lin et al. 2015). Tailpipe emissions may contain Zn, Pb, Ni, and V, whereas brake wear

and mechanical abrasion release Cu, Cd, Fe, Mn, and Sb. Resuspended road dust can also contribute elements such as Ca, Mg, and K.

The Sb/Cu ratio (~ 0.22) observed in this study is comparable to previously reported values for brake-wear particles, as indicates the importance of non-exhaust traffic emissions (Sternbeck et al., 2002).

Several measured elements are classified as carcinogenic by the International Agency for Research on Cancer (IARC). The average concentrations of these toxic metals were: As: $1.06 \pm 1.21 \text{ ng m}^{-3}$, Cd: $0.21 \pm 0.27 \text{ ng m}^{-3}$, Cr: $2.22 \pm 4.35 \text{ ng m}^{-3}$, Ni: $2.10 \pm 4.25 \text{ ng m}^{-3}$, Pb: $4.70 \pm 5.49 \text{ ng m}^{-3}$, Sb: $0.84 \pm 0.73 \text{ ng m}^{-3}$, Co: $0.094 \pm 0.11 \text{ ng m}^{-3}$.

Lanthanide elements exhibited a distribution pattern consistent with natural crustal abundance. Specifically, the observed order $\text{Ce} > \text{La} > \text{Nd} > \text{Pr} > \text{Sm} > \text{Yb} > \text{Lu}$ corresponds well with typical crustal compositions (Hedrick, 1995), therefore these elements were mainly suggested from natural geological sources rather than anthropogenic emissions.

Source Apportionment of PM_{2.5} Metals

Pearson correlation analysis (Fig. 2) revealed strong correlations ($r > 0.7$) among several metals, such as Fe–Cu, Co–Ni, As–Pb, and Pb–Cd.

The suitability for multivariate analysis was verified using the Kaiser-Meyer-Olkin (KMO) test and Bartlett's test of sphericity, both confirmed the appropriateness of applying principal component analysis (PCA).

Based on PCA results, four major sources of particulate metals were identified. These sources were subsequently applied in positive matrix factorization (PMF) modeling. Elements with signal-to-noise ratios (S/N) lower than 0.5 were excluded from the analysis, leaving fifteen elements for source apportionment.

The PMF results indicate that PM_{2.5}-bound metals in the study area originate primarily from the following four sources: Fossil fuel combustion (33.1%), Soil and road dust resuspension (29.3%), Tire and brake wear (22.3%), Metal-related industrial emissions (22.3%).

Factor 1 was dominated by Cu (89.3%) and Al (54.5%), suggesting contributions from metal processing and smelting activities (Hsu et al. 2016; Querol et al. 2007). Factor 2 showed high loadings of V (87.1%), Pb (76.6%), Se (70.5%), and Ni (68.3%), which are commonly associated with fossil fuel combustion, including industrial boilers and diesel engines (Almeida et al. 2005; Hueglin et al. 2005). Factor 3 was characterized by Fe (79.9%), K (68.4%), Ti (57.8%), and Ba (55.5%), indicating crustal materials and road dust resuspension. Factor 4 was dominated by Mn (100%) and Zn (85.4%),

suggesting emissions from tire and brake wear, which are well-known non-exhaust traffic sources (Lough et al. 2005; Weckwerth 2001). The relative contributions of these sources to PM_{2.5}-bound metals are illustrated in Fig. 3.

Health Risk Assessment

Risk was evaluated for two population groups, children under 6 years old and adults, across three exposure routes. Aggregate incremental lifetime cancer risk was 1.73×10^{-4} and 8.64×10^{-5} for the children and the adults, respectively (Table 3), as a result of a combined lifetime risk of 2.60×10^{-4} . If the inhalation-derived surrogate slope factors conventionally applied to Cd and Ni, the cancer risk would be elevated to 4.72×10^{-4} , 2.19×10^{-4} and 6.91×10^{-4} respectively. Both lifetime estimates exceed two-fold of the acceptable risk of 1×10^{-6} . The dominant risk elements are As and Cr for both child and adults. It's noteworthy, the Cr might be released from the traffic-related sources

Table 3
Incremental lifetime cancer risk

		mean	SD	5%	25%	50%	75%	90%	95%
As	< 6 y	9.83×10^{-5}	1.12×10^{-4}	1.44×10^{-5}	3.49×10^{-5}	6.47×10^{-5}	1.20×10^{-4}	2.08×10^{-4}	2.92×10^{-4}
	adult	4.48×10^{-5}	5.13×10^{-5}	6.57×10^{-6}	1.59×10^{-5}	2.95×10^{-5}	5.47×10^{-5}	9.51×10^{-5}	1.33×10^{-4}
Cd ‡	< 6 y	8.39×10^{-5}	1.09×10^{-4}	1.01×10^{-5}	2.63×10^{-5}	5.12×10^{-5}	1.00×10^{-4}	1.83×10^{-4}	2.61×10^{-4}
	adult	3.76×10^{-5}	4.91×10^{-5}	4.52×10^{-6}	1.18×10^{-5}	2.30×10^{-5}	4.48×10^{-5}	8.21×10^{-5}	1.17×10^{-4}
Cr	< 6 y	7.25×10^{-5}	1.42×10^{-4}	4.16×10^{-6}	1.40×10^{-5}	3.29×10^{-5}	7.65×10^{-5}	1.65×10^{-4}	2.61×10^{-4}
	adult	4.02×10^{-5}	7.88×10^{-5}	2.31×10^{-6}	7.78×10^{-6}	1.82×10^{-5}	4.24×10^{-5}	9.17×10^{-5}	1.45×10^{-4}
Ni ‡	< 6 y	2.15×10^{-4}	4.55×10^{-4}	1.18×10^{-5}	4.04×10^{-5}	9.52×10^{-5}	2.26×10^{-4}	4.90×10^{-4}	7.79×10^{-4}
	adult	9.50×10^{-5}	2.01×10^{-4}	5.21×10^{-6}	1.78×10^{-5}	4.20×10^{-5}	9.96×10^{-5}	2.16×10^{-4}	3.43×10^{-4}
Pb	< 6 y	2.31×10^{-6}	2.71×10^{-6}	3.26×10^{-7}	8.01×10^{-7}	1.50×10^{-6}	2.80×10^{-6}	4.93×10^{-6}	6.86×10^{-6}
	adult	1.01×10^{-6}	1.19×10^{-6}	1.43×10^{-7}	3.51×10^{-7}	6.57×10^{-7}	1.23×10^{-6}	2.16×10^{-6}	3.01×10^{-6}
Total §	< 6 y	1.73×10^{-4}	1.81×10^{-4}	3.63×10^{-5}	7.41×10^{-5}	1.24×10^{-4}	2.09×10^{-4}	3.44×10^{-4}	4.68×10^{-4}
	adult	8.64×10^{-5}	9.39×10^{-5}	1.79×10^{-5}	3.65×10^{-5}	6.09×10^{-5}	1.04×10^{-4}	1.71×10^{-4}	2.35×10^{-4}
	lifetime	2.60×10^{-4}	2.75×10^{-4}	5.43×10^{-5}	1.11×10^{-4}	1.85×10^{-4}	3.13×10^{-4}	5.15×10^{-4}	7.03×10^{-4}
Total ‡§	< 6 y	4.72×10^{-4}	5.01×10^{-4}	1.29×10^{-4}	2.30×10^{-4}	3.53×10^{-4}	5.54×10^{-4}	8.69×10^{-4}	1.17×10^{-3}
	adult	2.19×10^{-4}	2.27×10^{-4}	6.03×10^{-5}	1.07×10^{-4}	1.64×10^{-4}	2.58×10^{-4}	4.02×10^{-4}	5.40×10^{-4}
	lifetime	6.91×10^{-4}	7.27×10^{-4}	1.90×10^{-4}	3.38×10^{-4}	5.17×10^{-4}	8.11×10^{-4}	1.27×10^{-3}	1.71×10^{-3}

‡ Cd and Ni have no established oral cancer slope factor; the values shown apply the inhalation-derived surrogates of 6.3 and 1.7 (mg kg⁻¹ day⁻¹)⁻¹ in common use but unsupported by IRIS. § The first Total block restricts oral and dermal risk to As, Cr

and Pb and counts Cd and Ni by inhalation only; the second ($\pm$ §) includes the surrogates. Stratum risks are additive because both are averaged over the same 70-year lifespan; the lifetime row aggregates the 6-year childhood and 24-year adult windows only, a total of 30 exposure-years. Cr risk assumes all measured chromium is Cr(VI).

whereas the significant source of As was not identified by the PMF model. One may suggest the semiconductor manufacturing may contribute a significant amount of As into the environment. In addition, childhood risk exceeded adult risk by a factor of 2.0 despite the childhood window being one quarter the length of the adult window; expressed per year of exposure the disparity is a factor of 8.0, with $2.89 \times 10^{-5} \text{ y}^{-1}$ in children against $3.60 \times 10^{-6} \text{ y}^{-1}$ in adults.

The non-carcinogenic results isolate the age contrast more cleanly, since hazard quotients are not scaled by exposure duration. The hazard index was 4.65 in children under 6 years against 0.69 in adults, a ratio of 6.7, so the threshold of unity was exceeded in children only. Arsenic was the dominant contributor in both groups, with a hazard quotient of 2.61 in children, and lead approached unity at 0.90; no other element exceeded 0.6 in either group. At the 95th percentile the childhood hazard index reached 10.59, and the adult value of 1.49 also crossed unity, indicating that adults are not free of concern under upper-percentile exposure conditions even though their central estimate is well below the threshold. The differential arises entirely from physiology and behavior rather than from any difference in ambient concentration: children were assigned a dust ingestion rate twice that of adults against a body weight less than one quarter, and a skin adherence factor approximately three times higher.

Route resolution showed incidental ingestion to be dominant in both groups, contributing 91.6% of cancer risk in children and 84.7% in adults, with dermal contact supplying most of the remainder. Inhalation contributed a hazard index of 0.20 in both groups, identical across ages because inhalation exposure under the concentration-based framework is independent of body weight and ventilation rate; all age differentiation therefore enters through the ingestion and dermal routes.

Applying age-dependent adjustment factors for the mutagenic mode of action of Cr(VI) widened the gap further (Table 3). The weighted adjustment factor was 5.33 for children under 6 years against 1.00 for adults, raising chromium risk in children from 7.25×10^{-5} to 3.87×10^{-4} and making it a leading contributor to childhood risk. Because these adjustments apply only where C(VI) is actually present, and the present analysis conservatively assumes that all measured chromium is Cr(VI), direct speciation measurement would materially constrain the childhood estimate.

The estimates of ingestion and dermal exposure remain subject to uncertainty, since these estimates may be inversely related to particle mass. Thus, the estimates using the measured PM_{2.5} concentration range yielded the childhood hazard index from 1.95 to 22.9. These values should be interpreted as formal bounds rather than realistic exposure scenarios, since low-mass days would not carry mean metal loadings. A more important source of uncertainty is the conventional assumption assigned the composition of fine particulate matter to 100–200 mg day⁻¹ of ingested soil and dust; however, PM_{2.5} in this study area was enriched relative to surface soil by approximately one-fold higher for several elements contributing substantially to the risk estimates. Since ingested dust is expected to originate predominantly from coarse resuspended soil rather than the fine particulate fraction, the ingestion and dermal risk estimates should therefore be considered conservative upper-bound estimates. When an apportionment factor (f) was introduced to account for the contribution of PM_{2.5} to ingested dust, the childhood hazard index decreased to 0.65, 0.42, and 0.25 for f = 0.10, 0.05, and 0.01, respectively. In contrast, the inhalation estimates require no analogous extrapolation and are directly constrained by the measured airborne metal concentrations. Importantly, the relative ranking of exposure between children and adults remained unchanged under both assumptions, because the same scaling factor was applied to both age groups.

Conclusion

This study provides a comprehensive assessment of ambient PM_{2.5} mass concentrations, metal composition, source contributions, and associated human health risks in the vicinity of a major high-tech industrial region in northern Taiwan.

The mean PM_{2.5} concentration was 19.33 µg m⁻³, indicating compliance with the WHO Interim Target II guideline; however, pronounced spatial heterogeneity was observed, with elevated concentrations and localized hotspots in the vicinity of active science parks and industrial areas. These spatial patterns highlight the influence of industrial activities and traffic-related emissions on local PM_{2.5} pollution.

Source apportionment identified four major contributors to the atmospheric metal burden: fossil fuel combustion (33.1%), resuspended crustal and road dust (29.3%), metal-processing industrial emissions (22.3%), and non-exhaust traffic emissions associated with tire and brake wear (22.3%). The results demonstrate that metal pollution in high-tech industrial environments arises from multiple interacting sources rather than from a single dominant emission sector. Health-risk assessment further indicated that, although the overall PM_{2.5} mass concentrations were moderate, exposure to PM_{2.5}-bound metals warrants attention. In particular, arsenic (As) and hexavalent chromium [Cr(VI)] were the principal contributors to carcinogenic risk, with the estimated lifetime cancer risk reaching 2.60×10^{-4} . Non-carcinogenic risks were more pronounced among children, with a hazard index of 4.65, largely associated with ingestion of contaminated dust. However, the ingestion and dermal exposure estimates are subject to considerable uncertainty because they depend on assumptions regarding the fraction of fine particulate metals represented in ingested soil and dust.

Overall, these findings indicate that compliance with PM_{2.5} mass-based standards alone may not adequately capture the potential health implications of metal-rich particulate pollution in high-tech industrial regions. Effective risk management should therefore complement PM_{2.5} mass monitoring with source-oriented control of toxic metals, particularly As and Cr(VI), together with improved management of industrial emissions and non-exhaust traffic sources. Continued spatially resolved monitoring and metal speciation are also recommended to better characterize localized exposure and support targeted environmental health protection for nearby communities.

Declarations

Ethics approval and consent to participate

Not applicable.

Consent for publication

Not applicable.

Competing Interests

The authors have no relevant financial or non-financial interest to disclose.

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Author Contribution

Dr. Tser-Sheng Lin contributed to the study conception, design and supervision. The air sample collection, pretreatment as well as quantitation were carried out by Dr. Tzu-Ting Yang. Data analysis was performed by Miss Hisaoyun Chiou. The

manuscript was prepared by Dr. Tser-Sheng Lin and all authors commented on previous versions of the manuscript. All authors approved the final manuscript.

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Availability of data and material

All data are available based on reasonable request.

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Figures

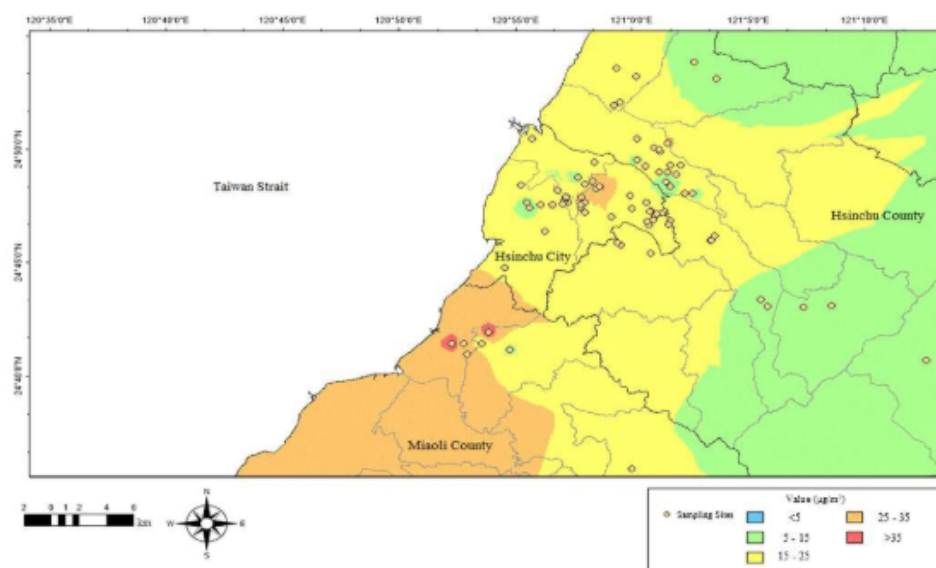


Figure 1

Sampling sites and ambient spatial distribution of PM_{2.5} in a high-tech area.

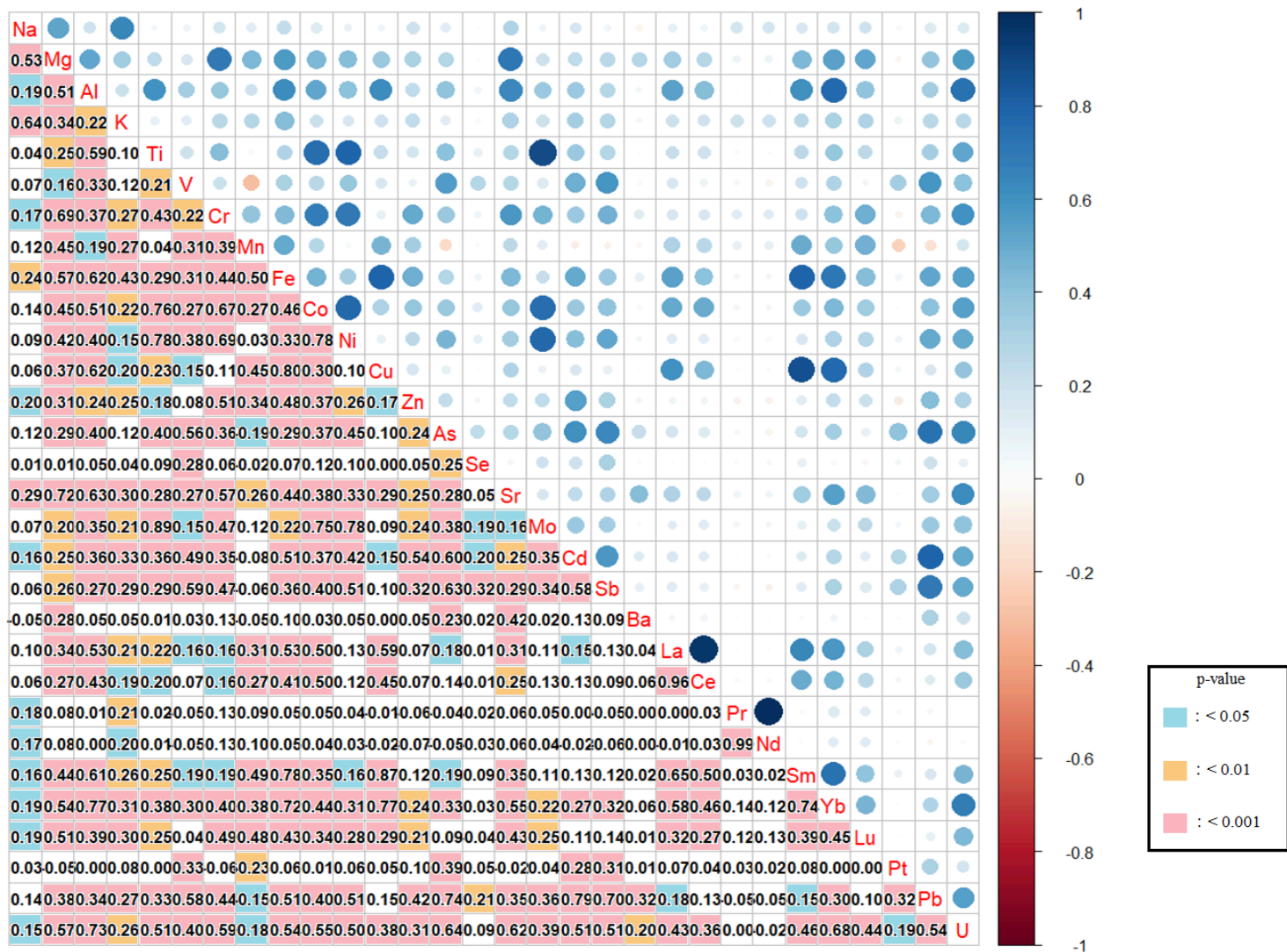


Figure 2
Pearson correlation between ambient particulate metals.

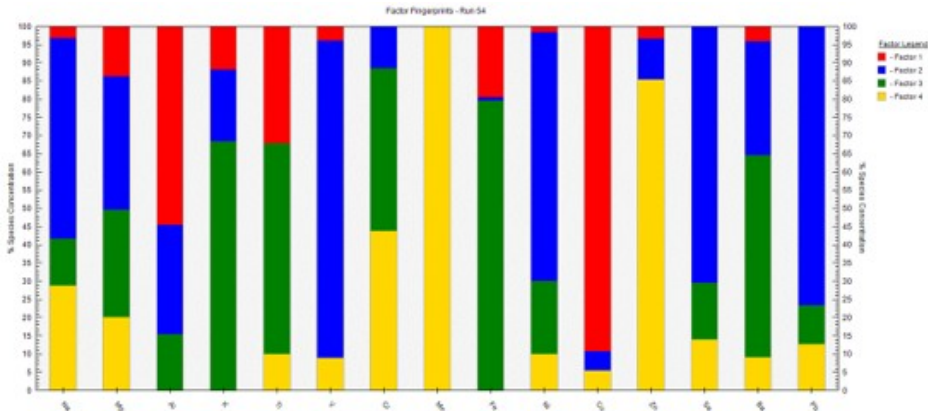


Figure 3
The contribution of different PM_{2.5} sources to ambient particulate metals.

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