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Inhalation Health Risks of $PM_{2.5}$-Bound Metals among Urban Commuters: Evidence from Personal Exposure Monitoring in Bangkok, Thailand

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Abstract— *Urban commuters are increasingly exposed to fine particulate matter ($PM_{2.5}$) enriched with toxic metals originating from non-exhaust traffic emissions and mechanical abrasion. However, evidence based on personal exposure across diverse public transport microenvironments in tropical megacities remains limited. This study conducted personal exposure monitoring of $PM_{2.5}$ and $PM_{2.5}$-bound metals, including Mn, Fe, Cu, Zn, Cd, and Pb, across five public transport modes in Bangkok, Thailand, namely air-conditioned buses, non-air-conditioned buses, taxis, the BTS Skytrain, and the MRT subway, during both wet and dry seasons. $PM_{2.5}$ mass concentrations were determined gravimetrically, and metal concentrations were quantified using ICP–OES following U.S. EPA and ISO/IEC 17025 aligned analytical protocols. Mean $PM_{2.5}$ concentrations were elevated across all transport modes and increased substantially during the dry season. The highest exposures were consistently observed in non-air-conditioned bus, reaching up to 164.5 μg/m³, whereas rail-based systems exhibited comparatively lower $PM_{2.5}$ mass concentrations. Metal composition showed clear mode-specific patterns. Road-based transport was dominated by Zn, Cd, and Pb, reflecting the influence of resuspension and non-exhaust traffic emissions, while rail systems were enriched in Fe, Mn, and Cu, indicating abrasion-related sources. Correlation analysis identified two distinct metal clusters associated with sources. The first cluster was strongly associated with $PM_{2.5}$ mass and included Zn, Cd, and Pb, with correlation coefficients up to 0.934. The second cluster consisted of Mn, Fe, and Cu and was indicative of mechanical abrasion processes, with inter-metal correlations reaching 0.946. Inhalation health risks were evaluated using the U.S. EPA risk assessment framework. Carcinogenic risk, expressed as incremental lifetime cancer risk for Cd and Pb, ranged from 1.35×10^{-7} to 7.95×10^{-7} and was highest in road-based bus environments, with cadmium contributing approximately 85–95% of the total cancer risk. Non-carcinogenic risk remained below the threshold of concern, with hazard index values ranging from 0.062 to 0.206, but consistently higher values were observed in bus environments driven mainly by Mn and Cd. Although all estimated risks were within commonly accepted screening benchmarks, pronounced intermodal and seasonal gradients indicate disproportionate exposure burdens in road-based public transport, particularly non-air-conditioned buses. These findings highlight the need for targeted mitigation strategies that focus on improved cabin filtration and ventilation, dust resuspension control, and the optimization of brake and rail materials and maintenance practices in tropical megacities.*

Keywords— *Commuter exposure; Transport microenvironments; $PM_{2.5}$-bound metals; Traffic emissions; Health risk assessment.*

I. INTRODUCTION

Air pollution is widely recognized as a serious public health challenge, responsible for about 4.2 million premature deaths globally each year and significantly contributing to the burden of cardiovascular, respiratory, and neurological diseases [1],[2]. Fine particulate matter ($PM_{2.5}$; aerodynamic diameter ≤ 2.5 μm) is among the most dangerous air pollutants because it can penetrate deep into the lower respiratory tract, enter the bloodstream, and cause multi-organ toxicity. Evidence from epidemiological studies repeatedly shows that both short- and long-term exposure to $PM_{2.5}$ increases the risk of ischemic heart disease, stroke, chronic obstructive pulmonary disease, and cognitive decline, even at levels below current regulatory standards [3], [4], [5]. These health impacts disproportionately affect urban populations in low- and middle-income countries, where exposure levels remain high, and mitigation efforts are often limited.

In Southeast Asia, rapid urban growth and increased motor vehicle use have significantly increased residents' exposure to traffic-related air pollution, particularly among commuters who spend extended periods in transportation environments. Over 650 million people in Asian megacities rely daily on public transit systems that often operate in congested traffic conditions, characterized by high emissions, poor ventilation, and limited air quality controls [6], [7]. Bangkok, Thailand, exemplifies this public health concern. Chronic traffic congestion, diverse vehicle fleets mainly consisting of diesel buses, and frequent atmospheric stagnation under tropical climate conditions lead to consistently high $PM_{2.5}$ levels. Both ambient and in-transit $PM_{2.5}$ concentrations in Bangkok regularly surpass the U.S. Environmental Protection Agency (EPA) 24-hour standard (35 μg/m³) and the WHO (2021) Air Quality Guideline (15 μg/m³), with peak levels during rush hours in transportation corridors often exceeding 100–150 μg/m³ [8]. Such high exposure levels pose significant health risks to daily commuters, especially low-income populations who mainly rely on public buses with limited filtration and ventilation.

From a public health perspective, however, assessing the health risk from $PM_{2.5}$ exposure cannot rely solely on mass concentration. Increasing evidence suggests that the toxicity of

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$PM_{2.5}$ is heavily influenced by its chemical makeup, particularly the presence of redox-active and toxic trace metals. In transportation microenvironments, $PM_{2.5}$ often contains enriched levels of iron (Fe), manganese (Mn), copper (Cu), zinc (Zn), cadmium (Cd), and lead (Pb), mainly coming from non-exhaust emissions such as brake and tire wear, lubricant oil residues, and resuspended road dust, rather than just tailpipe exhaust (Knibbs et al., 2018; Zhang et al., 2021). Non-exhaust sources contribute heavily to Zn-, Cd-, and Pb-rich particles in road transport, while rail systems generate $PM_{2.5}$ dominated by Fe-, Mn-, and Cu-rich particles from wheel–rail contact, braking, and electrical abrasion.

This source-specific metal profile is crucial for public health risk assessments. Transition metals like Fe and Cu can drive the formation of reactive oxygen species, leading to oxidative stress and inflammation, while chronic inhalation of Cd and Pb is linked to kidney damage, neurotoxicity, and cancer. Long-term exposure to Mn-containing particles has also been associated with adverse neurological effects, even at relatively low concentrations typical of daily commuting environments [9]. Therefore, analyzing the metal composition of $PM_{2.5}$ is vital for identifying dominant sources, understanding exposure pathways, and developing targeted strategies to reduce health risks for commuters.

Despite the recognition of non-exhaust emissions as a major component of urban $PM_{2.5}$, research from public transportation microenvironments remains limited and scattered. Many previous studies have depended on ambient or roadside monitoring, short-term measurements, or focused on single transport modes, limiting their relevance to real-world commuter exposures. Moreover, inconsistencies in methodology, including differences in sampling design, analytical methods, metal digestion procedures, and inadequate quality assurance and quality control (QA/QC), have hindered comparisons across studies and reduced their policy impact [10],[11]. Few investigations have combined personal exposure monitoring, detailed metal speciation, and standardized QA/QC frameworks aligned with ISO 17025 and U.S. EPA standards, particularly in tropical megacities where exposure dynamics differ from those in temperate areas.

Bangkok offers a fitting case study to address these gaps. The city features a variety of transportation environments, from open-window buses and air-conditioned buses to enclosed electric rail systems such as the BTS Skytrain and MRT. These diverse microenvironments provide an excellent opportunity to distinguish non-exhaust traffic emissions from rail-specific mechanical wear through studying $PM_{2.5}$ metal composition under real-world commuting conditions. Understanding these sources is key to designing effective mitigation strategies, including improving vehicle filtration systems, reducing non-exhaust emissions from braking, and optimizing rail maintenance and ventilation.

This study aims to measure personal exposure to $PM_{2.5}$ and related heavy metals (Fe, Mn, Cu, Zn, Cd, and Pb) across five major public transportation modes in Bangkok (air-conditioned bus, non-air-conditioned bus, taxi, BTS, and MRT) during both wet and dry seasons. Besides characterizing concentration levels and metal profiles linked to specific sources, the study will evaluate the potential health risks, both carcinogenic and non-carcinogenic, associated with inhalation of these bound metals, using the U.S. EPA risk assessment framework. By integrating personal measurements, metal speciation, and standardized health risk assessments, the goal is to identify key emission sources driving risks for commuters and to support science-based public health and transportation policies. The findings aim to foster strategies that lower exposure, protect vulnerable populations, and promote healthier, more sustainable urban transportation in Bangkok and other rapidly growing cities worldwide.

II. Materials and Methods

A. Study Design and Sampling Methods

This study employed a cross-sectional personal exposure monitoring design to characterize commuter exposure to fine particulate matter ($PM_{2.5}$) and associated traffic-related pollutants across major public transportation microenvironments in Bangkok, Thailand. The investigation was conducted under real-world commuting conditions, without experimental manipulation, to ensure ecological validity and relevance for public health and transport policy applications.

Personal exposure measurements were carried out across five representative public transport modes that collectively reflect Bangkok's dominant commuting patterns: air-conditioned (A/C) bus, non-air-conditioned (non-A/C) bus, taxis, the Bangkok Mass Transit System (BTS) Skytrain, and the Mass Rapid Transit (MRT) subway. These modes capture a broad spectrum of exposure scenarios, ranging from open or semi-enclosed road-based vehicles with direct traffic influence to fully enclosed, electrically powered rail systems with distinct mechanical emission profiles.

B. Transit Mode Selection Framework

The selection of transport modes was guided by ridership statistics, route coverage, and operational characteristics to ensure representativeness of daily commuter exposure in Bangkok. Road-based modes (A/C bus, non-A/C bus, and taxi) are heavily influenced by traffic congestion, roadside emissions, and ventilation practices, whereas rail-based systems (BTS and MRT) provide physically separated microenvironments dominated by mechanical abrasion.

The inclusion of both A/C and non-A/C buses allowed assessment of the role of ventilation configuration and cabin enclosure in modulating pollutant infiltration and personal exposure. This framework aligns with established transport microenvironment exposure assessment methodologies and facilitates comparison between non-exhaust traffic-related emissions and rail-specific mechanical sources.

C. Route Selection and Spatial Characteristics

A structured route selection protocol was developed to ensure spatial comparability across transport modes while capturing exposure variability within Bangkok's heterogeneous urban landscape. Three primary monitoring routes, Route A, Route B, and Route C, were selected to represent road-based and rail-based commuting corridors traversing densely

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populated commercial, business, and residential districts (Figure 1).

• Route A (approximately 22 km) was assigned to road-based transport modes (A/C bus, non-A/C bus, taxi). It passed through major arterial roads, including Phahonyothin, Phayathai, Rama I, Phloen Chit, and Sukhumvit Roads. This corridor is characterized by high traffic density, multi-lane street-canyon configurations (6–8 lanes), limited dispersion, and severe congestion during peak hours, creating conditions conducive to elevated in-vehicle pollutant exposure.

• Route B, corresponding to the BTS Sukhumvit Line, ran broadly parallel to Route A with comparable start and end points. This alignment enabled direct comparison between elevated rail and on-road transport systems operating within similar urban and land-use contexts.

• Route C, representing the MRT Blue Line, also paralleled Routes A and B. This underground rail corridor extended approximately 22 km with closely spaced stations (~1 km apart), facilitating assessment of exposure in enclosed subterranean environments influenced by rail–wheel interaction, braking, and ventilation dynamics.

Together, the three routes captured two dominant urban typologies: high-density commercial/business zones and residential areas, allowing evaluation of exposure patterns across varying land-use and traffic intensity conditions.

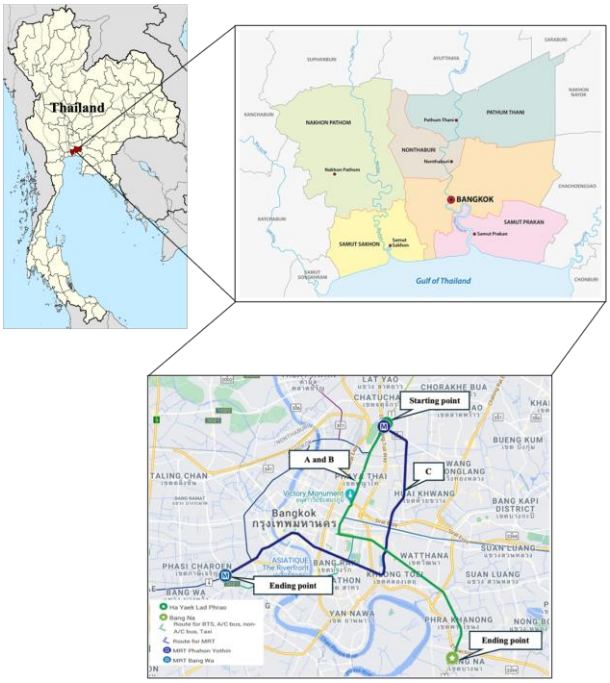

Fig. 1. Map of the study routes. Route A (green line) covers A/C buses, non-A/C buses, and taxis. Route B (green line) represents the Sukhumvit Line of the elevated train (BTS), while Route C (blue line) corresponds to the MRT Blue Line of the underground metro (MRT)

D. Temporal and Seasonal Sampling Framework

To reflect typical commuter behavior and capture worst-case exposure scenarios, sampling was conducted during peak commuting periods, defined as morning rush hours (07:00–09:00 a.m.) and evening rush hours (5:00–7:00 p.m.). These periods correspond to maximum passenger density, elevated vehicle emissions, and reduced ventilation efficiency in transport microenvironments.

Given Bangkok's monsoonal climate, monitoring was conducted during two distinct seasonal campaigns to account for meteorological influences on pollutant levels and dispersion:

• Wet season: June–July 2022 (southwest monsoon)

• Dry season: January–February 2023 (northeast monsoon)

The dry season is characterized by lower precipitation, frequent atmospheric stagnation, and reduced pollutant washout, making it particularly relevant for capturing peak $PM_{2.5}$ exposure. Ambient temperatures during sampling ranged from 27–33 °C, with relative humidity between 65–80%, and no rainfall was recorded during monitoring sessions.

Sampling was performed simultaneously across all five transport modes on matched routes to minimize temporal bias and facilitate intermodal comparison. Each seasonal campaign included measurements on both weekdays and weekends (four days per season), capturing variability associated with commuting patterns and traffic intensity.

E. Characteristics Personal Exposure Monitoring Procedures

Personal exposure monitoring was conducted by trained field researchers traveling as regular passengers without notifying drivers or operators, thereby maintaining typical operating conditions and avoiding behavioral modification. Sampling equipment was positioned within the breathing zone of the participant, and all measurements were conducted while seated away from doors and windows to minimize localized disturbances.

$PM_{2.5}$ mass concentration, $PM_{2.5}$-bound metals, and BTEX were measured concurrently across all transport modes. To ensure sufficient particulate mass for chemical analysis, $PM_{2.5}$ samples collected during morning and evening peak periods were composited into a single daily sample for each mode. All instruments were pre-calibrated prior to deployment, and standardized protocols were applied across routes and modes to ensure methodological consistency and data comparability.

Overall, this study design integrates spatially and temporally resolved personal exposure monitoring across diverse transport microenvironments, enabling robust comparison of $PM_{2.5}$ concentrations, metal composition, and source-related characteristics. The approach provides a strong methodological foundation for subsequent source apportionment and health risk assessment, supporting evidence-based public health and transport policy interventions in Bangkok and other rapidly urbanizing cities.

F. Sample Analysis for $PM_{2.5}$ Mass

$PM_{2.5}$ mass was determined gravimetrically following NIOSH Method 0600 [12] and the U.S. EPA Compendium Method for particulate mass determination. $PM_{2.5}$ was collected on 37-mm PTFE membrane filters (pore size 2.0 μm). Prior to weighing, filters were visually inspected with non-metallic forceps and equilibrated in a controlled weighing environment at 25 ± 2 °C and 45 ± 5% relative humidity for at least 24 h both before and after sampling to minimize hygroscopic mass variability.

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Filter weights were measured using a high-precision microbalance (METTLER TOLEDO XP6, readability 1 μg). Each filter was weighed in triplicate at each stage (pre- and post-sampling), and the mean values were recorded as the pre-sampling mass W_1 and post-sampling mass W_2. Field blanks (handled identically to samples but without air collection) were included in each sampling round and used to correct for potential contamination and handling-related mass changes. The $PM_{2.5}$ mass concentration was calculated as:

$$C_{PM2.5} = \frac{(W_2 - W_1) - (B_2 - B_1)}{V} \qquad (1)$$

where $C_{PM2.5}$ is the $PM_{2.5}$ concentration (μg/m³), W_1 and W_2 are the mean pre- and post-sampling masses of sample filters (μg), B_1 and B_2 are the mean pre- and post-sampling masses of blank filters (μg), and V is the sampled air volume (m³). The sampled volume was computed from the calibrated flow rate and sampling duration:

$$V = \frac{Q \times T}{1000} \qquad (2)$$

where Q is the average flow rate (L·min⁻¹) and T is the sampling time (min); 1000 converts liters to cubic meters. Flow rate was calibrated before and after sampling, and samples with flow deviation exceeding ±5% were flagged for quality review. All weighing procedures were conducted following consistent static control and balance QA checks to ensure precision and reproducibility.

G. Heavy Metals in $PM_{2.5}$ Analysis

$PM_{2.5}$-bound trace metals were quantified for Mn, Fe, Cu, Zn, Cd, and Pb using microwave-assisted acid digestion based on U.S. EPA Method 3051A [13], followed by ICP–OES determination. Each PTFE filter was cut into small pieces using clean, non-metallic tools and transferred into pre-cleaned Teflon digestion vessels. Ten milliliters of 10% (v/v) ultrapure nitric acid (HNO_3) was added to each vessel under a fume hood, the vessels were sealed, and digestion was performed at 175 °C and 1000 W for 30 min. After cooling to room temperature, digests were filtered through Whatman No. 42 filter paper and diluted to a final volume of 50 mL using ultrapure deionized water (resistivity ≥ 18.2 MΩ·cm). Digested extracts were stored at 4 °C in acid-washed polyethylene bottles until instrumental analysis.

Metal concentrations were measured using ICP–OES (e.g., Agilent 5110) operated under optimized conditions (plasma power ~1300 W; argon plasma gas flow 15 L/min¹; auxiliary flow 0.2 L/min¹; nebulizer flow 0.8 L/min¹; sample uptake 1.5 mL/min¹). Analytical wavelengths were selected as follows: Mn (257.610 nm), Fe (259.940 nm), Cu (324.754 nm), Zn (213.856 nm), Cd (226.802 nm), and Pb (220.353 nm). Multi-point external calibration was conducted using traceable multi-element standards, and calibration acceptance required linearity with $R^2 \geq 0.995$ (typically ≥ 0.999). Instrument stability was verified using continuing calibration verification standards and procedural blanks. Metal concentrations in air were calculated from the measured solution concentration:

$$C_{metal} = \frac{C_s \times V_t}{V_a} \qquad (3)$$

where C_{metal} is the metal concentration in air (μg·m⁻³), C_s is the metal concentration in the digested solution (mg·L⁻¹), V_t is the final digest volume (L), and V_a is the sampled air volume (m³). The sampled air volume was computed as:

$$V_a = \frac{Q \times T}{1000} \qquad (4)$$

where Q is the average calibrated sampling flow rate (L·min⁻¹) and T is the sampling duration (min). Final concentrations originally obtained in ng/m³ were converted to μg/m³ where required for subsequent health risk assessment.

H. Quality Assurance and Quality Control (QA/QC)

A quality assurance and quality control (QA/QC) program was implemented throughout sampling, sample handling, and laboratory analysis to ensure data accuracy, precision, and reproducibility. All sampling equipment, digestion vessels, and consumables were acid-washed and pre-cleaned to minimize contamination, and sample preparation was conducted under controlled laboratory conditions.

Field blanks and laboratory blanks were included in each sampling and analytical batch and processed identically to samples. Blank values were subtracted from measured concentrations prior to calculation. For gravimetric $PM_{2.5}$ analysis, filters were conditioned under controlled temperature and humidity, repeatedly weighed until mass stability was achieved, and balance performance was routinely verified. Samples with flow-rate deviations exceeding ±5% were excluded, consistent with U.S. EPA gravimetric criteria.

For metal analysis, procedural blanks were digested and analyzed alongside samples. Instrument calibration was conducted using multi-point external standards, and continuing calibration verification standards were analyzed periodically to confirm instrumental stability. Analytical runs were accepted only when calibration linearity and verification results met U.S. EPA-recommended acceptance criteria.

Method detection limits (MDLs) were determined as three times the standard deviation (3σ) of blank measurements, while limits of quantification (LOQs) were defined as ten times the standard deviation (10σ), following U.S. EPA guidance. Only concentrations exceeding the LOQ were used for quantitative interpretation and health risk assessment.

Analytical precision was assessed through replicate analyses, and results were accepted when agreement fell within laboratory quality-control limits. Method accuracy was evaluated using spike recovery tests, with acceptable recoveries defined as 80–120% in accordance with U.S. EPA and ISO/IEC 17025 guidelines. All analyses were performed in laboratories operating under ISO/IEC 17025–aligned quality systems, ensuring the reliability of $PM_{2.5}$ mass and $PM_{2.5}$-bound metal data for subsequent source characterization and health risk assessment.

I. Health Risk Assessment of $PM_{2.5}$-Bound Heavy Metals

To evaluate potential non-carcinogenic and carcinogenic health risks from commuter inhalation exposure to $PM_{2.5}$-bound heavy metals, this study applied the U.S. Environmental Protection Agency (U.S. EPA) inhalation risk assessment framework. Risk characterization was conducted separately for

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(i) non-carcinogenic effects using Hazard Quotient (HQ) and Hazard Index (HI) and (ii) carcinogenic effects using excess lifetime cancer risk (CR), following standard EPA guidance for chronic inhalation exposure assessment [14].

Exposure Concentration (EC) for Commuter Inhalation

Personal exposure measurements yielded metal-specific air concentrations (C_A) in $\mu g/m^3$ (converted from ng/m^3 as needed). Long-term inhalation exposure was represented as an exposure concentration (EC), which normalizes exposure duration and frequency over an averaging time:

$$EC = C_A \times \frac{ET \times EF \times ED}{AT} \qquad (5)$$

Where:

- EC= exposure concentration ($\mu g/\ m^3$)
- C_A= measured metal concentration in air ($\mu g/\ m^3$)
- ET= exposure time (hours/day)
- EF= exposure frequency (days/year)
- ED= exposure duration (years)
- AT= averaging time (hours)

The exposure parameters used to determine health risk assessment are presented in Table I.

TABLE I. Exposure parameters used for inhalation health risk assessment (commuter scenario)

Parameter	Value	Reference / Rationale
CA	Measured concentration (μg/m³)	Personal exposure monitoring (this study)
ET	Average commuting duration (h/day)	Field monitoring (this study)
ET	250 days/year (working days, excluding holidays)	[15],[16]
ED	45 years (average adult working lifetime)	[16],[17]
AT_{nc}	394,200 h (ED × 365 × 24)	[17]
AT_{ca}	613,200 h (70 × 365 × 24)	[17]

Note: Non-carcinogenic effects: AT_{nc}=ED×365×24 (hours), Carcinogenic effects: AT_{ca}=70×365×24 (hours), assuming a 70-year lifetime

Non-Carcinogenic Risk Characterization (HQ and HI)

Non-carcinogenic risk for each metal was assessed using the Hazard Quotient (HQ):

$$HQ_i = \frac{EC_i}{RfC_i} \qquad (6)$$

Where:

- HQ_i= hazard quotient for metal i(unitless)
- EC_i= exposure concentration for metal $i(\mu g/\ m^3)$
- RfC_i= inhalation reference concentration for metal $i(\mu g\ /m^3)$

Cumulative non-carcinogenic risk from multi-metal exposure was evaluated using the Hazard Index (HI):

$$HI = \sum HQ_i \qquad (7)$$

Interpretation followed EPA conventions: HQ or HI ≤ 1.0 indicates that adverse non-carcinogenic effects are not expected, while values > 1.0 suggest potential concern and warrant closer evaluation (e.g., refinement of exposure assumptions, consideration of susceptibility, and uncertainty analysis).

Carcinogenic Risk Characterization (CR)

For metals with available inhalation carcinogenic potency values, lifetime excess cancer risk was estimated using the inhalation unit risk (IUR) approach:

$$ILCR_i = EC_i \times IUR_i \qquad (8)$$

Where:

- $IL\,CR_i$ = incremental lifetime cancer risk for metal i(unitless probability)
- IUR_i= inhalation unit risk for metal i

Carcinogenic risks were interpreted using widely adopted threshold values. An ILCR of ≤ 1×10^{-6} was considered an acceptable risk level; values between 1×10^{-6} and 1×10^{-4} were classified as tolerable; and ILCR values > 1×10^{-4} were regarded as unacceptable, indicating a definite carcinogenic risk [15]; [18].

The corresponding RfC and IUR values adopted in this study are summarized in Table II, which were obtained from authoritative databases such as U.S. EPA, WHO, IRIS, and relevant peer-reviewed literature. These parameters served as the foundation for calculating the Hazard Quotient (HQ), Hazard Index (HI), and Carcinogenic Risk (CR) in subsequent risk estimation procedures.

TABLE II. Response parameters of elements entering through the respiratory system

Element	RfC (mg/m³)	IUR ((μg/m³) ⁻¹)	IARC Classification
Mn	5.0×10^{-5}[19]	—	Group 2B
Fe	7.0×10^{-1}[20]	—	Group 3
Cu	4.0×10^{-2}[20]	—	Group 3
Zn	3.0×10^{-1}[20]	—	Group 3
Pb	3.5×10^{-3}[20]	1.2×10^{-5}[a][19]	Group 2A
Cd	1.0×10^{-5}[20]	1.8×10^{-3}[a][19]	Group 1

J. Statistical Analysis

Statistical analyses were performed to examine differences in pollutant concentrations and exposure characteristics across transportation modes and seasons, as well as to explore relationships among $PM_{2.5}$ and $PM_{2.5}$-bound trace metals. All statistical analyses were conducted using IBM SPSS Statistics (version 29.0.2), with statistical significance defined at $p < 0.05$.

Descriptive statistics, including arithmetic mean, standard deviation, median, and percentile values, were calculated separately by transport mode and season to summarize central tendency and variability in exposure levels. Before inferential analysis, data distributions were evaluated for normality and homogeneity of variance using standard diagnostic tests. When necessary, appropriate data transformation or robust statistical interpretation was applied.

Differences in pollutant concentrations among transport modes were assessed using one-way analysis of variance (ANOVA), followed by post-hoc multiple comparison tests to identify specific intermodal differences. Seasonal differences between the wet and dry seasons were evaluated using independent-samples t-tests, enabling comparison of exposure levels under contrasting meteorological conditions.

To investigate associations between $PM_{2.5}$ mass concentrations and individual trace metals, Pearson correlation coefficients were calculated. Correlation patterns were

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interpreted to infer potential common emission sources and source groupings, with particular attention to distinguishing non-exhaust traffic-related emissions (e.g., brake and tire wear, road dust resuspension) from rail-related mechanical abrasion processes. These statistical relationships supported subsequent interpretation of source contributions and their implications for commuter exposure and health risk assessment.

III. Results and Discussion

A. $PM_{2.5}$ concentrations across transport modes and seasons

A total of 80 personal $PM_{2.5}$ samples were collected across five transport modes (16 samples per mode; eight per season). Summary statistics are presented in Figure 4.1, which illustrates the intermodal and seasonal contrasts in mean $PM_{2.5}$ concentrations.

Across all transport microenvironments, $PM_{2.5}$ concentrations were consistently elevated, with mean values during the wet season ranging from 31.1 to 109.5 μg/m³ and increasing markedly during the dry season to 54.3–164.5 μg/m³. The overall mean $PM_{2.5}$ concentration increased from 67.2 ± 36.8 μg/m³ in the wet season to 103.8 ± 44.1 μg/m³ in the dry season, representing an approximate 54% seasonal increase. This seasonal pattern reflects reduced atmospheric dispersion, limited wet deposition, and enhanced resuspension of road dust under dry meteorological conditions.

Pronounced differences were observed across transport modes. Non-A/C buse consistently exhibited the highest $PM_{2.5}$ concentrations in both seasons (109.5 ± 13.2 μg/m³ in the wet season; 164.5 ± 15.4 μg/m³ in the dry season), attributable to open-window ventilation that allows direct infiltration of roadside air. Air-conditioned (A/C) buses also showed elevated concentrations (77.0 ± 10.7 μg/m³ wet; 121.4 ± 17.7 μg/m³ dry), likely influenced by frequent door opening and limited filtration efficiency. Taxi commuters experienced intermediate exposure levels (70.3 ± 4.7 μg/m³ wet; 78.7 ± 17.5 μg/m³ dry), reflecting partial enclosure and variable ventilation and filter performance.

In contrast, substantially lower $PM_{2.5}$ concentrations were observed in rail-based transport systems. The BTS Skytrain recorded the lowest levels (31.1 ± 5.0 μg/m³ wet; 54.3 ± 8.8 μg/m³ dry), followed by the MRT subway (47.9 ± 9.4 μg/m³ wet; 73.9 ± 16.5 μg/m³ dry). These systems operate in electrically powered, physically separated, and enclosed environments, which limit direct exposure to roadway emissions and contribute to more stable in-cabin air quality.

All measured mean $PM_{2.5}$ concentrations across transport modes and seasons exceeded the WHO 24-hour Air Quality Guideline (15 μg·m⁻³) and the revised Thai National Ambient Air Quality Standard (37.5 μg/m³), underscoring the potential public health implications for daily commuters. Dry-season concentrations were approximately 1.4–1.6 times higher than wet-season values across modes, indicating that seasonal meteorology amplifies, but does not override, the influence of transport microenvironment characteristics.

Statistical analysis confirmed significant intermodal differences in $PM_{2.5}$ concentrations (one-way ANOVA, $p < 0.05$). Post hoc Tukey HSD tests demonstrated that non-A/C buses had significantly higher concentrations than all other modes, while BTS concentrations were significantly lower than those of all road-based modes ($p < 0.05$). These findings indicate that transport mode and microenvironmental characteristics exert a stronger influence on commuter $PM_{2.5}$ exposure than seasonal variation alone.

Overall, the results highlight road-based public transport, particularly non-A/C bus, as a critical exposure microenvironment for $PM_{2.5}$ in Bangkok. The clear contrast between road-based and rail-based systems emphasizes the importance of improving in-vehicle filtration and ventilation in buses and taxis, alongside expanding low-emission rail transit, as effective strategies to reduce commuter inhalation exposure and associated health risks.

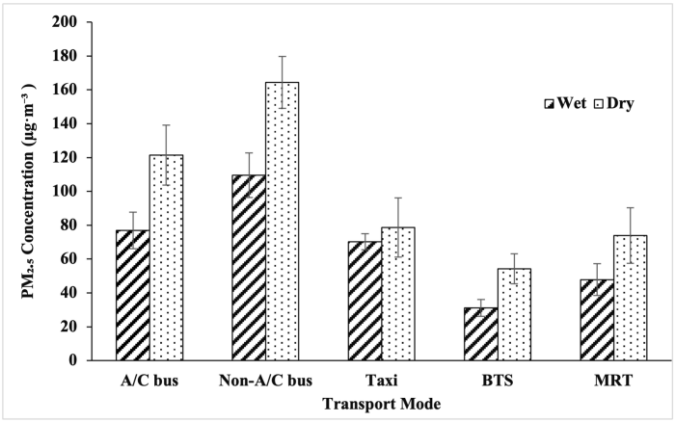

Fig. 2. Mean (±SD) concentrations of $PM_{2.5}$ (μg·m⁻³) by transport mode during wet and dry seasons

B. Trace Metals in $PM_{2.5}$ across Transport Modes

The concentrations of $PM_{2.5}$-bound trace metals (Mn, Fe, Cu, Zn, Cd, and Pb) varied substantially across transport modes and seasons, reflecting differences in emission sources, ventilation characteristics, and microenvironmental conditions (Figure 3). Across all modes, metal concentrations were consistently higher during the dry season, coinciding with elevated $PM_{2.5}$ levels, reduced wet deposition, enhanced resuspension of road dust, and intensified mechanical abrasion under dry meteorological conditions.

Iron (Fe) was the most abundant metal across all transport modes and seasons, accounting for the largest fraction of $PM_{2.5}$-bound metal mass. Rail-based systems exhibited particularly elevated Fe concentrations, with the highest levels observed in the MRT during the dry season (3,003 ± 341 ng/m³), followed by the BTS (2,357 ± 408 ng/m³). This enrichment reflects dominant contributions from wheel–rail contact, braking systems, and steel component wear in enclosed rail environments. In contrast, road-based modes showed lower but still substantial Fe levels, attributable to a combination of brake wear, engine-related abrasion, and resuspended crustal material. Taxi commuters experienced the lowest Fe exposure, consistent with reduced mechanical wear intensity and partial cabin isolation.

Zinc (Zn) was the second most abundant metal and displayed pronounced contrasts between road-based and rail-based transport. The highest Zn concentrations were consistently recorded in non–A/C buses, increasing from 397 ±

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20 ng/m³ in the wet season to 564 ± 46 ng/m³ in the dry season. Elevated Zn levels were also observed in A/C buses and taxis, reflecting contributions from tire wear, lubricant oil additives, and resuspended roadway dust. In comparison, BTS and MRT showed substantially lower Zn concentrations, indicating limited influence from external traffic emissions and a greater dominance of internal mechanical sources.

Copper (Cu) exhibited a mode-dependent pattern similar to Zn but with a stronger signal from mechanical abrasion. The highest Cu concentrations were observed in rail-based systems, particularly the MRT during the dry season (407 ± 42 ng/m³), followed by BTS and non–A/C buses. Seasonal increases of approximately 30–50% across all modes highlight the sensitivity of Cu emissions to braking intensity and dry surface conditions. Taxi cabins exhibited comparatively lower Cu concentrations, consistent with reduced braking frequency and greater use of air recirculation.

Manganese (Mn) enrichment was most pronounced in rail-based transport modes. Both BTS and MRT showed significantly higher Mn concentrations than road-based systems, particularly during the dry season, reflecting emissions from steel alloy wear and wheel–rail friction. In contrast, Mn concentrations in buses and taxis were lower and more closely associated with resuspended road dust and vehicular wear. These patterns reinforce Mn as a tracer of rail-related mechanical abrasion in urban transport microenvironments.

Although present at lower absolute concentrations, lead (Pb) and cadmium (Cd) were consistently detected across all transport modes. Road-based systems, especially non–A/C bus, recorded the highest Pb and Cd concentrations, with dry-season levels nearly doubling those observed during the wet season. These increases are indicative of enhanced contributions from legacy road dust contamination, lubricant oil residues, and resuspended industrial particulates. Despite contributing less than 5% of total metal mass, Pb and Cd are of particular public health concern due to their toxicity, persistence, and potential carcinogenicity.

Overall, the intermodal hierarchy of $PM_{2.5}$-bound metal exposure followed the order non–A/C bus > A/C bus > MRT ≈ BTS > taxi, while metal composition clearly distinguished road-based systems dominated by Fe–Zn–Cu assemblages from rail-based systems enriched in Fe–Mn–Cu, characteristic of mechanical abrasion sources. These findings demonstrate that commuter exposure to toxic metals in Bangkok is governed by a combination of ventilation efficiency, proximity to emission sources, and seasonal meteorology.

C. Correlation Analysis and Source-Oriented Interpretation of $PM_{2.5}$-Bound Metals

Pearson's correlation analysis was conducted to examine relationships between $PM_{2.5}$ mass concentration and associated trace metals across all transport microenvironments (Table III). The results revealed two distinct metal groupings, reflecting different dominant emission processes influencing commuter exposure. The first group comprised Zn, Cd, and Pb, which showed strong and statistically significant positive correlations with $PM_{2.5}$ mass concentration. Zn exhibited the strongest association with $PM_{2.5}$ ($r = 0.934$, $p < 0.001$), followed by Cd ($r = 0.777$, $p < 0.001$) and Pb ($r = 0.614$, $p < 0.001$). These elements are widely recognized as tracers of non-exhaust traffic-related sources, including lubricant oil additives, tire and brake wear, and resuspended road dust. The strong coupling between $PM_{2.5}$ and these metals indicates that surface-related emissions and resuspension processes are major contributors to in-cabin $PM_{2.5}$ mass, particularly in road-based transport modes.

In contrast, Fe, Mn, and Cu showed weak or non-significant correlations with $PM_{2.5}$ mass (e.g., Fe: $r = -0.200$, $p = 0.075$; Cu: $r = -0.050$, $p = 0.657$; Mn: $r = 0.176$, $p = 0.119$), but were strongly correlated among themselves, forming a second source cluster. Notably, Fe–Cu exhibited an exceptionally strong correlation ($r = 0.946$, $p < 0.001$), while Mn–Fe ($r = 0.840$, $p < 0.001$) and Mn–Cu ($r = 0.883$, $p < 0.001$) also showed robust associations. This pattern is characteristic of mechanical abrasion sources, such as brake wear, wheel–rail contact, and metallic component erosion, which are particularly prominent in rail-based transport systems.

Based on these correlation structures, two source-oriented metal groups were identified: (i) a non-exhaust and resuspension-related group (Zn–Cd–Pb) that largely governs $PM_{2.5}$ mass variability, and (ii) a mechanical abrasion group (Mn–Fe–Cu) representing metal-rich emissions that are relatively independent of $PM_{2.5}$ mass concentration. These findings are consistent with previous urban transport exposure studies and reinforce the growing evidence that commuter $PM_{2.5}$ exposure in modern cities is increasingly driven by non-exhaust and mechanical processes rather than tailpipe exhaust alone. Overall, the correlation analysis provides mechanistic support for source attribution and underscores the need for mitigation strategies focusing on dust resuspension control, improved cabin filtration, and brake and rail material management, particularly in high-traffic and rail-based transport microenvironments.

D. Health Risk Assessment of $PM_{2.5}$-Bound Metals

Carcinogenic risks

Carcinogenic risks from inhalation exposure to $PM_{2.5}$-bound Cd and Pb were quantified as incremental lifetime cancer risk (ILCR) using compound-specific IUR values. As summarized in Table IV, ILCR values varied systematically across transport modes and seasons, with road-based systems consistently exhibiting higher risks than rail-based systems.

The highest ILCR values were observed in non–A/C buses (4.03×10^{-7} in the wet season; 7.95×10^{-7} in the dry season), followed by A/C buses (3.09×10^{-7}; 5.77×10^{-7}) and taxis (2.24×10^{-7}; 3.74×10^{-7}). In contrast, BTS and MRT showed substantially lower risks, ranging from 1.35×10^{-7} to 1.60×10^{-7} in the wet season and 2.19×10^{-7} to 2.60×10^{-7} in the dry season. Across all modes, ILCR values increased during the dry season, reflecting enhanced resuspension of metal-containing particles and reduced atmospheric dispersion.

Cadmium was the dominant contributor to carcinogenic risk, accounting for approximately 85–95% of total ILCR across all transport modes. For example, in non–A/C buses, Cd contributed 3.81×10^{-7} (wet) and 7.64×10^{-7} (dry), whereas Pb contributed only 2.17×10^{-8} and 3.04×10^{-8}, respectively.

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Although all estimated ILCR values fell within the commonly accepted regulatory range (<1×10^{-6}), the consistently higher risks in road-based bus systems indicate a disproportionate exposure burden for frequent commuters and drivers with prolonged daily exposure.

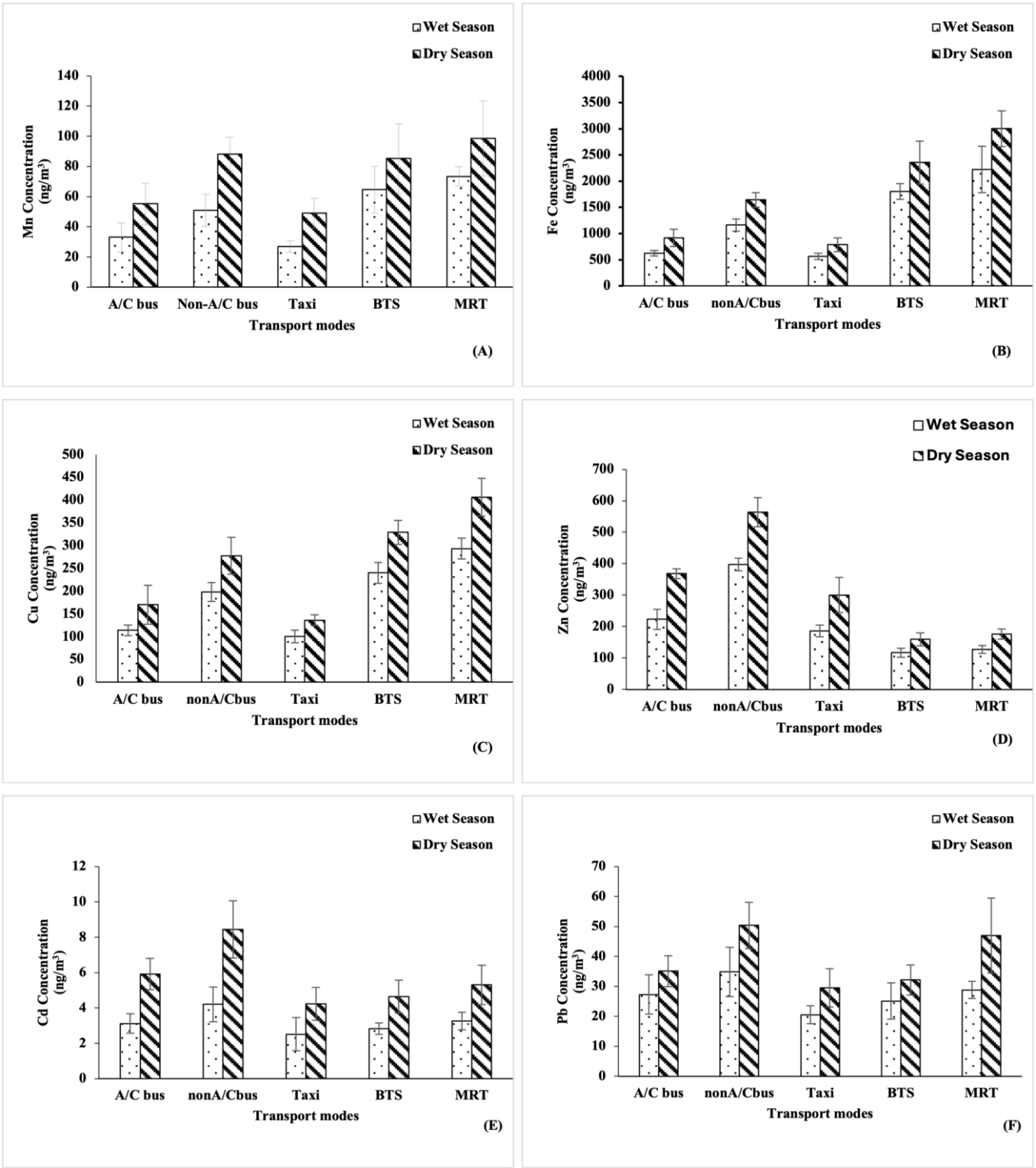

Fig. 3. Heavy metal concentrations in $PM_{2.5}$ across transport modes during wet and dry seasons: (A) Mn concentrations, (B) Fe concentrations, (C) Cu concentrations, (D) Zn concentrations, (E) Cd concentrations, and (F) Pb concentrations

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TABLE III. Pearson's correlation coefficients between $PM_{2.5}$ and $PM_{2.5}$-bound metals (N=80)

Variables	$PM_{2.5}$	Mn	Fe	Cu	Zn	Cd	Pb
$PM_{2.5}$ (r)	1	0.176	−0.200	−0.050	0.934	0.777	0.614
p-value	—	0.119	0.075	0.657	< 0.001*	< 0.001*	< 0.001*
Mn	0.176	1	0.840	0.883	0.090	0.593	0.641
p-value	0.119	—	< 0.001*	< 0.001*	0.429	< 0.001*	< 0.001*
Fe (r)	−0.200	0.840	1	0.946	−0.286	0.273	0.451
p-value	0.075	< 0.001*	—	< 0.001*	0.010	0.014*	< 0.001*
Cu (r)	−0.050	0.883	0.946	1	−0.142	0.383	0.546
p-value	0.657	< 0.001*	< 0.001*	—	0.210	< 0.001*	< 0.001*
Znv(r)	0.934	0.090	−0.286	−0.142	1	0.720	0.524
p-value	< 0.001*	0.429	0.010*	0.210	—	< 0.001*	< 0.001*
Cd (r)	0.777	0.593	0.273	0.383	0.720	1	0.776
p-value	< 0.001*	< 0.001*	0.014*	< 0.001*	< 0.001*	—	< 0.001*
Pb (r)	0.614	0.641	0.451	0.546	0.524	0.776	1
p-value	< 0.001*	< 0.001*	< 0.001*	< 0.001*	< 0.001*	< 0.001*	—

r = Pearson Correlation; * Correlation is significant at the 0.05 level

TABLE IV. Carcinogenic risk (ILCR) of $PM_{2.5}$-bound heavy metals across transport modes

Transport mode	Compounds	EC (μg·m⁻³)		IUR (μg·m⁻³)	ILCR	
		Wet	Dry		Wet	Dry
A/C	Cd	1.62E-04	3.08E-04	1.80E-03	2.92E-07	5.55E-07
	Pb	1.42E-03	1.83E-03	1.20E-05	1.70E-08	2.19E-08
	Sum	**1.58E-03**	**2.14E-03**	**Sum**	**3.09E-07**	**5.77E-07**
non-A/C	Cd	2.12E-04	4.25E-04	1.80E-03	3.81E-07	7.64E-07
	Pb	1.81E-03	2.53E-03	1.20E-05	2.17E-08	3.04E-08
	Sum	**2.02E-03**	**2.96E-03**	**Sum**	**4.03E-07**	**7.95E-07**
Taxi	Cd	1.18E-04	1.99E-04	1.80E-03	2.13E-07	3.57E-07
	Pb	9.64E-04	1.39E-03	1.20E-05	1.16E-08	1.67E-08
	Sum	**1.08E-03**	**1.59E-03**	**Sum**	**2.24E-07**	**3.74E-07**
BTS	Cd	7.06E-05	1.16E-04	1.80E-03	1.27E-07	2.09E-07
	Pb	6.26E-04	8.03E-04	1.20E-05	7.51E-09	9.63E-09
	Sum	**6.97E-04**	**9.19E-04**	**Sum**	**1.35E-07**	**2.19E-07**
MRT	Cd	8.39E-05	1.36E-04	1.80E-03	1.51E-07	2.45E-07
	Pb	7.40E-04	1.21E-03	1.20E-05	8.88E-09	1.45E-08
	Sum	**8.24E-04**	**1.34E-03**	**Sum**	**1.60E-07**	**2.60E-07**

These findings are consistent with previous studies reporting Cd as the primary contributor to $PM_{2.5}$-related cancer risk in urban transport environments, including Hong Kong, Thailand, and [21], [22],[23].

Non-Carcinogenic Risk

Non-carcinogenic risks were assessed using hazard quotients (HQs) for individual metals and the cumulative hazard index (HI). As shown in Table V, all transport modes exhibited HI values well below the threshold of concern (HI ≤ 1), indicating that non-carcinogenic adverse effects are unlikely under typical commuting conditions.

Nevertheless, clear intermodal differences were observed. The highest HI values occurred in non–A/C buses (0.114 in the wet season; 0.206 in the dry season), followed by A/C buses (0.080; 0.139) and taxis (0.058; 0.103). Rail-based systems exhibited lower HI values, ranging from 0.062 to 0.085 for BTS and 0.072 to 0.101 for MRT. These patterns reflect differences in ventilation efficiency and proximity to traffic-related emission sources, with open or semi-open road-based modes allowing greater infiltration of ambient aerosols than enclosed rail systems.

Across all modes, manganese (Mn) and cadmium (Cd) were the dominant contributors to HI due to their relatively low RfC values. In non–A/C buses, Mn accounted for HQ values of 7.96×10^{-2} (wet) and 1.38×10^{-1} (dry), while Cd contributed 3.29×10^{-2} and 6.60×10^{-2}, respectively. Contributions from Fe, Cu, Zn, and Pb were comparatively minor (HQ $\leq10^{-3}$–10^{-4}), reflecting lower toxicological potency at observed concentrations.

Although non-carcinogenic risks remained below regulatory thresholds, the consistently higher HI values in road-based bus systems highlight a relatively greater health burden for commuters relying on these modes. Similar patterns have been reported in other Asian cities, where Mn- and Cd-driven risks were elevated in open or semi-enclosed transport microenvironments [21],[22].

Overall, the health risk assessment indicates that while both carcinogenic and non-carcinogenic risks from $PM_{2.5}$-bound metals remain within acceptable regulatory limits, exposure is unevenly distributed across transport modes. Road-based buses, particularly non–A/C buses, represent priority microenvironments for risk mitigation due to consistently higher $PM_{2.5}$ mass concentrations and metal-related health risks. These findings support targeted interventions such as improved cabin filtration, vehicle fleet modernization, and traffic dust control, which may substantially reduce commuter exposure and associated long-term health risks in Bangkok and similar megacities.

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IV. Discussion

A. $PM_{2.5}$ exposure contrasts across transport microenvironments and seasons

This study demonstrates that commuter exposure to $PM_{2.5}$ in Bangkok is strongly shaped by transport microenvironment characteristics and is amplified during the dry season. The overall mean $PM_{2.5}$ increased from 67.2 ± 36.8 μg/m³ (wet) to 103.8 ± 44.1 μg/m³ (dry), representing an approximate 54% seasonal increment, consistent with reduced wet scavenging and enhanced resuspension under drier conditions. Across modes, the non–A/C bus consistently produced the highest exposures (109.5 ± 13.2 μg/m³ wet; 164.5 ± 15.4 μg/m³ dry), indicating that open-window ventilation and direct proximity to roadway emissions create a high-infiltration environment. In contrast, rail modes exhibited substantially lower $PM_{2.5}$ mass concentrations (BTS: 31.1 ± 5.0 to 54.3 ± 8.8 μg/m³; MRT: 47.9 ± 9.4 to 73.9 ± 16.5 μg/m³), consistent with physical separation from traffic, controlled ventilation, and reduced in-cabin exchange with roadside air. These findings reinforce that, in densely trafficked megacities, microenvironment enclosure and ventilation design can dominate exposure contrasts even when seasonal meteorology shifts background levels.

TABLE V. Non-carcinogenic risk (HQ and HI) associated with $PM_{2.5}$-bound heavy metals across transportation modes during wet and dry seasons.

Transport mode	Compounds	EC (mg/m³)		RfC (mg·m⁻³)	HQ	
		Wet	Dry		Wet	Dry
A/C	Mn	2.68E-06	4.48E-06	0.00005	5.36E-02	8.96E-02
	Fe	5.07E-05	7.42E-05	0.7	7.25E-05	1.06E-04
	Cu	9.23E-06	1.38E-05	0.04	2.31E-04	3.45E-04
	Zn	1.81E-05	2.98E-05	0.3	6.02E-05	9.95E-05
	Cd	2.53E-07	4.79E-07	0.00001	2.53E-02	4.79E-02
	Pb	2.21E-06	2.84E-06	0.0035	6.31E-04	8.12E-04
	Sum	**8.32E-05**	**1.26E-04**	**HI=∑HQ**	**7.99E-02**	**1.39E-01**
non-A/C	Mn	3.98E-06	6.90E-06	0.00005	7.96E-02	1.38E-01
	Fe	9.09E-05	1.28E-04	0.7	1.30E-04	1.83E-04
	Cu	1.55E-05	2.17E-05	0.04	3.87E-04	5.42E-04
	Zn	3.11E-05	4.41E-05	0.3	1.04E-04	1.47E-04
	Cd	3.29E-07	6.60E-07	0.00001	3.29E-02	6.60E-02
	Pb	2.81E-06	3.94E-06	0.0035	8.03E-04	1.13E-03
	Sum	**1.45E-04**	**2.06E-04**	**HI=∑HQ**	**1.14E-01**	**2.06E-01**
Taxi	Mn	1.97E-06	3.57E-06	0.00005	3.93E-02	7.15E-02
	Fe	4.14E-05	5.75E-05	0.7	5.92E-05	8.21E-05
	Cu	7.34E-06	9.93E-06	0.04	1.83E-04	2.48E-04
	Zn	1.36E-05	2.19E-05	0.3	4.52E-05	7.31E-05
	Cd	1.84E-07	3.09E-07	0.00001	1.84E-02	3.09E-02
	Pb	1.50E-06	2.16E-06	0.0035	4.28E-04	6.17E-04
	Sum	**6.60E-05**	**9.54E-05**	**HI=∑HQ**	**5.84E-02**	**1.03E-01**
BTS	Mn	2.51E-06	3.31E-06	0.00005	5.01E-02	6.62E-02
	Fe	7.00E-05	9.15E-05	0.7	9.99E-05	1.31E-04
	Cu	9.32E-06	1.28E-05	0.04	2.33E-04	3.19E-04
	Zn	4.52E-06	6.18E-06	0.3	1.51E-05	2.06E-05
	Cd	1.10E-07	1.81E-07	0.00001	1.10E-02	1.81E-02
	Pb	9.74E-07	1.25E-06	0.0035	2.78E-04	3.57E-04
	Sum	**8.74E-05**	**1.15E-04**	**HI=∑HQ**	**6.18E-02**	**8.50E-02**
MRT	Mn	2.93E-06	3.94E-06	0.00005	5.86E-02	7.87E-02
	Fe	8.88E-05	1.20E-04	0.7	1.27E-04	1.71E-04
	Cu	1.17E-05	1.62E-05	0.04	2.93E-04	4.06E-04
	Zn	5.08E-06	7.02E-06	0.3	1.69E-05	2.34E-05
	Cd	1.31E-07	2.12E-07	0.00001	1.31E-02	2.12E-02
	Pb	1.15E-06	1.87E-06	0.0035	3.29E-04	5.36E-04
	Sum	**1.10E-04**	**1.49E-04**	**HI=∑HQ**	**7.24E-02**	**1.01E-01**

B. Mode-specific metal profiles indicate distinct emission processes

Beyond $PM_{2.5}$ mass, the elemental composition shows clear mode-dependent signatures that reflect different dominant generation mechanisms. Fe was the most abundant metal across all transport modes, but rail environments exhibited particularly high Fe levels, reaching 3,003 ± 341 ng/m³ in MRT (dry) and 2,357 ± 408 ng/m³ in BTS (dry). Such enrichment is consistent with abrasion-related sources typical of rail systems, including wheel–rail contact, braking, and wear of steel components within semi-enclosed platforms and carriages. In contrast, road-based modes displayed lower Fe levels yet higher markers associated with traffic-related non-exhaust processes. For example, Zn peaked in non–A/C buses (397 ± 20 ng/m³ wet; 564 ± 46 ng/m³ dry) and also increased in A/C buses and taxis, supporting contributions from tire wear, lubricant-associated emissions, and resuspended roadway dust. Cu and Mn further differentiated rail and road modes: Cu was highest in MRT (407 ± 42 ng/m³ dry) and BTS, while Mn was also elevated in rail modes, consistent with abrasion of steel alloys and braking systems. Together, these patterns indicate that lower $PM_{2.5}$ mass in rail systems does not necessarily imply lower exposure to metal-rich particles, highlighting the importance of considering both concentration and composition when

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evaluating commuter exposure.

C. Correlation structure supports a two-source clustering consistent with non-exhaust/resuspension and abrasion

The correlation analysis provides mechanistic support for this source-oriented interpretation. $PM_{2.5}$ mass was strongly correlated with Zn ($r = 0.934$, $p < 0.001$), and also with Cd ($r = 0.777$, $p < 0.001$) and Pb ($r = 0.614$, $p < 0.001$), indicating that variability in $PM_{2.5}$ across commuters is largely driven by processes that simultaneously elevate these elements, namely resuspension and non-exhaust traffic emissions. In contrast, the weak/non-significant associations between $PM_{2.5}$ and the abrasion-related metals (e.g., Fe: $r = -0.200$, $p = 0.075$; Cu: $r = -0.050$, $p = 0.657$; Mn: $r = 0.176$, $p = 0.119$) suggest that rail/brake-derived metal emissions can vary independently of total $PM_{2.5}$ mass. The strong intercorrelations within this cluster, particularly Fe–Cu ($r = 0.946$, $p < 0.001$), Mn–Cu ($r = 0.883$, $p < 0.001$), and Mn–Fe ($r = 0.840$, $p < 0.001$), are consistent with a shared mechanical abrasion origin. Importantly, this two-cluster behavior implies that exposure mitigation must address both (i) mass-driving resuspension/non-exhaust processes and (ii) mode-specific abrasion sources that may elevate toxic metals even when $PM_{2.5}$ mass is comparatively lower.

D. Interpretation of inhalation risks: low screening risks but clear mode/season gradients

Risk estimates based on the U.S. EPA inhalation framework indicate that both carcinogenic and non-carcinogenic risks remained within commonly used screening benchmarks, yet display systematic mode- and season-dependent gradients relevant for risk management. For carcinogenic risk from $PM_{2.5}$-bound metals, total ILCR (Cd and Pb) ranged from 1.35×10^{-7} to 7.95×10^{-7}, with road-based modes consistently higher than rail modes. The non–A/C bus showed the highest ILCR (4.03×10^{-7} wet; 7.95×10^{-7} dry), followed by A/C bus (3.09×10^{-7}; 5.77×10^{-7}) and taxi (2.24×10^{-7}; 3.74×10^{-7}), whereas BTS and MRT were lower (1.35×10^{-7}–2.60×10^{-7}). Across all modes, dry-season increments in ILCR align with increased particle loading and enhanced resuspension.

A key finding is that Cd dominated cancer risk, contributing approximately 85–95% of total ILCR across modes (e.g., non–A/C bus Cd: 3.81×10^{-7} wet; 7.64×10^{-7} dry, compared with Pb: 2.17×10^{-8}; 3.04×10^{-8}). This dominance is expected given the relatively high carcinogenic potency used in inhalation unit risk frameworks and the tendency for Cd to be associated with traffic-influenced fine particles. While all ILCR values remained below 10^{-6}, the consistent elevation in bus environments indicates a disproportionate burden for high-frequency commuters and occupational drivers whose time in microenvironments may exceed typical commuter assumptions.

For non-carcinogenic risk, all HI values were well below 1, ranging from 0.062 to 0.206, indicating a low likelihood of adverse effects under standard exposure assumptions. Nonetheless, the hierarchy again highlights road-based buses as priority environments: non–A/C bus (0.114 wet; 0.206 dry) > A/C bus (0.080; 0.139) > taxi (0.058; 0.103) > MRT (0.072; 0.101) ≈ BTS (0.062; 0.085). Across modes, Mn and Cd were the principal drivers of HI because of low RfC values. For example, in the non–A/C bus, Mn HQ increased from 7.96×10^{-2} (wet) to 1.38×10^{-1} (dry), and Cd HQ from 3.29×10^{-2} to 6.60×10^{-2}, whereas Fe, Cu, Zn, and Pb were comparatively minor contributors (generally $\leq10^{-3}$–10^{-4}). This pattern aligns with the observation that non-exhaust and mechanical sources can increase toxic constituents without proportionally increasing $PM_{2.5}$ mass.

E. Comparison with previous studies: consistency in magnitude and dominant contributors

When compared with prior assessments in transport and urban exposure settings, the magnitude of risk in this study falls within established ranges. The observed ILCR values (10^{-7}–10^{-6}) are comparable to urban personal exposure studies reporting Cd-driven cancer risks, including Hong Kong [23] and traffic-influenced environments in Thailand and other Asian cities. Similarly, HI values in this study (0.062–0.206) align with previously reported non-carcinogenic risks for $PM_{2.5}$-bound metals, where Mn and Cd frequently dominate the cumulative index due to their toxicological thresholds. Notably, the upper end of HI observed for Bangkok's bus environments is comparable to settings characterized by strong dust resuspension and traffic proximity, while rail-mode HI values fall closer to ambient-based studies with stronger enclosure and less roadside infiltration.

Beyond magnitude, the dominant contributors in this work, Cd for ILCR and Mn/Cd for HI, are consistent across regions, suggesting that the toxicological profile of commuter metal exposure is controlled less by city-specific tailpipe composition and more by shared urban mechanisms such as resuspension, non-exhaust abrasion, and microenvironment ventilation. Importantly, the present results extend the literature by showing that Bangkok's rail modes, while lower in $PM_{2.5}$ mass, can exhibit metal-rich abrasion signatures (Fe–Mn–Cu), reinforcing the need to interpret "cleaner" transport options using both mass concentration and composition.

F. Implications for exposure control and risk-relevant mitigation

From a risk management perspective, the findings indicate that mitigation should prioritize road-based public transport, particularly non–A/C buses, through interventions that reduce in-cabin infiltration and resuspension (e.g., improved filtration/recirculation, door operation practices, fleet upgrades, and dust control along corridors). For rail systems, strategies targeting abrasion emissions (e.g., brake material optimization, rail/wheel maintenance, platform dust management) may be more relevant than measures aimed solely at reducing $PM_{2.5}$ mass. Because the correlation structure indicates that $PM_{2.5}$ mass is tightly linked to Zn–Cd–Pb, while Mn–Fe–Cu behaves as an abrasion cluster, a single control strategy is unlikely to reduce both exposure dimensions effectively.

Overall, the combined evidence from concentration patterns, elemental signatures, correlation structure, and risk metrics indicates that commuter inhalation exposure in Bangkok is driven primarily by non-exhaust/resuspension processes in road-based modes and by abrasion-related metal enrichment in rail environments, resulting in low screening-level risks but

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clear, actionable mode- and season-specific gradients. These findings directly inform the conclusions regarding priority microenvironments and targeted mitigation opportunities.

V. Conclusions

This study quantified personal exposure to $PM_{2.5}$ and $PM_{2.5}$-bound metals (Mn, Fe, Cu, Zn, Cd, Pb) across five Bangkok transport microenvironments and two seasons. It evaluated inhalation health risks using the U.S. EPA framework. $PM_{2.5}$ concentrations were consistently elevated in all modes and increased markedly during the dry season, with the highest exposure occurring in non–A/C buses and the lowest in rail systems. Metal composition differed systematically by mode, indicating distinct dominant sources: road-based modes showed stronger influence from non-exhaust/resuspension processes (e.g., Zn-, Cd-, Pb-associated), whereas rail systems were enriched in abrasion-related metals (Fe–Mn–Cu), consistent with wheel–rail contact and braking emissions.

Correlation patterns provided source-oriented evidence of two major clusters affecting commuter exposure: (i) a $PM_{2.5}$ mass-related group (Zn–Cd–Pb) strongly coupled with $PM_{2.5}$ (e.g., Zn with $PM_{2.5}$ $r = 0.934$; Cd $r = 0.777$; Pb $r = 0.614$; all $p < 0.001$), suggesting substantial contributions from surface-related non-exhaust emissions and resuspended dust; and (ii) an abrasion-related group (Mn–Fe–Cu) with strong inter-metal correlations (Fe–Cu $r = 0.946$; Mn–Cu $r = 0.883$; Mn–Fe $r = 0.840$; all $p < 0.001$), indicating mechanical wear sources that are comparatively less dependent on $PM_{2.5}$ mass.

Inhalation risk estimates for $PM_{2.5}$-bound metals were below common screening thresholds but showed clear intermodal and seasonal gradients. Carcinogenic risks (ILCR) for Cd and Pb were highest in road-based modes, particularly non–A/C buses, and increased in the dry season; Cd contributed ~85–95% of total ILCR across modes. Non-carcinogenic risks (HI) remained < 1 in all microenvironments, yet were consistently higher in buses driven primarily by Mn and Cd, highlighting disproportionate exposure burdens for routine commuters and potentially for occupationally exposed groups.

From a public health and policy perspective, the results indicate that reducing commuter risk in Bangkok will require strategies beyond tailpipe controls. Because non-exhaust/resuspension and mechanical abrasion processes strongly shape metal exposure profiles, mitigation should prioritize (i) improved cabin filtration and ventilation management in buses and taxis, (ii) dust suppression and roadway cleanliness in high-traffic corridors, and (iii) maintenance and material optimization for braking and rail components to reduce abrasion-derived emissions. Overall, integrating personal exposure monitoring with metal speciation and EPA-based risk metrics provides an actionable evidence base to support mode-specific interventions and more equitable transport policies in rapidly motorizing megacities.

VI. Recommendations for Further Study

While this study provides a comprehensive assessment of $PM_{2.5}$-bound metal exposure and associated inhalation health risks across major transport microenvironments in Bangkok, several knowledge gaps and methodological extensions warrant further investigation. First, future studies should adopt a 24-hour, multi-microenvironment exposure framework that integrates commuting, residential, occupational, and leisure-time environments. This approach would enable more accurate estimation of lifetime exposure and health risk, particularly for metals such as Cd and Mn that accumulate over time and dominate both carcinogenic and non-carcinogenic risk metrics.

Second, source apportionment analyses using advanced multivariate and receptor models (e.g., PMF, PCA-MLR, or hybrid approaches combining chemical tracers with activity data) should be applied to better quantify the relative contributions of non-exhaust traffic emissions, mechanical abrasion, and background sources. Incorporating isotopic or elemental ratio diagnostics could further strengthen discrimination between rail-related abrasion and road dust resuspension. Third, future research should examine metal bioaccessibility and speciation, rather than relying solely on total concentrations. In vitro lung fluid extraction or particle solubility assays would help refine risk estimates by accounting for the fraction of metals that is biologically available upon inhalation, thereby reducing uncertainty in health risk characterization.

Additionally, Fourth, longitudinal and high-resolution temporal monitoring is recommended to capture diurnal, weekly, and episodic variability in $PM_{2.5}$ and metal concentrations, especially during traffic peaks and extreme pollution events. The integration of low-cost sensors with periodic reference-grade calibration could support continuous exposure profiling in transport microenvironments. Fifth, vulnerable and high-exposure subpopulations such as bus drivers, traffic police, and frequent commuters should be explicitly assessed to evaluate cumulative occupational and commuting risks. Linking exposure data with biomarkers of effect (e.g., oxidative stress or inflammatory markers) would strengthen causal inference between metal exposure and early health outcomes. Finally, future studies should evaluate the effectiveness of mitigation and policy interventions, including upgraded cabin filtration systems, alternative brake and rail materials, dust suppression measures, and fleet modernization. Scenario-based risk modeling could be used to estimate potential reductions in HQ and ILCR under different intervention strategies, providing evidence to inform transport and air quality policy in Bangkok and other megacities.

Collectively, these research directions would enhance the scientific basis for inhalation health risk assessment and support the development of targeted, source-specific strategies to reduce commuter exposure to $PM_{2.5}$-bound metals.

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