

Dispersion normalisation method for improved long-term trend evaluation: Heavy Metals in ambient air in the Czech Republic, Central Europe (2010–2021)

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Abstract. Long-term trends in atmospheric concentrations of heavy metals subject to legislative immission limits (As, Cd, Pb, Ni —Arsenic (6.0 ng m^{-3}), Cadmium (5.0 ng m^{-3}), Lead (500 ng m^{-3}), and Nickel (20 ng m^{-3})— were evaluated at selected monitoring stations representing different environmental settings across the Czech Republic (in Central Europe) over ~~twelve~~ 12 years. Dispersion normalisation was applied to reduce the influence of meteorological variability on measured concentrations and ~~The dispersion normalisation method, which suppresses the influence of meteorological conditions on observed heavy metal concentrations, was employed~~ to assess the effectiveness of legislative emission control measures. The results demonstrated statistically significant decreasing trends ($p < 0.001$) across all station ~~types~~types, and for all monitored heavy metals, ~~except for~~aside from Nickel at industrial stations, where no significant trend was detected. Furthermore, ~~the~~ systematic differences between original and dispersion-normalised concentration data ~~confirm~~indicate that meteorological variability can, in some cases, mask ~~true~~actual emission levels, potentially leading to misinterpretation of air quality trends. Dispersion normalisation proved to be a suitable method for long-term trend assessment and for quantifying the impact of regulatory measures on air quality. ~~Its key advantages include methodological simplicity~~The method's key advantages include simplicity and the availability of ~~the~~input data required ~~for the calculation of~~to calculate the ventilation coefficient (e.g., from ERA5).

1 Introduction

Heavy metal (HM) pollution remains a significant environmental issue due to its persistence in the environment and its harmful effects on human health and ecosystems, ~~such as~~including toxicity and bioaccumulation in the human body (EEA, 2013; Mitra et al., 2022; Suvarapu and Baek, 2017). ~~HM in the atmosphere, heavy metals are usually released into the atmosphere as aerosol particles, in particulate form or become associated with aerosol particles after release. Once released into the atmosphere, HMs can be absorbed onto aerosol particles,~~ which ~~can then~~may be inhaled, ~~leading to severe health issues and contribute to adverse health effects. Additionally, these~~These HM-polluted particles can also be deposited into soil and water, ~~thereby contaminating the broader environment, contributing to wider environmental contamination.~~ The emission of HMs into the atmosphere is

30 ~~heavily~~ associated with anthropogenic ~~operationsactivities~~, including industry, transportation, and mining ~~activities~~ (Ma et al., 2024).

Despite ~~the~~ reductions in emissions and ambient levels since the 1990s (EEA, 2024; EEA, 2025), HMs still require strict regulation as they do not degrade in nature (e.g., Ahmad et al., 2021; Briffa et al., 2020; Mitra et al., 2022). This ~~demand is demonstrated by need is reflected in~~ regulations such as the Convention on Long-range Transboundary Air Pollution (CLRTAP) - Protocol on Heavy Metals, Directive 2004/107/EC, and Directive 2008/50/EC. The relevance of this issue is further underscored by ~~the release of~~ EU Regulation 2023/915, on maximum levels of ~~food contamination by selected elements, selected elemental contamination in food~~, including ~~heavy metals~~ HMs, and Directive (EU) 2024/2881, on ambient air quality and cleaner air for Europe. In the Czech Republic, national law integrates these EU regulations and directives concerning HM emission and immission limits through Act no. 201/2012 Sb. The immission limits for selected ~~heavy~~ ~~metals~~ HMs, such as ~~Arsenic-arsenic~~ (As, 6.0 ng m⁻³), ~~Cadmium-cadmium~~ (Cd, 5.0 ng m⁻³), ~~Lead-lead~~ (Pb, 500 ng m⁻³) and ~~Nickel-nickel~~ (Ni, 20 ng m⁻³), have been established in the Czech Republic.

This study examines trends in HM concentrations in relation to immission limits in the Czech Republic over the period 2010–2021, following a substantial decline in ambient HM concentrations. The analysis covers multiple types of environments and focuses on the effects of current regulatory measures. Long-term exceedance of immission limits is currently not observed across most of the Czech Republic.

Central Europe remains one of the regions with the highest concentrations and deposition rates of HMs in Europe (Travnikov et al., 2020). At the same time, air quality in this region has been steadily improving (Aas et al., 2024). To better understand these developments, this study addresses three main objectives. Dispersion normalisation is applied here for the first time in the context of HM assessment to reduce the influence of meteorological variability on observed concentrations and trends at air quality monitoring stations.

The main objectives of the study are:

~~This study investigates the evolution in HM concentrations with immission limits in the Czech Republic from 2010 to 2021 (the period after a substantial lowering of HM concentrations in ambient air) across various environments, focusing on the effects of current regulations. These limits are not being exceeded long term in most of the territory of the Czech Republic.~~

~~Central Europe is among the regions with the highest concentrations and deposition of HMs in Europe (Travnikov et al., 2020). However, air quality is significantly improving in this region (Aas et al., 2024), so we focused on these three sub-objectives in our study:~~

~~The application of dispersion normalization was used for the first time to remove the influence of meteorological changes in describing HM concentrations and trends at air quality stations.~~

~~The three main objectives were to investigate:~~

1. Evaluate long-term trends ~~of in HMs~~ HM concentration at different ~~types of~~ monitoring stations ~~types~~ in the Czech Republic.
2. Investigate the suitability of ~~the station's selection~~ selected monitoring stations and establish a representative ~~station~~ group.

A critical research question ~~addresses-is~~ whether a single monitoring station can ~~serve as a representative sample bc~~

~~representative for~~ the entire network or whether multiple stations are necessary to ensure data reliability. The study aims to identify potential methodological challenges and limitations associated with the selection and grouping of monitoring stations ~~the assessment of emission-related trends. for emission assessment purposes.~~

3. Examine the influence of meteorological variables on measured immission concentrations in relation to evaluating ~~the manifestation of the regulation of emission amount~~ the effects of emission regulations. The study evaluates whether normalisation techniques constitute an appropriate methodological tool for isolating emission signals from meteorological influence. Specifically, the study ~~is focused~~ focuses on dispersion normalisation using the ventilation coefficient ~~index~~. The differences between the original (HM_{OR}) and dispersion normalised (HM_{DN}) heavy metalsHM concentration datasets are evaluated. The goal of this objective is to explore the potential masking ~~of the changes in emission~~ of changes in emission signals by due to meteorological effects.

2 Sources of HMs and ~~Emission~~ emission structure in the Czech Republic

75 ~~Heavy metals HMs~~ HMs, similar to many other pollutants, are released into the atmosphere ~~by through~~ natural and anthropogenic processes, ~~as well as many other pollutants. The typical~~ Typical natural sources ~~for of the~~ studied HMs are volcanic eruptions, soil erosion ~~or and~~ soil dust, and wildfires (EEA, 2013; Tchounwou et al., 2012). The primary anthropogenic sources of ~~arsenic (As)~~ are metal smelters and fuel combustion (EEA, 2013), and the manufacturing of insecticides, herbicides, and similar inhibitors (Tchounwou et al., 2012). ~~Cadmium Cd (Cd) input into the atmosphere~~ emissions

80 from anthropogenic sources ~~is via~~ mainly originate the emission from of the production of non-ferrous metals, stationary fossil fuel combustion, and waste incineration, ~~etc.~~ Anthropogenic ~~activities that produce lead (Pb)~~ emissions ~~are include~~ traffic, fossil fuel combustion, waste incineration, ~~and the~~ production of non-ferrous metals, ~~etc.~~ Another important source of Pb is the resuspension of road dust, especially applicable to ~~Especially for traffic location, resuspension of road dust is also a very important source of Pb~~ locations affected by road traffic. ~~Nickel~~ The main source of anthropogenic Ni emissions (Ni) ~~released into the atmosphere by anthropogenic sources is mainly emitted by~~ the combustion of oil (~~for whether it be for~~ from heating, shipping, or power generation), mining, and waste incineration, ~~etc.~~ (EEA, 2013).

85 ~~The emissions in the Czech Republic changed both in terms of the amount and structure of sources between~~ Between 1990 and 2021, emissions in the Czech Republic changed both in terms of magnitude and source structure. The most dramatic ~~evolution change~~ was recorded ~~by the for~~ total ~~amount of~~ As and Pb emissions, ~~with~~ As emissions ~~dropped~~ dropping to 1.8% of the 1990 value, i.e., by 98.2%, and Pb emissions ~~decreased~~ decreasing to 4.6%, i.e., by 95.4%. In contrast, ~~the~~ Cd emissions ~~amount~~ declined only to 24% of the 1990 value, i.e., by 76% (Table 1). The ~~diverse structure of sources~~ change in source structure between 1990 and 2021 is also visible. ~~Whereas~~ While one of the main source sectors for almost all studied HMs was ~~Iron~~ iron and steel production in 1990, ~~the situation was different~~ in 2021, ~~where~~ the most common emission source was Public public electricity and heat production. The ~~limitation restriction of using Pb as a component of fuels~~ of Pb content in fuel

95 is ~~pronounced~~ reflected by the lowering of the contribution ~~sector of the~~ Road ~~transport~~ Transport: Passenger ~~car~~ Car section

(Fig. 1). ~~For easier interpretation, only the first three emission sources are listed. Detailed information~~More detailed information is accessible in CHMI reports; e.g. CIEP (2024), CHMI (2026). The described changes ~~prove indicate~~ that the restrictions under CLRTAP and EU directives (mentioned in the introduction) helped lower HM concentrations and limited some sources.

Table 1: ~~Overview of the total amount of emissions in 1990 and 2021 in the Czech Republic. Overview of emission total amount in 1990 and 2021 in the Czech Republic. The structure of emissions developed-changed in response to based-on technological developments and the laws regulation regulatory measures.~~ Source: CEIP (2026).

Year	1990	2021	Reduction
Component	Emission Total emissions [t]		%*
As	69.5	1.3	98.2
Cd	5.3	1.3	75.9
Ni	55.0	4.3	92.1
Pb	316.6	14.7	95.4

* Reduction ~~compare to 100% in relative to~~ 1990 levels.

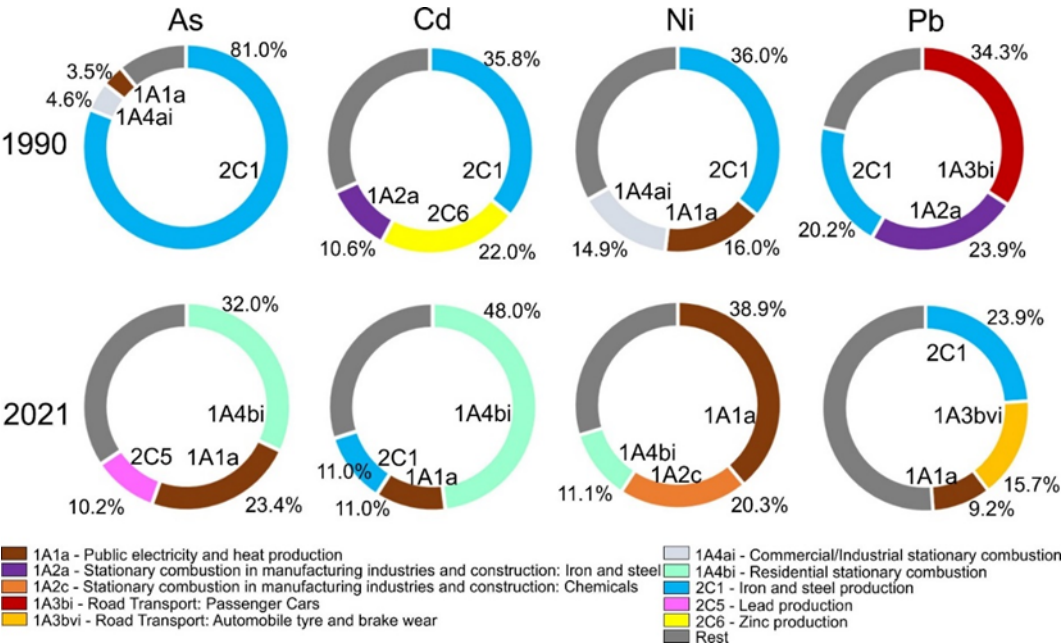


Figure 1: ~~Structure of selected HM emission sources according to economic sectors in 1990 and 2021. Only the first three sectors are visualised; the other sources are in the 'rest' category. Structure of selected HM emission sources according to NFR sectors in 1990 and 2021. Only first three sectors are visualised, the other sources in category rest.~~ Source: CEIP (20246).

A similar reduction in HM emissions observed in the Czech Republic was ~~recorded-observed~~ in the wider European region context (Fig. S1). ~~—~~Beyond domestic emission sources, transboundary transport should be considered as a significant contributor to elevated HMs concentrations. For instance, according to Travníkov et al. (2020), transboundary transport accounts for 7–99% of ~~cadmium~~ (Cd) deposition from non-national sources in EMEP member countries.

115 3 Atmospheric boundary layer and ~~Ventilation-ventilation~~ coefficient

~~The meteorological—~~Meteorological conditions affect measured pollutant concentrations. Some ~~of the~~ meteorological parameters are spatially variable. Their involvement in ~~the calculation of~~calculating their influence on pollution concentration is ~~too complicated~~complex and ~~ensure thus requires the use of similarly using~~ complex models (Gilliam et al., 2006; Williams, 2013; Zhang, 2019). To simplify calculations of the effect of meteorological conditions on pollutant concentrations, the
120 ventilation coefficient (VC) parameter can be used. VC_i is defined as the product of atmospheric mixing layer height (MLH_i) and wind speed (u_i) at a particular height for the selected time interval (i) (e.g., Chen et al., 2022; Ferguson, 2001), as shown in Equation (1):

$$VC_i = MLH_i \times u_i, \text{—} \quad (1)$$

The relation between MLH_i ~~and~~ wind speed, and air pollution ~~is has been regularly~~ studied regularly (e.g., Ferguson, 2001; Holzworth, 1967; Iyer and Raj, n.d.; Soleimanpour et al., 2023; Venegas and Mazzeo, 1999). An example of VC calculation
125 can be found in Ashrafi et al., (2009); Wu et al., (2023).

The ~~character of~~variation in MLH is an important factor enabling the dilution of atmospheric pollution. MLH is a sublayer of the atmospheric boundary layer (ABL), ~~that reaches~~reaching heights from ~~the ground- level~~ to hundreds of ~~meters-metres and units of up to several~~ kilometres. The dominant turbulent motion in this layer is driven by mechanical and thermal convection.
130 Mechanical ~~convection effect is the result~~results from ~~of~~ wind shear, while thermal convection is caused by the interaction of ~~the~~ Earth's ground-surface with incoming solar radiation and ~~the~~ subsequent transfer of heat to a relatively cooler atmosphere (Berg et al., 2013; Stull, 2003, 2016). The diurnal height variation of MLH and ABL is the result of ~~the character of~~ meteorological factors, such as temperature, moisture, solar radiation, ~~—~~wind, and turbulence, ~~etc.~~ (Stull, 2016).

The VC and MLH parameters capture the evolution of ~~a~~key meteorological factors and can be used for dispersion
135 normalisation. Accounting for meteorological conditions allows for more reliable evaluation and comparison of concentration data.

4 Methods

4.1 Selection of stations

HMs are regularly monitored within the National Air Quality Monitoring Network of the Czech Hydrometeorological Institute (CHMI). ~~This-The~~ monitoring network comprises over 30 manual stations and ~~about-ca.~~ 100 automatic air quality stations. However, for this study, only results from manual stations were ~~deemed~~ suitable. Further station selection was necessary to evaluate ~~results-only-from-the-only~~ measurements with a complete data series ~~of-for~~ all selected parameters from 2010 to 2021. The data series had to fulfil ~~the~~ criteria ~~set-outlined~~ in Directive 2008/50/EC, Annex I and Directive 2004/107/EC, Annex IV for the presentation and validity of air pollution characteristics—, i.e., ~~Which-is~~ a minimum of 90% data coverage in an individual time period (CHMI, 2023). Under these rules, 16 stations across the Czech Republic were selected for this study (Fig. 2). These stations are located in different geographical areas.

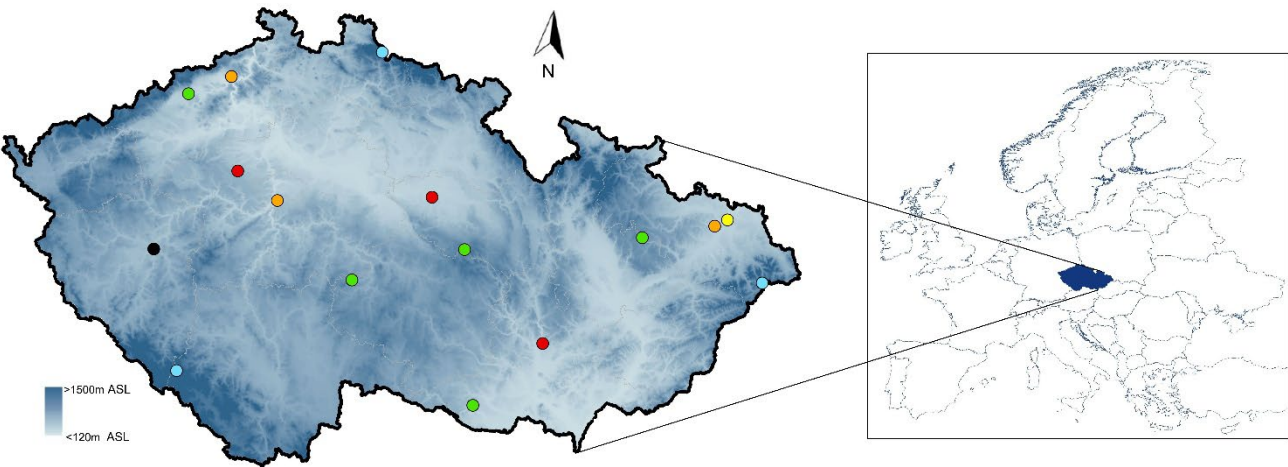


Figure 2: Spatial distribution of selected stations ~~measured-measuring~~ HM in ~~the~~ period of 2010–2021 ~~in the~~, Czech Republic, Europe. Colours representing particular station types. Background Rural Mountain stations— (BRM) are coloured light blue, Background Rural Lowland stations —(BRL) in green, Background Urban stations —(BU) in red, Background Suburban stations —(BS) in orange, Industrial station (I) —in yellow and Traffic station —(T) in black. ~~The map data were proecessed in ArcGis programmesoftware;~~ ASL denotesis metres above sea level ~~The map data were processed in ArcGis programme~~ (ESRI, 2018).

Data were processed and evaluated in individual categories based on station type and altitude. The dataset was split into the following six categories: Background Rural Mountain stations —(BRM), Background Rural Lowland stations —(BRL), Background Urban stations —BU(BU), Background Suburban stations —(BS), Industrial station —(I), and Traffic station —(T). Basic characteristics are listed in Table 2 and S1. The ~~categorization-categorisation~~ was done according to EOI classification, which ~~itself~~ is based on Council Decision 97/101/EC (CHMI, 2023). Background rural stations were split into

two subcategories based on ~~altitudes of stations~~station altitude. The border between lowland and mountain was set to 800 m a.s.l. ~~(metres above sea level)~~.
The ~~one-main~~ criterion for data processing was to maintain consistency and comparability within the time series; ~~as such, the period 2019–2021 was chosen. (the period 2010–2021 was chosen).~~ Data after 2021 ~~may exhibit~~exhibited minor inconsistencies, as technical issues with laboratory devices in 2022 and subsequent years resulted in temporary delays in analyses and ~~limited-produced~~ gaps in the dataset.

Table 2: Names and categories of selected stations.

<u>Station name</u>	<u>Station abbreviation</u>	<u>Category</u>
<u>Brno-Líšeň</u>	<u>BBN</u>	<u>BU - background urban station</u>
<u>Bílý Kříž</u>	<u>BKR</u>	<u>BRM - background rural mountain station</u>
<u>Červená hora</u>	<u>CER</u>	<u>BRL - background rural lowland station</u>
<u>Churáňov</u>	<u>CHU</u>	<u>BRM - background rural mountain station</u>
<u>Jizerka</u>	<u>JIZ</u>	<u>BRM - background rural mountain station</u>
<u>Kladno-Švermov</u>	<u>SKL</u>	<u>BU - background urban stations</u>
<u>Košetice</u>	<u>KOS</u>	<u>BRL - background rural lowland station</u>
<u>Kuchařovice</u>	<u>KUCH</u>	<u>BRL - background rural lowland station</u>
<u>Lom</u>	<u>LOM</u>	<u>BRL - background rural lowland station</u>
<u>Ostrava-Poruba</u>	<u>OPO</u>	<u>BS - background suburban station</u>
<u>Ostrava-Prívov</u>	<u>OPR</u>	<u>I - industrial station</u>
<u>Pardubice Dukla</u>	<u>EPA</u>	<u>BU - background urban station</u>
<u>Plzeň-Slovany</u>	<u>PLL</u>	<u>T- traffic station</u>
<u>Praha 4-Libuš</u>	<u>LIB</u>	<u>BS - background suburban station</u>
<u>Svratouch</u>	<u>ESV</u>	<u>BRL - rural lowland station</u>
<u>Ústí n.L.-Kočkov</u>	<u>ULK</u>	<u>BS - background suburban station</u>

4.2 Dispersion ~~normalization~~normalisation

~~Data set~~The final dataset covers 12 years of data measured under different meteorological conditions. Therefore, ~~the~~ rightcorrect evaluation ~~needs-requires~~ data ~~normalization-normalisation~~ and ~~the exclusion-reduction~~ of meteorological conditions. The reduction of meteorological influence was calculated according to a recently developed method intended for dispersion ~~normalization-normalisation~~ of Positive Matrix Factorization (~~DN~~PMF) analysis. This approach is widely used by

the research community (e.g., Alfeus et al., 2024; Chen et al., 2022; Hopke, 2021; Mbengue et al., 2023; Wu et al., 2023; Yang et al., 2022). -Dispersion ~~normalization-normalisation~~ (DN)-concentration is calculated according to Equation (2):

$$C_{DN} = C_{iOR} \times \frac{VC_i}{\overline{VC}}, \quad (2)$$

Where C_{DN} ~~is-denotes~~ dispersion ~~normalized-normalised~~ concentrations, C_{iOR} ~~means-is~~ measured (original) concentration, VC_i is the value of the ventilation coefficient and $\overline{VC}$ is the averaged ventilation coefficient under the whole studied period.

VC in this study is used from Numerical Weather Prediction model Aladin operation by CHMI. ~~Calculation-The calculation~~ of VC for Czech conditions is based on methodology listed in Ferguson (2001) ~~and-for-calculation-with-is-used~~the thickness of MLH ~~andand~~ average wind speed in the MLH being used (Škáchová, 2020; Škáchová and Keder, 2025).

4.3 Long-term trends

Trends in concentrations of individual components were evaluated using the Theil-Sen method (Carslaw and Ropkins, 2012; Sen, 1968; Theil, 1950), ~~which-is~~ based on the non-parametric Mann-Kendall approach. This combination ~~makes-is an~~ effective tool for ~~analyzing-analysing~~ nonparametric data. In this analysis, the mean monthly values of the 75th or 95th percentiles of concentrations are considered (used e.g. in- Lhotka et al., 2019).

The average slope of the trend, denoted by the T parameter, is calculated as follows, Equation (3):

$$T[\% \cdot \text{yr}^{-1}] = 100 \cdot \left(\frac{C_{End}}{C_{Start}} - 1 \right) / N_{years}, \quad (3)$$

where:

N_{years} is the number of years over which measurements were taken.

C_{End} and C_{Start} represent the mean concentrations at the end and start of the measurement period, respectively.

The long-term trends were evaluated for both original and normalised data; however, the main focus was on the HM_{DN} results.

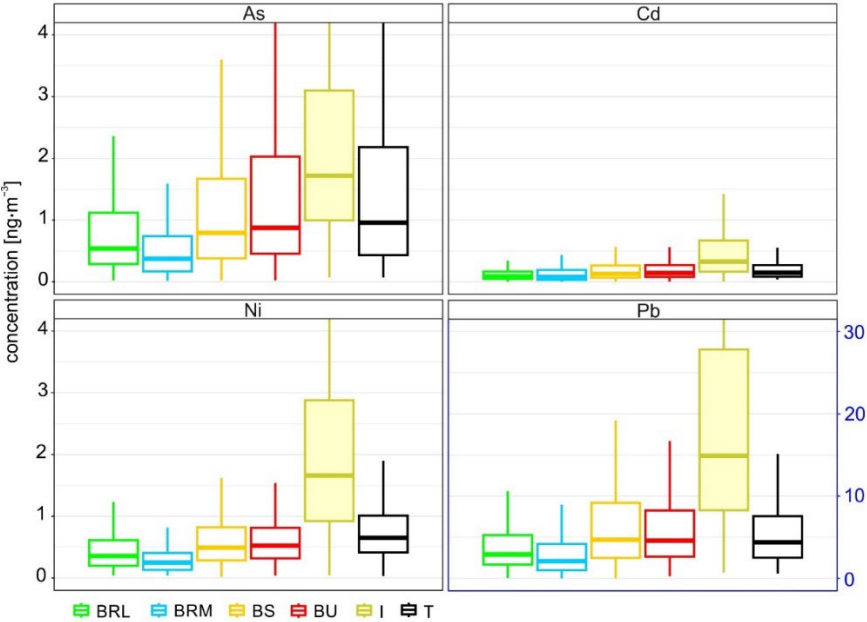
4.4 Sampling and ~~analysing-analysis~~ methods

The evaluated data were obtained from direct sampling at the selected stations. Each station measured the concentration of selected HMs in ~~the size fraction~~ PM_{10} size fraction with a sequential sampling device (sequential sampler SQ47/50, Sven Leckel, Germany). The measurement was scheduled ~~in-at a~~ frequency of 1×2 days; the air was continuously sampled for 24 hours starting at 0:00 UTC. Samples were, after exposure, analysed in accredited laboratories of the CHMI (according to the ČSN EN ISO/IEC 17025 standard) by ~~-mass-spectrometry-with~~ inductively coupled plasma mass spectrometry (ICP-MS) ~~method~~-(CHMI, 2023).

5 Results

205 5.1 Basic overview of HM concentrations measured at different station types

The HM concentrations varied according to station type. The lowest concentrations were observed at ~~mountain background stations (BRM) stations~~, where median values reached 0.35 ng m⁻³ for As, 0.08 ng m⁻³ for Cd, 0.21 ng m⁻³ for Ni, and 2.1 ng m⁻³ for Pb. The second category exhibiting relatively low concentrations ~~is was~~ represented by ~~lowland background stations (BRL stations)~~, with median concentrations of 0.54 ng m⁻³ for As, 0.09 ng m⁻³ for Cd, 0.33 ng m⁻³ for Ni, and 2.57 ng m⁻³ for Pb. ~~Stations Background suburban (BS), background urban (BU), and traffic (T) stations~~ demonstrated comparable concentration levels across all evaluated HMs. Notably, As concentrations were elevated at ~~BS, BU, and T station types~~, while Ni concentrations at ~~traffic T stations~~ differed from those recorded at BU and BS locations. ~~Ni at T station differed from BU and BS.~~ The median concentrations at BS, BU, and T stations were 0.79, 0.88, and 0.96 ng m⁻³ for As; 0.13, 0.14, and 0.14 ng m⁻³ for Cd; 0.48, 0.51, and 0.64 ng m⁻³ for Ni; and 4.70, 4.61, and 4.36 ng m⁻³ for Pb, respectively. The highest concentrations were measured at ~~the industrial (I) station type~~. The median As concentration reached 1.72 ng m⁻³, Cd 0.33 ng m⁻³, Ni 1.64 ng m⁻³ and Pb 14.90 ng m⁻³ (Fig. 3). ~~Comparison A comparison~~ between OR and DN results ~~is visualise~~ visualised in Fig. S2.



220 Figure 3: HM concentration overview at all station types, 2010–2021. Colours representing particular station types. Background Rural Lowland stations ~~(BRL) in green~~, Background Rural Mountain stations ~~(BRM) are coloured in light blue~~, Background Suburban stations ~~(BS) in orange~~, Background Urban stations ~~(BU) in red~~, Industrial station ~~(I) in yellow~~, and Traffic station ~~(T) in black~~.

5.2 Integration of stations into groups

The stations were selected based on the availability of time-series data and grouped by type (according to EOI classification) and geographical location. The ~~group's~~ representativeness of each group was ~~necessary to test its eligibility and tested to assess eligibility and~~ confirm the relevance of the presented results. The data used for this verification ~~was were~~ HM_{DN} to exclude ~~the~~ meteorological variability at individual stations.

5.2.1 Correlation between individual stations and particular HM_{DN}

In order to compare ~~the~~ results from individual stations, Spearman's correlation coefficient (R_s) values were calculated for HM_{DN} concentrations (Table S2–5). The correlation results were evaluated primarily for individual HM_{DN} ~~between using~~ station pairs. Following ~~(Mukaka, (2012) and (Schober and Schwarte, (2018), - an~~ $R_s = -of$ 0.70 was chosen as the threshold for strong correlation. For As_{DN} , nine ~~pairs of stations~~ station pairs showed strong correlations, with the strongest observed for LOM ($R_s > 0.7$ in 3 cases). For Cd, 22 stations pairs demonstrated strong correlations, most often observed for LIB ($R_s > 0.7$ in 6 cases) and EPA ($R_s > 0.7$ in 6 cases), and for KOS and LOM ($R_s > 0.7$ in 5 cases). No stations pairs with strong correlations were identified for Ni_{DN} (R_s 0.22–0.54). For Pb_{DN} , 19 pairs of stations showed strong correlations, most frequently involving KOS ($R_s > 0.7$ in 5 cases) and KUCH, ESV, LIB, BBN, and EPA ($R_s > 0.7$ in 4 cases each). These results suggest that Cd and Pb sources were relatively uniform across the Czech Republic and, by extension, Central Europe. In contrast, Ni concentrations and sources showed the most variability among all evaluated HM_{DN} (~~confirming the findings presented inconsistent with~~ Sections 5.2.2 and 5.3.1). The lowest correlations ($R_s < 0.6$) were found for HM_{DN} concentrations at BRM stations. Dissimilar air masses impacting these locations on the Czech Republic's borders likely ~~amplify increase~~ differences between ~~the~~ stations compared to ~~those at~~ inland locations. ~~Frequently, low~~ Low R_s occurred frequently at the OPR industrial station, especially for As and Pb. These differences could be due to specific sources of As and Pb emissions in the Ostrava region, including ~~(steelworks, metallurgical plants, and mines).~~

The overall R_s ~~result indicates~~ pattern suggests that the relationship among station results is also influenced by ~~their~~ mutual distance (Fig. 4). ~~-~~High R_s values (over 0.77) were found for OPR and OPO, BBN and KUCH, and LIB and SKL stations (except Ni_{DN}). When comparing correlations between ~~the~~ all four studied HM_{DN} at one station (Table S6–9), the highest correlation coefficients were found for Pb–Cd pairs ($R_s = 0.72–0.94$, median = 0.90) and Pb–As pairs ($R_s = 0.70–0.87$, median = 0.76), which could be related to their similar sources (industrial production/public energy/residential heating). An exception is the correlation between Pb and Cd at JIZ ($R_s = 0.62$), where ~~the~~ glass industry emissions influences Cd concentrations. Conversely, the lowest correlations were found for Ni–As and Ni–Pb pairs ($R_s = 0.32–0.66$), confirming ~~a the~~ different trend in Ni concentrations compared to the remaining HMs.

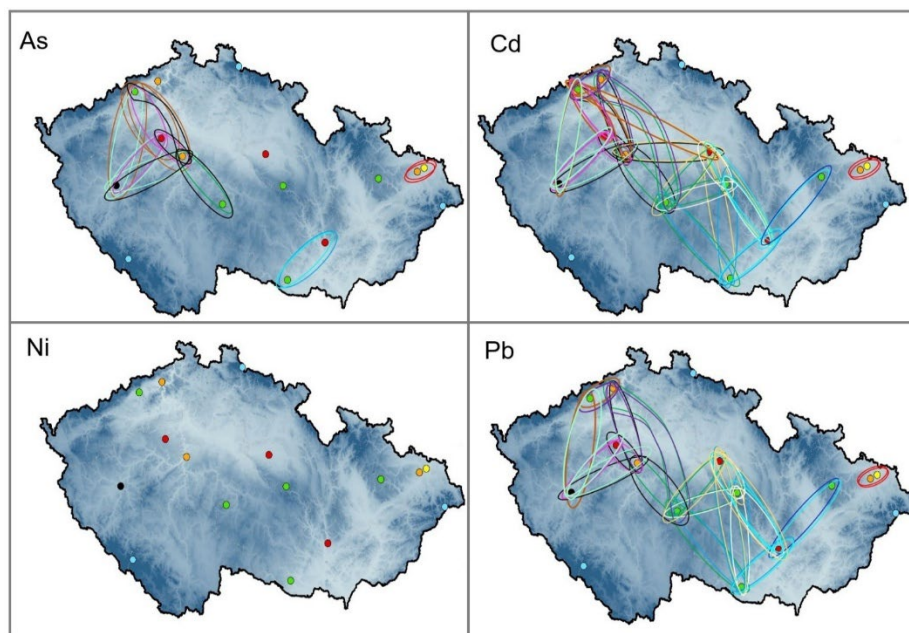


Figure 4: Spatial relationships among monitoring stations based on individual HM_{DN} correlations ($R_s > 0.7$). Coloured lines connect pairs of stations with strong correlations. Ellipses (circles) indicate spatial clusters of mutually correlated stations. The spatial relationships among stations according to individual HM_{DN} R_s values > 0.7 . Colours of dots representing particular station type. Background Rural Mountain stations (BRM) are coloured light blue, Background Rural Lowland stations (BRL) in green, Background Urban stations (BU) in red, Background Suburban stations (BS) in orange, Industrial station (I) in yellow and Traffic station (T) in black.

5.2.2 Verification by the using HM_{DN} long-term trends

Each station and the particular each HM were was evaluated separately for the long-term trend evolution (decreasing/increasing/stable) and its statistical significance. The As_{DN} concentration trend was generally decreasing ($p < 0.001$) at all stations (Table 23), Except except for three stations. The LOM and ESV stations (both BRL station types) showed a slight decreasing trend with no statistical significance. The EPA station (BU) was characterised by a decreasing trend on at the significance level $p < 0.5$ (Table S10). Results from the trend calculations for Cd_{DN} and Pb_{DN} concentrations showed not only a decreasing trend at the highest significance level across groups, but also for all individual stations. The Ni_{DN} concentrations differed during the study period from those of the remaining HMs. The concentration trend analyses from individual stations are consistent with the overall trend analyses for groups. Note that BMR trend results show a significant trend ($p < 0.001$); however, two stations (BKR and CHU) station shows showed a decreasing trend on at a lower significance level ($p < 0.01$). Similar situation occurred for BS stations, with a similar result occurring for BS stations. The overall BS trend was at a significant level $p < 0.001$, however, the ULK station indicated a decreasing trend on at $p < 0.01$, and OPO showed a very slight decreasing

tendency with no statistically significant trend. The industrial station OPR is represented by a slight increasing tendency with no statistically significant trend.

The individual HM concentrations trend analyses confirmed that the data series from individual stations are suitable for group calculations and further evaluation. However, the individual behaviour of HM concentrations at a particular station should be taken into account in the data interpretation.

The statistical significance results were the same for the grouped HM_{DN} and HM datasets. However, the percentage increase/decrease and confidence interval values were higher for HM results (Table 23).

Table 32: The results of long-term trend analyses of HM concentrations for the grouped stations (according to station type – EOI classification and altitude). Background Rural Lowland stations – (BRL), Background Rural Mountain stations – (BRM), Background Suburban stations – (BS), Background Urban stations – (BU), Industrial station (I) and Traffic station – (T). The overall trend is mentioned as a percentage increase/decrease per year, and the 95-% confidence intervals in the slope are listed in $\pm$ %/year. Significance level: $p < 0.001 = *$.**

Station type	As %/year	Cd %/year	Ni %/year	Pb %/year
BRL _{DN}	-3.76 [-4.76, -2.41] ***	-4.6 [-5.56, -3.63] ***	-3.39 [-4.29, -2.05] ***	-4.54 [-5.67, -3.65] ***
BRL	-4.73 [-6.11, -3.50] ***	-5.65 [-6.97, -4.16] ***	-3.56 [-4.64, -2.29] ***	-5.39 [-6.26, -4.25] ***
BRM _{DN}	-5.06 [-5.94, -4.19] ***	-5.93 [-6.63, -4.90] ***	-4.00 [-4.71, -2.90] ***	-4.76 [-5.48, -3.89] ***
BRM	-5.76 [-6.67, -4.96] ***	-6.82 [-7.79, -5.85] ***	-4.62 [-5.29, -3.81] ***	-5.46 [-6.10, -4.75] ***
BS _{DN}	-4.27 [-5.74, -2.70] ***	-6.18 [-7.15, -5.32] ***	-3.07 [-3.98, -2.01] ***	-5.78 [-6.72, -4.66] ***
BS	-4.17 [-5.56, -2.30] ***	-6.24 [-7.05, -5.08] ***	-3.14 [-3.97, -2.03] ***	-5.90 [-6.95, -4.75] ***
BU _{DN}	-3.46 [-4.81, -2.12] ***	-5.44 [-6.55, -4.30] ***	-3.66 [-4.48, -2.73] ***	-5.46 [-6.33, -4.30] ***
BU	-4.61 [-5.49, -3.10] ***	-5.99 [-7.27, -4.63] ***	-4.02 [-4.83, -3.05] ***	-6.16 [-6.94, -4.92] ***
I _{DN}	-3.73 [-4.87, -2.63] ***	-5.73 [-6.60, -4.73] ***	0.74 [-1.68, 4.14] ***	-4.95 [-5.82, -4.01] ***
I	-4.41 [-5.66, -3.42] ***	-6.19 [-7.44, -5.26] ***	0.23 [-1.8, 3.31] ***	-5.85 [-6.54, -5.09] ***

T _{DN}	-4.54 [-6.31, -2.54] ***	-5.36 [-6.89, -4.08] ***	-5.02 [-5.92, -3.97] ***	-5.86 [-6.77, -5.50] ***
T	-5.09 [-6.41, -3.00] ***	-6.04 [-7.16, -4.72] ***	-4.81 [-5.84, -3.78] ***	-6.52 [-7.34, -5.49] ***

5.3 HM concentrations and patterns across station types

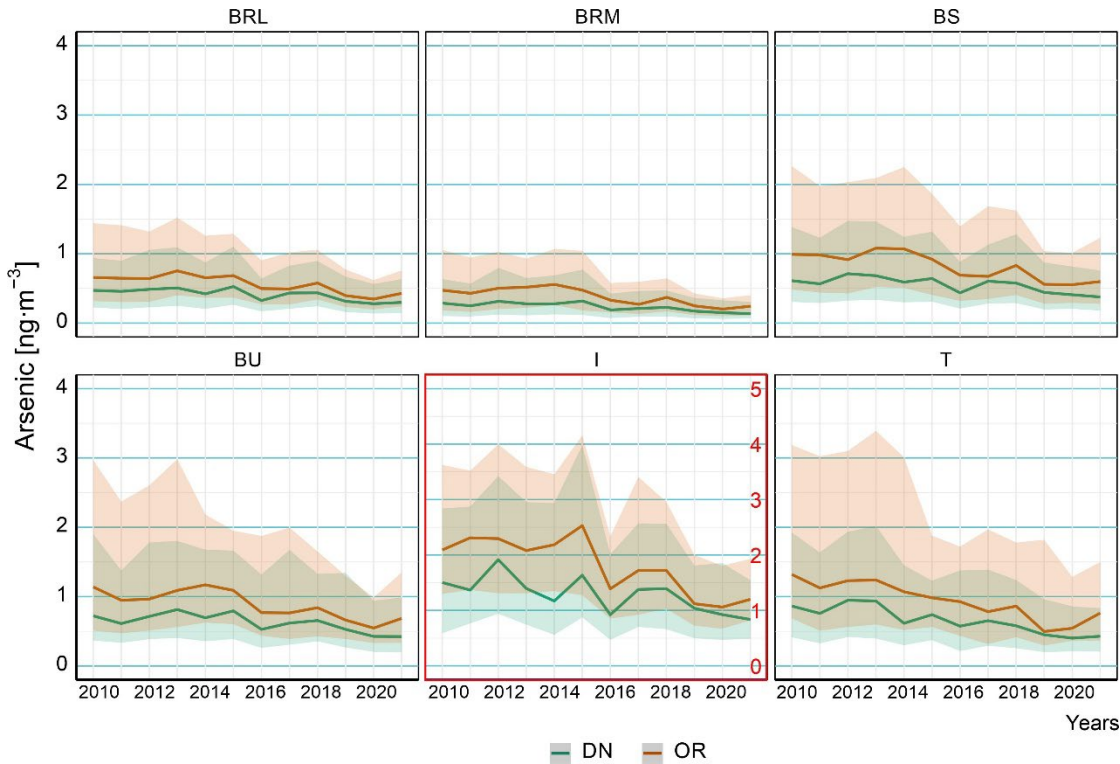
Across all investigated HMs, a divergence between original (OR) and dispersion normalised (DN) concentration datasets was observed, indicating meteorological influence on measured immission concentrations~~measurement immission~~. The magnitude of differences varies among station types. ~~The A relatively~~ stable response to meteorological conditions was ~~recorded-observed~~ at BRM (OR and DN difference interval ~~was of~~ 19–51%) and BRL stations (OR and DN divergence ~~reached of~~ 12–39%). However, ~~the~~ topography or microenvironmental characteristics were most evident at the T station type (OR–DN range: 8–52%), where street canyon effects and building configurations probably likely modify meteorological influences on pollutant dispersion and accumulation. The OR–DN convergence over the years ~~has~~ predominantly ranged from 7 to 52% across station types and the studied HM. The highest OR–DN convergence was observed in 2017 across the entire dataset (OR–DN range: 8–35%), indicating good dispersion conditions reflected at across all HM and environmental types. ~~Especially especially~~ for Cd at (BU and I stations) and Pb at (BU and T stations).

All station types exhibited statistically significant declines in As, Cd, Ni, and Pb ($p < 0.001$), providing clear evidence of the effectiveness of emission-reduction policies. The consistency of these trends across rural, suburban, urban, industrial, and traffic environments highlights the regional-scale impact of legislative measures rather than isolated local improvements. ~~The specific~~ Specific behaviour was recorded ~~in for~~ Ni, ~~where~~; ~~the~~ constant OR–DN divergence occurred at all station types. No trend was observed for the I station, which is an unusual pattern in the datasets studied.

5.3.1 Concentrations of As, Cd, Pb, and Ni at different station types

~~Arsenic-As shows~~ As showed statistically significant declines across all station types ($p < 0.001$), with annual reduction rates ranging from 3.45% (BU) to 5.93% (BRM). This consistent pattern suggests that the observed decrease is at least partly associated with legislative measures targeting As emissions. The highest AS_{DN} concentrations were recorded at I station ($0.83–1.92 \text{ ng m}^{-3}$), as expected, due to industry being the dominant source of As. Differences between AS_{OR} and AS_{DN} (7–47%) likely reflect not only episodic emission events but also the influence of meteorological conditions. A short-term increase in emission-related concentrations during 2011–2013 is more clearly captured in the dispersion-normalised data, particularly at T and I stations, where it is less apparent in the original time series due to meteorological variability. This effect can be observed in Fig. 5, where the DN data reveal a distinct, temporary deviation from the long-term trend, which is partially masked in the OR data.

315 This universal pattern confirms the effectiveness of legislative measures targeting As emissions. The highest As_{DN} concentrations were recorded at I station ($0.83\text{--}1.92\text{ ng m}^{-3}$), as expected, because industry is the primary source of As. As_{OR} – As_{DN} range: 7–47% probably reflected not only episodic emission events but also changes in meteorological influences. The short term increase in emission enrichment during 2011–2013 was partially masked by meteorological conditions in the As_{OR} data at the T and I stations. In those cases, the added value of the DN method is visible. The cleanest monitoring environment was BRM, based on As_{DN} , with values ranging from 0.13 to 0.32 ng m^{-3} (Fig. 5).



325 **Figure 5: Arsenic concentration at Background Rural Lowland stations —(BRL), Background Rural Mountain stations —(BRM), Background Suburban stations —(BS), Background Urban stations —(BU), Industrial station —(I) and Traffic station —(T), 2010–2021. The green line shows-represents the median concentration of dispersion normalised data (DN), and the orange line, is the median concentration of the original data series (OR). The coloured area visualises the interquartile range (25th–75th percentile).**

330 Cadmium exhibits-exhibited statistically significant declining trends across all station types ($p < 0.001$), with annual reduction rates ranging from 4.77% (BRL) to 6.15% (BS). The highest reduction rate at BS is probably-likely connected with-to emission control measures, —reflecting a combination of traffic emission reductions and decreased industrial activity in peri-urban zones. The meteorological influence pattern A pattern influenced by meteorology is evident during 2010–2014, when Cd_{OR} – Cd_{DN} ranged from 20% to 52% across all environments. However, from 2015, better dispersion conditions were observed

(Cd_{OR}–Cd_{DN}: 8%–46%) at several stations (BRL, BRM, BS, BU, I), particularly in 2017. Cd_{OR}–Cd_{DN} convergence was ~~probably likely~~ a result not only of the stabilisation of meteorological conditions but also of the consistency of emission sources ~~steadying~~ (Fig. 6).

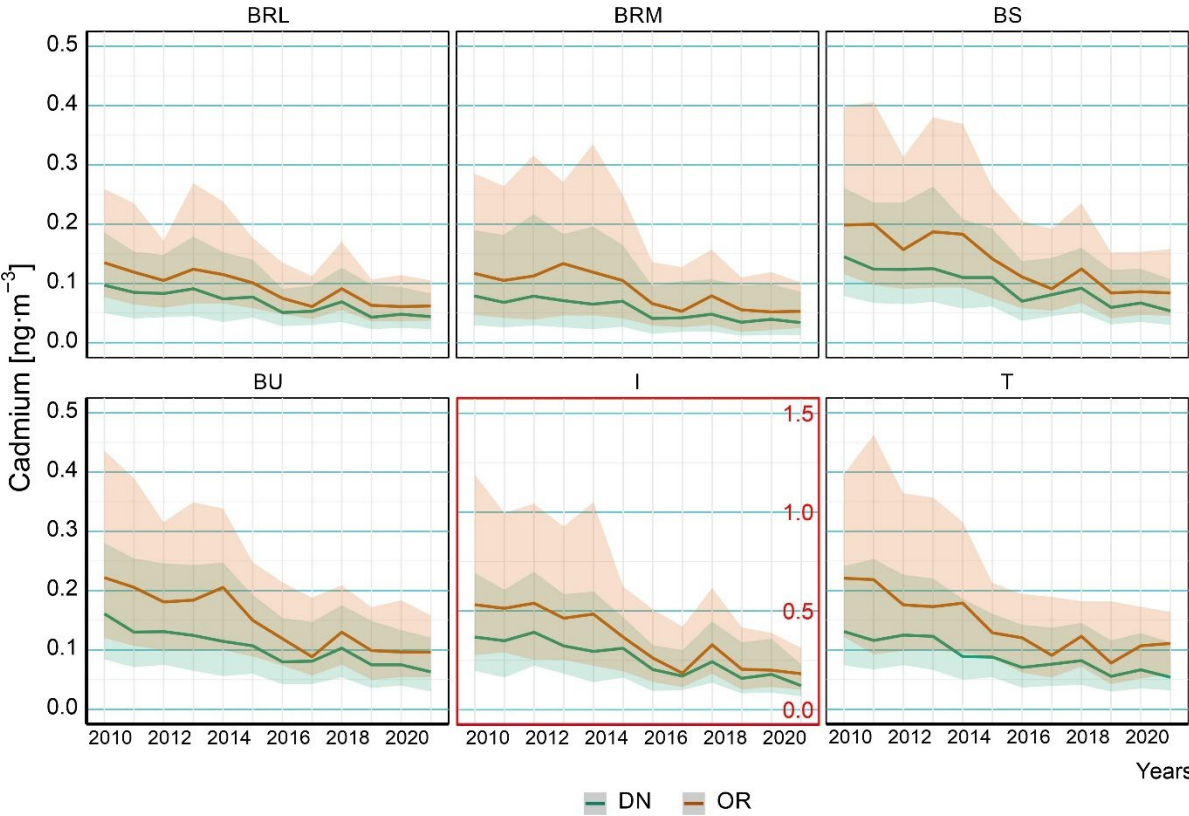


Figure: 6 Cadmium concentration at Background Rural Lowland stations ~~–(BRL)~~, Background Rural Mountain stations ~~–(BRM)~~, Background Suburban stations ~~–(BS)~~, Background Urban stations ~~–(BU)~~, Industrial station ~~–(I)~~ and Traffic station ~~–(T)~~, 2010–2021. The green line ~~shows-represents~~ the median concentration of dispersion normalising data (DN), and the orange line, ~~is~~ the median concentration of the original data series (OR). The coloured area visualises the interquartile range (25th–75th percentile).

The evolution of Ni concentration was ~~in-line~~consistent ~~with As~~with As, Cd, and Pb. Statistically significant declining trends were observed at all station types except the I station. Annual reduction rates ranging from 3.07% (BS) to 5.02% (T) showed no significant trends. ~~The industrial sources~~ probably likely suppress the regulatory measures more effectively than other HMs. Ni emissions were governed by sources with an operation regime less affected by seasonality, such as ~~(industrial processes, and tyre and brake abrasion from traffic. traffic—tyres and brakes abrasion).~~ The consistently high OR–DN divergence (20–50%) indicates that ~~nickel Ni~~ atmospheric behaviour of Ni is sensitive to meteorological conditions, expressed in all environments (Fig. 7).

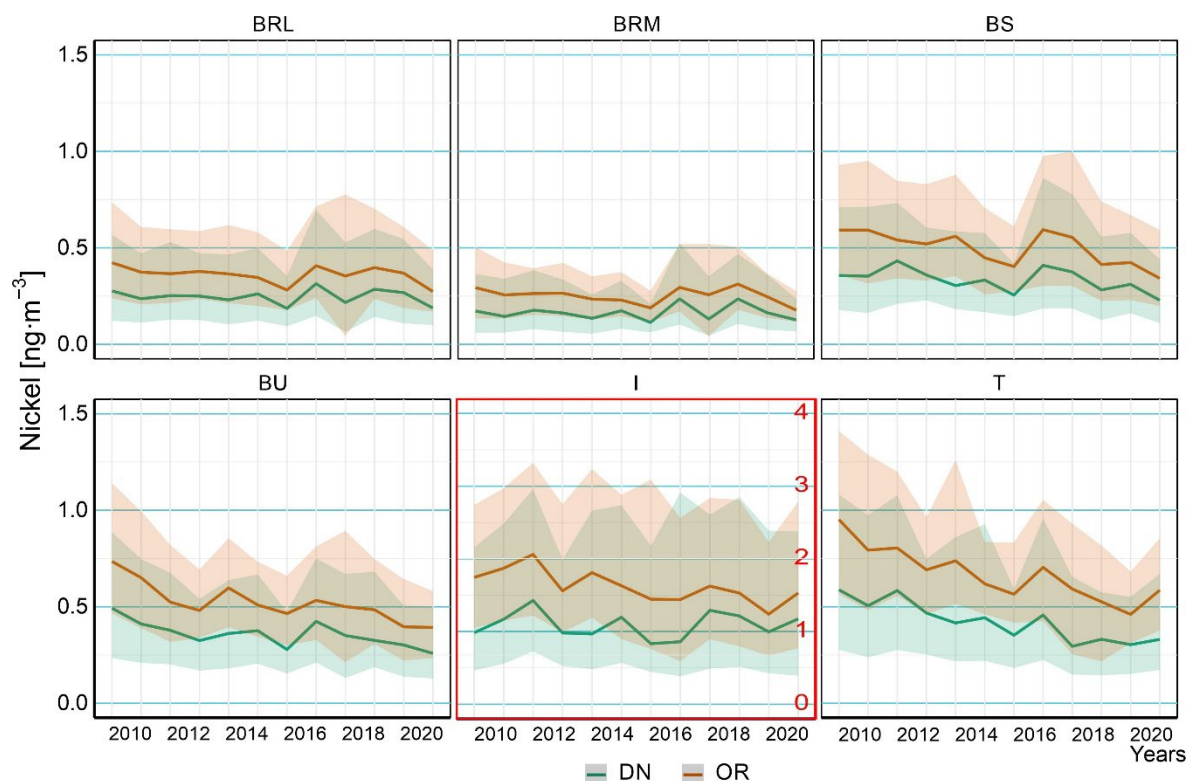


Figure 7: Nickel concentration at Background Rural Lowland stations (BRL), Background Rural Mountain stations (BRM), Background Suburban stations (BS), Background Urban stations (BU), Industrial station (I) and Traffic station (T), 2010–2021. The green line shows represents the median concentration of dispersion normalising data (DN), and the orange line, is the median concentration of the original data series (OR). The coloured area visualises the interquartile range (25th–75 percentile).

Lead demonstrates showed the most pronounced concentration reductions across the monitoring network, with statistically significant declines at all station types ($p < 0.001$) and annual reduction rates ranging from 4.59% (BRL) to 5.87% (T). The concentration level Pb concentrations at the I station type ($6.07\text{--}16.69\text{ ng m}^{-3}$) was were an order of magnitude higher than at BRM locations, which recorded the lowest concentrations ($0.89\text{--}2.17\text{ ng m}^{-3}$). This steep decline in Pb_{DN} likely reflects the effectiveness of lead-Pb phase-out policies in gasoline and industrial processes. The episodic peaks at Episodic peaks observed at I stations (2012, 2015, 2018) suggest that, despite overall declining trends, certain industrial processes continue to generate periodic increases in emissions. The exceptional High meteorological sensitivity at T stations (8–50% OR–DN divergence) combined with the highest reduction rate (5.87%) demonstrates indicates that while traffic-related Pb emissions have declined most rapidly substantially, street canyon effects continue to create complex complicate the relationships between emissions and observed concentrations. The second-highest reduction rate of Pb_{DN} values was detected observed at BS station types (5.78%) (Fig. 8).

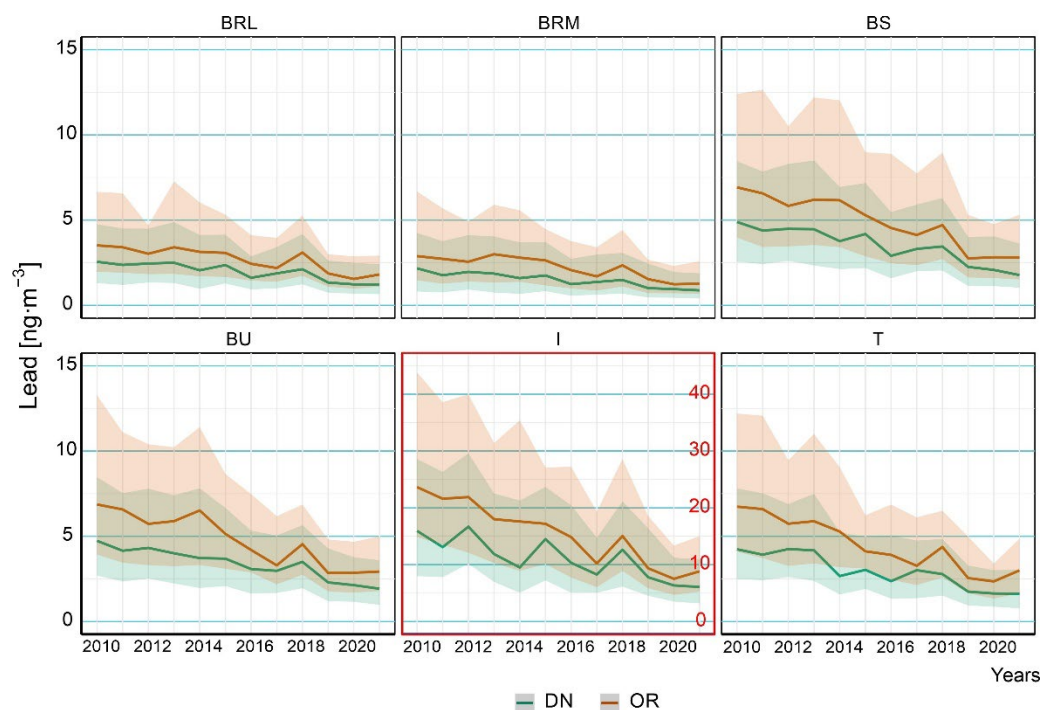


Figure 8: Lead concentration at Background Rural Lowland stations (BRL), Background Rural Mountain stations (BRM), Background Suburban stations (BS), Background Urban stations (BU), Industrial station (I) and Traffic station (T), 2010–2021. The green line shows represents the median concentration of dispersion normalising data (DN), and the orange line is the median concentration of the original data series (OR). The coloured area visualises the interquartile range (25th–75th percentile).

The dispersion conditions at the individual station are visualised in Figs. S3 and S4 according to the method-CHMI (2026) method and Škáchová and Keder (2025).

5.3.2 Patterns across station types: BRL, BRM, BS, BU, I, T

BRL stations Stations-BRL stations show consistent patterns for all HMs, characterised by a relatively narrow interquartile span, minimal temporal variability, and the lowest OR-DN divergence. This pattern was manifested-demonstrated by As_{DN} (0.28–0.51 ng m⁻³, 12–36% divergence), Cd_{DN} (0.04–0.09 ng m⁻³, 13–35% divergence), Ni_{DN} (0.13–0.27 ng m⁻³, 26–39% divergence), and Pb_{DN} (1.37–2.88 ng m⁻³, 13–36% divergence). Stable emission sources and minimal local anthropogenic influences could characterise the BRL environment. HM_{DN} concentrations (As, Cd, Ni and Pb) demonstrate statistically significant declining trends (3.75%, 4.77%, 3.39%- and 4.59% annually, respectively; $p < 0.001$).

BRM sStations-BRMBRM stations recorded the lowest concentrations of HM (As_{DN} N: 0.13–0.32 ng m⁻³, Cd_{DN} : 0.03–0.08 ng m⁻³, Ni_{DN} : 0.06–0.17 ng m⁻³ and Pb_{DN} : 0.89–2.17 ng m⁻³), confirming the least anthropogenically impacted monitoring locations. Stations-BRM stationsBRM stations- are represented-characterised by econstantpersistent permanent-OR-DN divergence (As: 20–50%, Cd: 21–47%, Ni: 31–49%, Pb: 19–43%; Table S11), indicating pronounced meteorological

sensitivity. ~~Probably, long~~Long-range atmospheric transport and meteorological dispersion were ~~the-likely the~~ dominant factors controlling the HM concentrations in this clean environment. The BRM stations demonstrate the highest annual reduction rates for Cd_{DN} (5.93%, $p < 0.001$), ~~subsequently followed by~~ reductions ~~for in~~ As_{DN} (5.06%, $p < 0.001$) and Pb_{DN} (4.75%, $p < 0.001$) and Ni_{DN} (4.00%, $p < 0.001$).

~~Stations BS stations BS stations~~ recorded the mean HM_{DN} concentration within the studied data series (As: 0.37–0.71 ng m⁻³, Cd: 0.05–0.14 ng m⁻³, Ni: 0.21–0.43 ng m⁻³, Pb: 1.77–4.89 ng m⁻³), reflecting the transitional nature between rural and urban environments. The highest annual reduction rates for Cd_{DN} (6.15%, $p < 0.001$) and Pb_{DN} (5.78%, $p < 0.001$) ~~are is probably likely~~ a result of emission control measures. The OR-DN divergence at BS stations (As: 11–44%, Cd: 11–40%, Pb: 18–38%; Table S11) could be influenced by ~~both the~~ local and regionally transport ~~of pollutionpollutants-ed~~ and changes in meteorological conditions (except ~~for~~ Ni: 20–46%).

~~Stations BU stations BU stations~~ ~~were represented by observed~~ concentration levels exceeding ~~concentration those~~ at BRL and BS stations (As_{DN}: 0.42–0.81, Cd_{DN}: 0.06–0.16 ng m⁻³, Ni_{DN}: 0.21–0.49 ng m⁻³, Pb_{DN}: 1.92–4.74 ng m⁻³) with a broader interquartile span. ~~The~~ OR-DN divergence ~~within particular for individual~~ HMs (As: 20–41%, Cd: 8–44%, Ni: 20–40%, Pb: 9–42%; Table S11) ~~could may~~ be caused by fluctuating emission sources and ~~the~~ dynamics of meteorological conditions in the urban environment. The lowest annual reduction rates among all station types were observed for As_{DN} (3.46%, $p < 0.001$); the remaining HMs showed decreases of Cd_{DN}: 5.37%, Pb_{DN}: 5.72%, ~~and~~ Ni_{DN}: 3.66% ($p < 0.001$). Despite the gradual decline, the influence of local sources or unfavourable meteorological conditions occurred episodically in the BU environment (e.g., ~~in~~ 2014).

Industrial stations consistently recorded the highest HM concentrations (As_{DN}: 0.83–1.91 ng m⁻³, Cd_{DN}: 0.12–0.36 ng m⁻³, Ni_{DN}: 0.83–1.42 ng m⁻³, Pb_{DN}: 6.07–16.69 ng m⁻³), confirming their proximity to major emission sources. ~~The episodic~~Episodic emission events and the response to dispersion conditions ~~are were~~ reflected in the OR-DN divergence (As: 7–47%, Cd: 8–39%, Ni: 20–47%, Pb: 15–46%; Table S11). The periodic influence of local emission sources was recorded, e.g., for Pb in 2012, 2015, and 2018. Despite the highest HM_{DN} concentrations, ~~medium moderate~~ annual reduction rates were observed (As_{DN}: 3.73%, Cd_{DN}: 5.73%, Pb_{DN}: 4.94%; all $p < 0.001$). Ni_{DN} ~~shows showed~~ a slight, non-significant increase (0.74%), suggesting that although emission controls have been partially effective, ~~the continued influence of~~ industrial sources may ~~be present in this location~~ still influence this type of station. Additionally, Ni_{DN} annual concentrations exceeded 1.00 ng m⁻³ in half of the study period, indicating the presence of Ni emission sources.

At T station, concentration ranges ~~for were~~ As_{DN}: 0.40–0.95 ng m⁻³, Cd_{DN}: 0.06–0.13 ng m⁻³, Ni_{DN}: 0.30–0.59 ng m⁻³, Pb_{DN}: 1.62–4.26 ng m⁻³. ~~The high~~High annual reduction rates were ~~registered observed~~ for all HMs - As_{DN} (4.50%, $p < 0.001$), Cd_{DN} (5.20%, $p < 0.001$), Ni_{DN} (5.02%, $p < 0.001$) and Pb_{DN} (5.87%, $p < 0.001$). The declining trend ~~is a result of likely reflects~~ a reduction in traffic-related emissions, ~~possibly thus~~ indicating the effectiveness of policies targeting vehicular emissions, including fuel quality improvements and fleet modernisation. ~~The~~ OR-DN divergence range (As: 9–44%, Cd: 20–52%, Pb: 8–50%, Ni: 27–50%; Table S11) ~~indicated the influence of showed not only the influence of both emission sources (probably ground-level emissions) but also the and~~ dispersion conditions ~~connected associated~~ with street canyon effects.

6 Discussion

The present study demonstrates ~~the positive impact of that~~ international and European legislative measures have had a positive effect on HM concentrations in ambient air in the Czech Republic, while also highlighting the ~~continuing~~ continued influence of industrial sources. The observed total annual declines in HM concentrations (1.7–8.1% per year) are substantially higher
 430 than the decline in total PM₁₀ concentrations across European background stations (1.8% per year) reported by Aas et al. (2024). This contrast supports the assumption that regulatory measures specifically targeting HM emissions have been more effective than general particulate matter mitigation strategies.

Comparable long-term trends were reported by Kyllönen et al. (2020), ~~who evaluated where~~ HM concentrations at a subarctic background station in Finland were evaluated. However, their study identified lower levels of statistical significance for As
 435 (no significant trend), Cd and Ni ($p < 0.05$), and Pb ($p < 0.001$) during the overlapping period. In contrast, our results show statistically significant decreasing trends ($p < 0.001$) for most station types and metals, indicating that emission ~~reduction~~
~~measures~~ reduction measures ~~probably may have~~ had a stronger impact in Central Europe, where industrial density and historical emission burdens were considerably higher.

A notable feature in the time series is the elevated HM concentrations observed in 2018. This increase coincided with ~~the an~~
 440 exceptionally dry and warm spring–summer period that affected large parts of western, northern, and central Europe (Bastos et al., 2020). The prolonged drought reduced wet deposition and enhanced atmospheric stability, leading to the accumulation of pollutants, including ~~heavy metals~~ HMs in the atmosphere. Similar meteorologically driven increases in pollutant concentrations and deposition have been documented within the EMEP network (Travnikov et al., 2020). This finding supports the need for ~~dispersion normalisation~~ DN approaches to ~~disentangle~~ separate emission trends from meteorological variability.
 445 The year 2018 serves as an example of how meteorological conditions can mask or amplify true emission signals in measured immission data. In the present study, the divergence between ~~original (OR)~~ and ~~dispersion normalised (DN)~~ concentrations reached up to 52% across station types and individual metals, demonstrating a substantial impact of atmospheric dispersion conditions on measured concentrations, particularly during years with unfavourable dispersion conditions, notably in the cold season of 2010–2013. Another such example is the increase in As and Cd concentrations at a traffic station during 2020 and
 450 2021, which could be incorrectly attributed to the impact of coronavirus pandemic-related restrictions. However, our results suggest that these increases were more likely attributable to adverse meteorological dispersion conditions rather than changes in emission levels. Without ~~dispersion normalisation~~ DN, such meteorologically induced anomalies could be incorrectly interpreted as actual changes in emission levels.

The ventilation coefficient (~~VC~~), which combines ~~combines mixing layer height~~ (MLH) and wind speed,

455 -proved to be an effective parameter ~~for capturing their~~ representing the dominant meteorological factors influencing pollutant dispersion. By applying DN, the underlying emission-driven trends become more identifiable, providing a more robust basis for assessing the effectiveness of legislative measures. This methodological approach is particularly valuable in datasets spanning multiple years, where interannual meteorological variability, as documented by Travníkov et al. (2020), who reported concentration changes of up to $\pm 50\%$ attributable solely to meteorological variability, could otherwise obscure long-term trends.

460 The only location ~~of among~~ all observed stations and HMs that exceeded the air pollution limit (specifically for As) was the SKL station ~~(urban background station type)-~~ classified as an urban background station. The annual air pollution limit for As ($6.0 \text{ ng} \cdot \text{m}^{-3}$) was exceeded ~~here at this station~~ in 2010, 2012, and 2013. The elevated As concentrations at this station are consistent with the influence of residential coal combustion, a well-documented source of As in urban environments, particularly during the heating season.

465 ~~Urban stations HM concentrations~~ HM concentrations at urban stations in the Czech Republic ~~can be compared~~ are comparable to with mean values reported for two urban background stations in Germany during 2008–2010 (Dimitriou and Kassomenos, 2017). While As concentrations were higher at Czech urban stations, Cd and Pb concentrations at German urban stations were ~~comparable similar~~ to those measured across Czech urban, industrial, and traffic sites. In contrast, Ni concentrations at German urban sites exceeded those observed at Czech industrial locations. ~~The German study identified traffic~~ In their study, traffic was identified as the dominant source of As, Cd, and Pb, whereas fuel oil combustion for residential heating was the primary source of Ni. These different source characteristics (traffic for As, Cd, and Pb vs. oil combustion for Ni) are consistent with the spatial patterns observed in our dataset, particularly the distinct behaviour of Ni at individual monitoring sites and the different trends of Ni at the industrial station.

475 According to the European Environment Agency EEA (2026), a continued overall decline in HM emissions has been observed across the EU, ~~confirming supporting~~ the long-term effectiveness of legislative frameworks such as the CLRTAP Heavy Metals Protocol and EU air quality directives. Nevertheless, Germany and Poland – two countries bordering the Czech Republic – remain among the largest HM emitters in the EU. Notably, Poland showed one of the smallest reductions in Cd (-2.7%) and Pb (-17.6%) emissions between 2005 and 2023. Given the documented importance of transboundary transport in Central Europe (Travníkov et al., 2020), ~~it is reasonable to assume that long-range transport contributes~~ long-range transport may contribute to the measured concentrations in the Czech Republic, particularly at background and mountain sites. Additionally, Slovakia and Hungary have been reported among countries with relatively high Pb and Cd concentrations within the EMEP region, further supporting the regional dimension of HM pollution.

480 Our findings of declining Pb and Cd concentrations across the Czech Republic are consistent with Hůnová et al. (2023), ~~who examined where~~ depositional fluxes of these substances were examined, and ~~documented a decline in Pb and Cd deposition from 2012 to 2019~~ was observed.

The ~~Absence-absence~~ of a statistically significant trend in Ni concentrations at the industrial station is a notable exception within the dataset. This pattern may reflect the local specificity ~~specific nature of Ni emission sources in the Ostrava region~~.

where metallurgical processes are largely independent of seasonal cycles, in contrast to combustion-related sources that drive trends in other HMs. This pattern may reflect the specific character of Ni emission sources in the Ostrava region, where metallurgical processes operate independently of seasonal cycles, unlike the combustion-related sources that drive trends in other heavy metals (Font et al., 2022).

7 Conclusion

The long-term trend of HMs at different station types in the Czech Republic was evaluated and showed with findings of a decreasing trend at the highest statistical significance level ($p < 0.001$) during the period 2010–2021. The results confirm the positive impact of restriction-regulatory measures implemented after 2008. Given the Czech Republic's central location and its exposure to regional emission patterns, the findings may also be relevant to the wider Central European context as outlined in Objective 1 of the Introduction. ; the representativeness of the results could be generalised to Central Europe, as the Czech Republic is located in the middle of Central Europe (see point 1 in the set objectives in the Introduction). The 16 stations with 16 stations with daily measurements over a 12-year period 12 year long daily measurement data series were studied, and the groups were evaluated according to their classification and spatial locations. The representativeness of the group was confirmed by the statistical significance of HM concentrations at individual stations ($p < 0.001$). The 9% of cases (6 results from 64) did not confirm the decreasing trend observed at the rest of the stations, which was probably-likely due to a specific pollution source at the location. The correlation analyses proved-showed the spatial connection between the stations, especially for Cd and Pb. The low-correlation Low correlations were found for Ni across all stations R_s ($R_s = 0.22$ – 0.52) was found for Ni across all stations. Three features characterise mountain sites: low HM concentrations, weak inter-station correlations ($R_s < 0.7$), and a relatively stable OR–DN difference (19–51%). This indicates a significant influence of dispersion conditions on HM concentrations. Three factors demonstrated specifications of mountain locations: 1. the lowest HM concentrations, 2. the result of R_s under 0.7 among all the HM_{DN} and particular stations, 3. a constant difference of OR–DN values that fluctuates between 19 to 51%. This indicates a non-negligible influence of the dispersion conditions on HM concentrations.

Generally, it was confirmed that grouping stations by classification yields representative results for station and environmental types. However, the specificity of the pollution sources must be taken into account, so more than one station is recommended for calculating the representative group (see point 2 in the set objectives in the Introduction).

The dispersion-normalised DN data evolution clearly demonstrated the effects-results of the regulation-regulatory measures. The specific-Specific meteorological conditions that characterise the dispersion from the mountains to the street canyon were observed were observed across the range of station environments, from mountain locations to street canyons. The OR–DN ratio confirms the potential of meteorological conditions to mask the real emission level. Our results confirm that within the law restriction-evaluation when evaluating HM emission regulations, dispersion-normalisation DN is a suitable method, providing

~~clearer results~~[the use of DN can provide clearer results](#). Although in this study, the ~~VC ventilation coefficient~~ was obtained from the Aladin model, which is not available to the broad user community, the variables used to calculate the ventilation coefficient are accessible via sources such as the Hysplit model (e.g., Rolph et al., 2017; Stein et al., 2015) or ERA5 (Copernicus Climate Change Service, 2023). The user community can apply ~~the dispersion normalisation~~[the DN](#) method without significant limitations.

Supplement link

The supplementary material is published alongside this article.

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[The text has undergone English language proofreading by Laurence Widell.](#)

Competing interests

The authors declare that they have no conflict of interest.

Data availability

The data used in this article were collected and processed with support from projects (ARAMIS and ACTRIS-CZ) and the Czech Hydrometeorological Institute. Therefore, data sharing is subject to specific conditions. The data are available from the corresponding author upon reasonable request.

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Review statement

The review statement will be added by Copernicus.

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