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Dust Pollution as an Indicator of Indoor Environment Quality

DOCTORAL DISSERTATION

Natural Sciences
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ABBREVIATIONS

µm - Micrometer (a unit of measurement)
AAA - Aplinkos Apsaugos Agentūra
ABS - Dermal absorption factor (unitless)
ADD - Average Daily Dose
AF - Skin adherence factor (mg/cm^2)
As - Arsenic
AT - Average time (days)
Ba - Barium
BW - Body weight (kg)
C - Concentration of the element (mg/kg)
Cd - Cadmium
CF - Conversion factor
CO - Carbon Monoxide
Co - Cobalt
CO_2 - Carbon Dioxide
Cr - Chromium
Cu - Copper
ED - Exposure duration (year)
ED-XRF - Energy Dispersive X-Ray Fluorescence.
EF - Enrichment Factor.
EF - Exposure frequency (days/year)
EU - European Union
Fe - Iron
g/cm^3 - Grams per cubic centimeter (a unit of measurement)
GmbH - Gesellschaft mit beschränkter Haftung (company with limited liability in German).
HCA - Hierarchical Clustering Analysis
Hg - Mercury
HI - Hazard Index
HQ - Hazard Quotient
HVAC - Heating, Ventilation, and Air Conditioning
IAPs - Indoor Air Pollutants
IDW - Inverse Distance Weighted
IngR - Ingestion rate (mg/day)
InhR - Inhalation rate (m^3/day)
km - Kilometer (a unit of measurement)
MAL - Maximum Allowable Limit

Max - Maximum
Mg - Magnesium
Mn - Manganese
Mo - Molybdenum
Ni - Nickel
NOAA - National Oceanic and Atmospheric Administration
Pb - Lead
PCA - Principal Component Analysis
PC – Principal Component
PEF - Particle emission factor (m^3/kg)
pH - Potential of Hydrogen (a measure of acidity or alkalinity)
PLI - Pollution Load Index
PM - Particulate Matter
PM10 - Particulate Matter with a diameter of 10 micrometers or less
PM2.5 - Particulate Matter with a diameter of 2.5 micrometers or less
PMF - Positive Matrix Factorization
Pt - Platinum
Rb - Rubidium
RfD - Reference dose (mg/kg day)
SA - Surface area of the skin exposed to heavy metals (cm^2)
Sc - Scandium
SINPHONIE - School Indoor Pollution and Health: Observatory network in Europe
Sr – Strontium
THI - Total Hazard Index
UK - United Kingdom
USA - United States of America.
V - Vanadium
VF - Volatilization factor (m^3/kg)
VOCs - Volatile Organic Compounds
WHO -World Health Organization
XL2 - Model designation for the Niton XRF Analyzer.
XRF - X-Ray Fluorescence.
Zn - Zinc

INTRODUCTION

Soil, a porous and dynamic substrate, facilitates critical interactions between air, water, and biota, directly influencing ecosystem health. Alterations in soil processes have a significant impact on ecosystem functioning, as the complex balance between inorganic and organic substances within the soil gives rise to a range of environmental issues. In order to maintain the quality and functionality of soil, it is essential that all soil types are maintained in a sustainable condition. The presence of heavy metals can have both beneficial and detrimental effects. While metals such as zinc, copper, and iron are essential for plant growth in trace amounts, their accumulation above phytotoxic concentrations (such as $Zn > 100–300$ mg kg^{-1}) can disrupt soil processes and negatively impact ecosystem functioning (Ignatavičius et al. 2022; Kabata-Pendias 2011).

Soil's role in the environment extends beyond the ground, as it can be a source of airborne pollutants, particularly dust. Dust, which is composed of solid particles in the form of fine powder (less than 100 μm), is commonly referred to as particulate matter (PM). It is highly polluting due to its ability to be easily transported through the air (Yesilkanat et al. 2021; Muhamad-Darus et al. 2017; Radhi et al. 2021). In urban areas, street dust plays a significant role in pollution. It is a complex mixture of particles that can contain various components such as organic matter, heavy metals, inorganic substances, mold spores, dander, and pollen. These particles can be resuspended into the air by vehicle movement and wind, thus becoming a major source of atmospheric pollution. They can settle on impermeable surfaces within cities, including roads and roofs (Sezgin et al. 2004; Al-Khashman et al. 2004; Suryawanshi et al. 2016; Trujillo-González et al. 2016). A study, such as the analysis of urban dust in six Mexican cities, emphasizes the widespread presence of heavy metal pollution and its associated health hazards, particularly for children (Aguilera et al. 2022).

Recent studies on dust sourcing have uncovered various environmental pollutants, primarily in outdoor environments, this encompasses research on the concentration and composition of elements in dust (Zeider et al. 2023; Vlasov et al., 2022; Gunawardana et al., 2012; Charlesworth et al. 2011), the impact of dust and aerosol particle size on the transportation of contaminants from outdoor to indoor environments (Rodríguez-Chávez et al. 2021), and the movement of polluted soil and airborne particles into indoor dust (David and Paloma, 2009). However, these studies predominantly focus on emissions from industries and vehicles, with limited consideration given to indoor dust pollutants found in schools.

This connection between outdoor and indoor environments underscores the need to evaluate not only human-caused pollution but also natural sources of contamination. For instance, the 2011 Grimsvotn eruption in Iceland brought attention to the impact of natural phenomena, such as volcanic eruptions, on environmental pollution. This eruption notably affected atmospheric aerosol concentrations in Vilnius, Lithuania, which is nearly 3000 km away from Iceland (Kvietkus et al. 2013). This occurrence emphasizes the importance of taking into account both human-caused and naturally occurring sources when evaluating indoor dust pollution.

While outdoor sources of pollution are well-studied, indoor dust poses a significant, yet often overlooked, concern for human health, especially in indoor environments like schools. Indoor dust poses a significant concern for individuals who work, live, or spend a majority of their time indoors. This dust is a combination of particulate matter derived from both interior and exterior sources, and it can accumulate within indoor environments. Indoor dust serves as a notable source of metal exposure for people and this dust originates from various internal sources, including cooking, smoking, sweeping, wall erosion, rubber carpet materials, painting, building and furniture materials, consumer products, and other interior activities. On the other hand, external pollution sources contribute to indoor dust through the infiltration of emissions from traffic, auto repair, welding, waste burning, playground dust, and so on (Muhamad-Darus et al. 2017; Radhi et al. 2021). Comparative studies of indoor dust pollution across various global locations reveal significant variations in heavy metal concentrations. For instance, studies show elevated levels of Cr, Cu, Zn, Pb, and Fe in indoor dust from Malaysia, Iraq, Hong Kong, and Nigeria, among others (Muhamad-Darus et al. 2017; Radhi et al. 2021; Olujimi et al, 2015; Sulaiman et al. 2016; Nkansah et al. 2015). Such variations are indicative of the widespread and diverse nature of indoor dust pollution, necessitating a deeper understanding of its sources and impacts on human health. Indoor dust in schools can have a significant impact on the health of children who study in classrooms. Children may be exposed to these heavy metals present in indoor dust through various routes such as inhalation, direct consumption of contaminated soils or food, and skin contact with polluted school materials (Moghtaderi et al. 2020). Children and the elderly are more vulnerable to the effects of heavy metals than adults. This vulnerability arises due to their behaviors, like hand-to-mouth contact, crawling activities, and their faster respiratory rate. These factors increase the likelihood of children ingesting heavy metals present in dust and inhaling more contaminated air compared to adults; also, As, Cd, Cr, and Pb are common environmental contaminants that might cause cancer as well as development disorders (Tan et al. 2016). Although there has been much global research on heavy metal contamination in urban contexts, specifically in street and interior dust (Aguilera et al., 2021; Rashed, 2008), there is a notable lack of knowledge in Lithuanian research, particularly when it comes to schools. This discrepancy is substantial, considering the proven health hazards linked to high levels of heavy metals found in indoor dust, as established by research conducted in nations such as the United Kingdom (Shi and Wang, 2021), Nigeria (Abdulraheem et a. 2022), and China (Roy et al. 2023; Wang et al. 2023). Moreover, the increased vulnerability of children to these contaminants, as documented in educational environments (Abdulraheem et a. 2022; Kunt and Turkyilmaz, 2023), emphasizes the need for further research in this field.

The primary contribution of this work is in its thorough examination of metal contamination in indoor dust within schools in Vilnius, a topic that has been rarely explored in Lithuanian research. The lack of research in local studies provides a distinct chance for our investigation to clarify the origins and composition of indoor dust pollution in educational environments, which is an important issue that is frequently disregarded. This study aims to answer whether the indoor dust found in these schools contains increased concentrations of heavy metals originating from a diverse range of both external and internal sources of pollution.

The findings from this study can help school administrators and public health authorities develop targeted actions to reduce indoor dust pollution. This will directly improve indoor air quality in schools and help protect children's health.

Our study aims to not only measure levels of heavy metals but also investigate their potential non-carcinogenic impacts on children in schools, specifically in the Vilnius region of Lithuania. By conducting pioneering research, we seek to establish a relationship between heavy metal levels in the topsoil and their accumulation in indoor dust within educational environments. This research will contribute to a better understanding of environmental routes of heavy metals in schools, filling a significant gap in both global and regional environmental health studies.

THE AIM AND OBJECTIVES OF THE STUDY

Aim of the Study:

To investigate the contribution of outdoor environmental pollution by HMs in soil and airborne PM to indoor dust contamination in schools as well as to evaluate the impact of this contamination on children's health risks.

Objectives:

To achieve this aim, the following objectives have been defined:

1. To evaluate the levels of HMs in indoor dust collected from public areas of general education schools, with the goal of identifying contamination levels and understanding patterns of pollutant accumulation.
2. To analyse the connection between concentrations of outdoor airborne PM and the concentrations of HMs in indoor dust within general education schools.
3. To investigate the relation between concentrations of HMs in the soil outside schools to their levels in the indoor dust of public spaces, thereby assessing the influence of soil contamination on the quality of the indoor environment.
4. To assess the potential health risk for schoolchildren exposed to HMs in dust from public areas, by integrating exposure modeling with epidemiological data and determining both non-carcinogenic and carcinogenic health risks.

THE NOVELTY OF THE RESEARCH

1. Provides a comprehensive analysis of HM concentrations in indoor dust within schools, offering new insights into contamination levels and distribution patterns in educational environments, a topic with limited prior research.
2. Uncovers the relationship between HM levels in outdoor soil and indoor dust, revealing the mechanisms by which pollutants transfer from outdoor to indoor environments. This enhances the scientific understanding of environmental contamination pathways and contributes new knowledge to the research field.
3. Establishes an association between outdoor air PM concentrations and HM in indoor dust. This finding contributes to understanding how outdoor air quality directly influences indoor environment.

4. By conducting a detailed health risk assessment that integrates exposure modeling with epidemiological data, the study provides new knowledge on the non-carcinogenic and carcinogenic health risks posed to children by exposure to HMs in indoor dust.

PROBLEMS

1. Insufficient data on the concentrations and distribution patterns of HMs in indoor dust within schools. This gap delays the assessment of contamination levels and the development of strategies to mitigate exposure risks for children.
2. Unclear mechanisms of pollutants transfer from outdoor soil as well as air PMs to indoor environments are not well understood. This limitation affects the ability to identify critical points of intervention to prevent indoor contamination in Vilnius.
3. Lack of comprehensive health risk assessments that integrate exposure modeling with epidemiological data to evaluate non-carcinogenic risks to children from HMs in indoor dust.

DEFENDED STATEMENTS

1. HMs are present in indoor dust within schools at concentrations that indicate significant contamination levels.
2. There is a correlation between the concentrations of HMs (such as As, Cu, Zn, and Pb) in outdoor soil and indoor dust.
3. A correlation exists between outdoor air PM concentrations and HM contamination in indoor dust.
4. Children attending schools are exposed to HM in indoor dust, which presents potential health risks, including both non-carcinogenic and carcinogenic effects.

1. Research Background

1.1. Metal Deposition

Air pollution, particularly from metals, is a serious and deadly problem. Human activities have caused an increase in the presence of metals in the biosphere, generally in urban areas. Stationary and mobile sources release large amounts of metals into the atmosphere, soil, and plants, which can cause damage to vegetation and crops, and ultimately affect human health if inhaled over time (Timothy and Williams, 2019).

Anthropogenic activities emit metals as aerosols, which travel long distances before depositing into soil and water systems. This deposition process leads to detrimental effects on natural water bodies as well as terrestrial and aquatic organisms (Soriano et al., 2012). While the deposition process helps cleanse the atmosphere, its eventual outcome involves transferring toxic atmospheric pollutants into water and soil environments (Muezzinoglu and Cizmecioglu, 2005). Metals are present in the atmosphere mainly in the form of particles. These particles can be transferred to land or water surfaces through various processes such as dry deposition (gravitational settling, impaction, turbulent transfer, and Brownian motion), wet deposition (nucleation and scavenging), and occult deposition (deposition of wetted particles through impaction or turbulent transfer) (Mohan et al., 2016; Collett, 1993). The size distribution of these particles is influenced by atmospheric transport processes, and the importance of each deposition process depends on the size of the particles. In rural and remote regions, dry deposition is primarily through impaction and turbulent transfer, while wet deposition occurs through nucleation and scavenging of wet aerosol particles (Shrivastav, 2001). Growth in urbanization and vehicle emissions is causing heightened air pollution. Consequently, research on urban air pollution is largely centered on road dust (Addo et al., 2012).

1.2. Metal Toxicity

Iron and copper aid in oxygen transport, zinc is essential for enzyme functioning, selenium acts as an antioxidant and hormone biosynthesis, and cobalt is significant for biosynthesis and metabolism. Vanadium and manganese regulate enzymes, while chromium, arsenic, and nickel have metabolic functions despite being toxic (Koller and Salah, 2018). Metals have important roles in human health, both as essential nutrients and as components of drugs (Gupta, 2018), they can act as carcinogens as well. When dissolved and released as pollutants in soil, water, and air, they enter the food chain and ultimately harm the cellular system, leading to an increased risk of cancer in humans. According to the International Agency for Research on Cancer, nonessential heavy metals like arsenic, cadmium, and chromium are significant cancer-causing agents (Rama Jyothi, 2020).

Heavy traffic on roads emits lead, cadmium, zinc, and nickel from anti-knock agents in fuel. Heavy metals in soil are useful for monitoring the impact of human activities, such as industrial and vehicular emissions, and atmospheric deposits. Factors such as traffic volume, highway characteristics, roadside distance, wind direction, rainfall, and local economy affect heavy-metal concentrations in soil. Natural factors, including parent material (rock, sediment, or organic matter), climate, and soil processes, influence the content of heavy metals in soil, while human activities such as industry, agriculture, and transportation further contribute to their presence and impact. Physical, chemical, and biological factors affect their bioavailability (Timothy and Williams, 2019).

One way to study heavy metal pollution is by analyzing bricks from various time periods taken from old buildings and monuments. This can provide insight into the levels of pollution during different eras (Shrivastav, 2001).

1.2.1. Source of Metals Pollution

Since the creation of the Earth, heavy metals have been in the crust naturally (Masindi et al., 2021). Geologic parent material or rock formations are the primary natural source of heavy metals. These contaminants can have both natural and human causes as their sources. Natural sources include volcanic eruptions and interactions with rocks that often include metals from the environment (Masindi et al., 2021).

Volcanoes can contribute to these pollutants by periodically releasing large amounts of gas, erupting violently, or emitting very little gas continuously, like when geothermal activity and degassing (Mather, 2015). Natural phenomena like volcanic activity, metal corrosion, metal evaporation from soil and water, sediment re-suspension, soil erosion, and geological weathering further exacerbate heavy metal pollution (Nagajyoti et al. 2010; Briffa et al., 2020). Wind-blown dust and volcanic eruptions are important when considering the number of heavy metals in ecosystem inventories and budgets. The wind-borne dust originating from arid regions like the Sahara contains significant concentrations of Fe and smaller quantities of Mn, Zn, Cr, Ni, and Pb (Ross, 1994). The key findings indicate that the concentrations of heavy metals are affected by the direction of the wind and decrease as the distance from pollution sources increases. Additionally, these concentrations change depending on various factors, including wind direction, atmospheric conditions, human activities, and the slopes of the terrain (Punia, 2021; Swain, 2024; Chu et al., 2023). The soil's heavy metal concentrations showed variation based on both elevation and distance from the mine. These concentrations notably decreased within a 2 km radius and were mostly controlled by the topography rather than wind (Ding et al. 2017). The impact of components released from sea sprays and mists on the ecosystem. It is commonly acknowledged that it is often transported over long distances inland. Cu and Mn originating from marine sources have been identified in rainwater that enters terrestrial ecosystems. The phenomenon of 'bubble bursting' is a natural mechanism that releases Cd, Cu, Ni, Pb, and Zn into the air through sea salt particles (Pacyna, 1986). Forest and prairie fires are one of the sources of airborne emissions of heavy metals. During such fires, metals like Hg, Pb, and Cr are released into the atmosphere as the vegetation and soil burn (Kristensen et al., 2012).

In addition to these atmospheric and environmental factors, the natural composition of the soil itself plays a crucial role in heavy metal presence. Soil formation primarily occurs from sedimentary rock; however, it is not a significant contributor of heavy metals due to its limited susceptibility to weathering. Nevertheless, a significant quantity of Mn, Co, Ni, Cu, and Zn is contributed to the soils by various igneous rocks, including olivine, augite, and hornblende. Among sedimentary rocks, shale shows the most elevated levels of Cr, Mn, Co, Ni, Cu, Zn, Cd, Sn, Hg, and Pb, followed by limestone and sandstone (Nagajyoti et al. 2010).

The term "human-made sources" encompasses various procedures used in industry, agriculture, and daily life. These procedures include the burning of fossil fuels and processing of metals in industry, the use of pesticides in agriculture, and the use of detergents and waste disposal in daily life. Anthropogenic activities, urbanization, and industrialization are primary drivers of increasing heavy metal pollution, with mining operations being a major source of emissions (Duruibe et al., 2007; Zamora-Ledezma et al., 2021). The research conducted by Loska et al. (2004) demonstrates substantial pollution of agricultural soils in Suszec, Poland, with heightened concentrations of Pb, As, and Hg. This contamination is mostly attributed to the impact of nearby industrial operations and emissions occurring in the surrounding region. Human activities, particularly those related to metal extraction, smelting, foundries, and various metal-based industries, are the main culprits behind heavy metal pollution. The contamination also arises from metal release through excretion, garbage dumps, livestock and chicken dung, road runoff, and landfills. An investigation conducted in the Southwestern region of Nigeria discovered substantial levels of hazardous heavy metals such as Cr, Cu, Fe, Pb, and Zn in soil samples taken from dumpsites. These concentrations were particularly high along a distance of 70 meters downhill from the center of each dumpsite (Awokunmi et al. 2010). In terms of horizontal distributions, the concentrations of metals in soil often drop as the distance from the mine or smelter site increases. This decline normally follows an exponential or negative-power decay curve. Typically, the metal concentrations in the topsoil layers were considerably higher than the background levels. Background levels refer to the values found several kilometers away from the smelter operations (Kabir et al. 2012). Unintentional discharges from accidents like shipwrecks, oil spills, mining operations, and fires as well as planned uses like the use of biocides for vector control and waste disposals such industrial effluents and sewage disposal.

Additionally, the use of heavy metals in herbicides, insecticides, fertilizers, and related products contributes to heavy metal contamination in agriculture. Although there has been a decrease in consumption, the yearly application of tons of industrial pesticides, which include neurotoxic organophosphates, still presents substantial health and environmental hazards, especially in undeveloped and emerging nations. Widespread soil arsenic contamination has been caused by the inclusion of heavy metals in certain herbicides, as well as the past utilization of arsenical pesticides (Li et al. 2016). According to Nagajyoti et al. (2010), fertilizers, particularly those containing phosphate (Alengebawy et al. 2021), may include different concentrations of heavy metals such as Cd, Cr, Ni, Pb, and Zn. Cd is of special significance due to its tendency to accumulate in plant leaves and subsequently infiltrate the food chain. Animal manure and sewage sludge serve as reservoirs of Mn, Zn, Cu, Co, Cr, Pb, Ni, and Cd in soil. The use of soil additives such as compost and nitrate fertilizers lead to an elevation in soil's heavy metal content, with liming having a more substantial impact than nitrates. Pesticides, particularly those employed in orchards, can lead to soil pollution with metallic elements such as Cu, As, Pb, Zn, Fe, Mn, and Hg. Irrigation can potentially transfer heavy metals such as Pb and Cd into the soil, depending on the water source.

As a result, heavy metals are now found everywhere in the environment, including the atmosphere, seas, and soil. Multiple avenues exist for pollution to enter the hydrosphere, lithosphere, and atmosphere. Oliva et al. (2007), study emphasized the significance of air deposition in the buildup of metals in soil. Temperature, surface water movements and directions, air mass circulation, and wind speed are a few examples of the variables that have an impact on the transport of heavy metals and other pollutants. The distribution and movement of these contaminants are additionally influenced by additional factors such as partition coefficient, polarity, vapor pressure, and molecule stability (Briffa et al., 2020). Lastly, Smaller particles, which have a greater surface area, often contain higher levels of metal(loid) substances. This contributes significantly to pollution through the deposition of particulate matter and affects the distribution and availability of heavy metals in oil aggregates formed by biogeochemical cycles (Adnan et al. 2022).

1.3. Metals In Soil Environment

Atomic densities more than 4 g/cm3 are distinguished as heavy metals and metalloids from other substances (Ali et al., 2022). Heavy metals have an influence on the physio-chemical characteristics of soil, including pH, organic matter content, and ion exchange capacity. These factors subsequently impact microbial development and density. Hu et al., (2018), observed that soil pH and organic content primarily affect the accumulation and transportation of heavy metals in soil, resulting in heightened environmental dispersion.

Certain metals can have a negative impact on soil fertility by reducing the accessibility and diversity of nutrients. The presence of organic material in soil serves as an environmental reservoir for heavy metals, facilitating the formation of stable complex compounds that decrease the leaching of metals and their absorption by plants. It augments the soil's ability to exchange ions, hence improving the adsorption of heavy metals. The accumulation of organic matter is crucial for the regulation of soil erosion and the preservation of agroecological sustainability (Ali et al., 2022). There is a strong relationship between the presence of OM and Zn levels in the soil, as OM helps retain Zn and prevents it from being lost through leaching. The leaching capability of Cu is limited because it forms complexes with soil OM, while the mobility of Zn is also influenced by its strong bonding capacity with organic complexes (Fageria, 2009; Codling et al., 2008). Over a span of 10 years, a research investigation analyzed the impact of a solitary application of compost made from municipal garbage on the soil covering a landfill. The study revealed noteworthy elevations in the amount of metals, particularly Cu, Zn, and Pb, in the enriched top layer of soil (Businelli et al. 2009).

The bioavailability, toxicity, and mobility of toxic metals and metalloids in soil are determined by their speciation, influenced by interactions with ligands, soil characteristics, pH, organic matter, and particle size. Cu, Cd, Zn, Pb, Hg, As, Ag, Cr, Fe, and Pt group elements are among the substances included in heavy metal category. These chemicals have a propensity for bioaccumulation since they are not biodegradable. Bioaccumulation is the process by which their concentration gradually rises over time within living organisms, in other explanation heavy metals are dangerous because they tend to build up in our systems after intake or inhalation (Zamora-Ledezma et al., 2021). Dotaniya et al., 2020, emphasize Pb as a harmful metal that reduces soil fertility, limits nutrient accessibility, and inhibits microbial diversity. The decrease in microbial activity results in a deceleration of organic matter decomposition and a decrease in soil fertility. Pb has a unique influence on earthworms, resulting in death and stunted growth. It also affects important soil characteristics such as pH and cation exchange capacity (Vega et al., 2010). Soil organic carbon and matter, especially humic substances, play a vital role in the absorption of metals and the stabilization of soil. The complexity and ever-changing nature of soil habitats make it difficult to analyze soil bacteria, even though they play a crucial role in converting plant biomass and mitigating pollution (Adnan et al. 2022).

It is known that poisonous metals can harm plants by disrupting their morphology and physiological functions by excessive accumulation. This includes unfavorable outcomes like slowed growth rates, slowed stomatal movement, nutritional imbalances, cellular injury, and photosynthesis inhibition (Pichhode and Kumar, 2016). The toxicity can be classified as primary, which occurs naturally, and secondary, which is more commonly caused by human activities. It changes depending on the characteristics of the soil and necessitates a thorough examination of various metals in plants to comprehend their combined impacts (Ahmad et al. 2022). Hg, Cd, Pb, and As are not necessary for metabolism or other fundamental biological processes. As a result, they have been recognized and designated among the top 20 hazardous compounds by the Agency for Toxic compounds and Disease Registry and the United States Environmental Protection Agency (Qin et al., 2021). Studies highlighted the challenges associated with the removal of heavy metals from contaminated soil (Ali et al., 2022).

1.3.1. Topsoil Pollution in Baltic and Scandinavian Countries

Soil contamination in the Baltic region is caused by a mixture of industrial operations within the region and the movement of pollutants from Western Europe through the atmosphere. In Estonia, particularly in the northeastern region, there is noticeable pollution originating from local sources such as the Narva and Kohtla-Järve power plants, which utilize oil-shale, as well as the Kunda cement factory. The presence of Cd, Fe, and V in the area is increased due to these factors (Ruhling et al. 1992; Nikodēmus and Brūmelis, 1994). Latvia's topography and meteorological circumstances have a significant influence on the situation in the country. The western slopes of uplands, especially in Western Latvia (Nikodēmus and Brūmelis, 1994), have higher levels of pollutants. This variation is influenced by factors such as altitude, precipitation, and wind direction. Geochemical mapping conducted in Tallinn, Estonia, has identified substantial irregularities in the distribution of heavy metals such as Pb, Zn, Cu, and Cr. Notably, there is a high concentration of Cu and Zn in the upper layer of soil, whereas lead Pb is predominantly present as carbonates. The findings suggest that there is a significant likelihood of metals being able to move and pose an ecological risk, especially in the case of Pb, Zn, and Cu. These metals are easily taken up by plants and play a role in biogeochemical cycles (Bityukova, 1993). Both Estonia and Latvia are significantly affected by pollution, In Latvia, 56 out of the 242 recognized contaminated regions are impacted by heavy metals (Burlakovs and Vircavs, 2012), from major industrial sectors, such as the oil-shale burning area in Narva, Estonia, and the steel industry in Liepāja, Latvia. The activities mentioned have a direct effect on the soil quality in the region, resulting in precipitation that is neither acidic nor alkaline (Nikodēmus and Brūmelis, 1994). This reflects the influence of urban planning and pollution control measures. A study was conducted in Estonia to examine the effects of power plants that burn oil shale on coniferous ecosystems. This was done by evaluating the levels of heavy metals in different parts of six coniferous stands. The investigation revealed that heavy metals were predominantly concentrated in soil organic layers, with the exception of Zn and Cu, which exhibited the highest concentrations in fine roots. These findings indicate that there has been a gradual buildup of heavy metals in the soil, which can then be transferred to the above-ground biomass of the trees (Napa et al. 2017).

The presence of heavy metals such as Pb and Zn in the top layer of soil can be linked to multiple sources, including the use of fertilizers and the deposition of pollutants from the atmosphere. In the northern countries, including Latvia and Lithuania, the occurrence of minerals such as Magnesium, which is concentrated in areas with dolomitic rocks, suggests that soil composition is influenced by geological factors. The difference in Rb concentrations implies differences in soil age and weathering mechanisms (Reimann et al. 2000). In these countries, application of de-icing agents, specifically sodium chloride, has been proven to enhance the dispersion and leaching of organic substances in soils, hence increasing the transportation of metals such as Cd, Cu, Pb, and Zn. Galvanized structures release substances that contaminate the soil, which is known as leaching (Werkenthin et al., 2014).

Cu-Ni industrial complex has a significant impact on the soil in Scandinavia and surrounding areas, causing high levels of pollution in the upper layers of organic soil, particularly with regards to Cu. The contamination is affected by variables such as human-induced deterioration of soil, reduction of organic matter, and competitive interactions among chemical components (Kashulina 2017; 2018). The research conducted near a smelter in Finland discovered a rise in Cu buildup in both soil and vegetation as the distance from the smelter decreased. This had a detrimental effect on soil bacteria and the overall health of the surrounding forest (Helmisaari et al. 1995). This emphasizes the enduring ecological consequences of heavy metal contamination in forest ecosystems. A comprehensive assessment conducted across Norway has identified significant concentrations of heavy metals in surface soils. The southern regions, in particular, show greater levels of heavy metals. This is mostly attributed to the long-distance transportation of metals from Europe and the emissions from local metal smelters. The surface soils of Southern Norway exhibit significant contamination with metals such as Zn, As, Cu, and Pb. This contamination is mainly caused by deposition from the atmosphere and is further influenced by the closeness to the coast (Steinnes et al., 1997; Steinnes et al., 1989). According to Andersson et al. (2010) The discontinuation of pollution sources in urban areas has resulted in a notable reduction in the levels of heavy metals in surface soils. All elements, with the exception of lead, have exhibited a statistically significant decline in concentration during a span of ten years. The soil investigation conducted in the Stockholm Municipality revealed substantial levels of heavy metals, primarily concentrated in the city center and old industrial zones, in sharp contrast to the rural areas. This discovery highlights the magnitude of soil pollution in metropolitan areas (Linde et al. 2001; Linde, 2005). In South Sweden focused on metal emissions in road traffic environments. The study analyzed a total of 148 topsoil samples and observed a notable rise in metals such as Cu originating from brake linings (Hjortenkrans et al. 2006). A study conducted in Uppsala, Sweden, found that the levels of metals in playground soils are mostly determined by soil features and the age of the site, rather than by land use. This emphasizes the significant role of historical variables in urban soil pollution (Ljung et al. 2006).

1.3.2. Topsoil Pollution Studies From Other Countries

The study conducted by Aradhi et al. (2023) investigated the spatial distribution of heavy metals present in the top layer of soil surrounding oil and natural gas drilling sites in Andhra Pradesh, India. The research emphasized the direct influence of industrial operations on the overall quality of soil. According to Alengebawy et al. (2021), topsoil contamination, caused by the buildup of heavy metals and pesticides from both natural and human sources, poses a threat to agricultural output. Environmental toxicants like lead, copper, and zinc are particularly concerning as they have the ability to harm agricultural ecosystems. Moreover, a study conducted on soil heavy metal pollution in China unveiled those locations across the country exhibited excessive levels of heavy metals such as mercury, arsenic, copper, lead, and zinc, beyond the established standards (Wu et al. 2022). This poses a substantial environmental issue. Another study corroborated these findings by documenting extensive lead contamination in Chinese soils, so underscoring the seriousness of heavy metal pollution in fast-advancing areas (Adnan et al. 2022).

Table 1. Regulatory Levels for Soil Metals Established Between Different Countries (mg/kg) (Kabir et al. 2012)

Order	Country	Pb	Zn	Cu	Cr	As
1	Australia	—	200	60	50	20
2	Japan	—	—	125	—	15
3	Taiwan	100	300	150	200	20
4	Turkey	150	500	100	—	—
5	EU	300	300	140	150	—
6	Netherlands	150	500	100	250	—
7	Spain	300	450	210	150	—
8	Germany	100	300	100	100	20
9	France	100	300	100	150	20
10	UK	550	280	140	600	10
11	USA	150	300	45	212	5.6
12	Canada	500	500	100	250	—

The regulatory levels for soil metals vary significantly across countries, highlighting differences in environmental standards. For example, Table 1 outlines the soil metal limits for elements like Pb, Zn, Cu, Cr, and As, comparing regulatory levels in countries such as Australia, Japan, and the USA (Kabir et al. 2012). These variations in acceptable metal concentrations underscore the need to consider both local environmental conditions and international standards when evaluating soil contamination.

A comprehensive analysis of soil contamination in European metropolitan areas was carried out, incorporating 174 scientifically evaluated studies completed over 143 urban locations in 29 countries in Europe. The comprehensive examination demonstrated that the presentation of urban soil metal data exhibits significant diversity throughout Europe (Binner et al. 2023). The presence of heavy metals, including arsenic, cadmium, chromium, copper, mercury, lead, zinc, antimony, cobalt, and nickel, has been detected in the topsoil of the European Union. This study reveals certain regions where there is a significant risk of heavy metal contamination. It indicates that approximately 28.3% of the total surface area of the European Union has to be further evaluated for this danger (Toth et al. 2016). El-Amier et al. (2018) found that Cd, Cr, Pb, Ni, Co, and Fe were the most prevalent heavy metals in the Mediterranean coast, with anthropogenic sources being the main contributor. In addition, the spatial analysis of mercury levels in the top layer of soil in the European Union demonstrates that soil acts as a significant repository for metals released via both natural processes and human activities. Panagos et al. (2013) provided a comprehensive overview of the situation in Europe, reporting that municipal and industrial wastes are the main contributors to soil contamination, with heavy metals being a significant component. Mercury, in particular, is listed third by the US Government Agency for Toxic Substances and Disease Registry due to its considerable risk to health (Ballabio et al. 2016).

Table 3 shows from different countries concentrations levels of Pb, Zn, Cr, Fe and As levels. An analysis of the levels of elements in soil affected by various industrial operations yields valuable information about environmental pollution. The data exhibits substantial disparities in concentrations among various industries and localities. An example of this is the significant concentration of toxic heavy metals such as lead (Pb), and zinc (Zn) in regions connected to smelting and metal industries. This suggests a clear link between industrial activities and the buildup of metals in soil. The heightened concentrations of these elements, specifically in areas such as Kosovska Mitrovica (Kosovo) and Zhuzhou (China), indicate significant environmental consequences, potentially compromising soil well-being and posing hazards to ecosystems and human health. The fluctuations in concentrations are indicative of the heterogeneity in industrial procedures, the effectiveness of pollution mitigation measures, and the geological characteristics specific to each region. It is crucial to monitor and reduce these levels in order to effectively manage the environment and preserve public health.

Table 2. Summary of Trace Element Concentrations in Soil Affected by Diverse Industrial Activities (mg/kg).

Order	Industry Type	Location	Pb	Zn	Cr	As	Reference
1	Pb and Zn smelter	Zhuzhou, China	472	907	—	38.0	Li and Huang, 2007
2	Zn Smelter	Celje, Slovenia	865	230	—	37.0	Voglar and Lestan, 2010
3	Smelter	Wales, UK	136	183	—	—	Davies, 1997
4	Smelter	Reppel-Bocholt, Belgium	899	137	—	—	Cappuyns et al. 2002
5	Smelter	Kosovska Mitrovica, Kosovo	973	560	418	87.1	Borgna et al., 2009
6	Metal industry	Potenza, Italy	87.7	62.8	—	18.3	Medici et al., 2011

Order	Industry Type	Location	Pb	Zn	Cr	As	Reference
7	Metal industry	Annaba, Algeria	53.1	67.5	30.9	—	Maas et al., 2010
8	Metal industry	Karak ind. Estate (S), Jordan	15.7	4.9	—	—	Al-Khashman, 2004
9	Copper smelter, steel industry	Port Kembla, Australia	20	42.0	12.0	3.20	Martley et al., 2004
10	Metal recycling, steel production	Murcia, Spain	18.3	21.8	19.4	—	Shallari et al., 1998.
11	Ferrous and non-ferrous metal smelters, iron and steel manufacturing,	Thessaloniki, Greece	24.2	68.3	98.7	36.1	Voutsa et al., 1996.

Order	Industry Type	Location	Pb	Zn	Cr	As	Reference
	metal recovery, manganese ore treatment, scrap, metal incineration						
12	Copper mines	Canchaque, Peru	156	235	—	279	Bech et al., 1997.
13	Smelters, mining	Elbasan, Albania	80	61.0	491	—	Shallari et al., 1998.
14	Smelters, mining	Rubik, Albania	135	250	256	—	
15	Smelters, mining	Munelle, Albania	103	111	91.0	—	
16	Smelters, mining	Tharsis, Spain	85.0	92.0	—	37.0	Chopin and Alloway, 2007.

Order	Industry Type	Location	Pb	Zn	Cr	As	Reference
17	Petrochemical	Tarragona, Spain	29.5	—	20.4	4.71	Nadal et al., 2007.
18	Petrochemical	Almowra, Spain	—	—	—	—	Soriano et al., 2012.
19	Petrochemical	Tarragona, Spain	36.3	—	13.8	5.50	Nadal et al., 2007.
20	Petrochemical	Catalonia, Spain	37.8	—	16.5	6.51	Rodríguez et al., 2004.
21	Chemical	Tarragona, Spain	24.5	—	14.8	6.87	Nadal et al., 2009.
22	Chemical	Valencia, Spain	22.2	—	17.5	6.24	Kashem and Singh, 1999.
23	Sulphuric acid-producing industry	Dhaka, Bangladesh	1.03	126	—	—	

Order	Industry Type	Location	Pb	Zn	Cr	As	Reference
24	Ceramic industries	Dhaka, Bangladesh	28.6	287	—	—	
25	Ceramic industries	Castellón, Spain	229	—	29.7	—	Deepali and Gangwar, 2010.
26	Tannery industries	Dhaka, Bangladesh	68.1	290	—	—	Kashem and Singh, 1999.
27	Tannery industries	Haridwar, India	—	—	744	—	Deepali and Gangwar, 2010.
28	Tannery industries	Peshawar, Pakistan	4.66	2.38	29.9	—	Tariq et al., 2006.
29	Tannery industries	Damascus, Syria	17.0	103	57.0	—	Moller et al., 2005.
30	Textile industries	Dhaka, Bangladesh	56.4	207	—	—	Al-Khashman and Shawabkeh, 2006.
31	Textile industries	Haridwar, India	191	—	568	—	Kashem and Singh, 1999.

Order	Industry Type	Location	Pb	Zn	Cr	As	Reference
32	Cement factory	Qadissiya, Jordan	55.0	45.0	22.2	—	Jordan et al., 2009. Li and Huang, 2007.
33	Battery manufacturing	Baoji, China	268	169	100	—	
34	Textile, plastic, furniture, industries	Thane - Belapur, India	—	191	521	—	Krishna and Govil, 2005.
35	Chemicals, dyes, textile, paint industries	Rajasthan, India	293	—	240	—	
36	Paint, battery, textile, milling and chemical industries	Lagos, Nigeria	—	—	—	—	Fakayode and Onianwa, 2002.

Order	Industry Type	Location	Pb	Zn	Cr	As	Reference
37	Ceramic, paper products, petrochemical industries	Belgrade, Serbia	37.7	103	—	—	Slavkovic et al., 2004.
38	Chemicals, pharmaceutical, batteries	Hyderabad, India	65.0	313	433	33.0	Govil et al., 2008.
39	Chemicals, furniture, printing, leather, textile	Kayseri, Turkey	74.8	112	29.0	—	Tokahoglu and Kartal, 2006.
40	Brewery, steel/ metal works, paints, pharmaceuticals,	Lagos, Nigeria	143	247	26.6	—	Fakayode and Onianwa, 2002.

Order	Industry Type	Location	Pb	Zn	Cr	As	Reference
	packaging, food processing, textiles and plastic products industries						
41	Tannery, textile, fertilizer, rerolling and casting, chemicals paints plastics	Jajmau, Unnao, India	38.3	159.9	265	—	Gowd et al., 2010.

1.3.3. European Union's Soil Health Initiatives

A comprehensive strategy was formed by the European Union to address soil health concerns, aiming to achieve optimal soil conditions by the year 2050. This objective aligns with the overarching objectives of the European Green Deal, which emphasizes the importance of soils in addressing climate change, promoting food security, conserving biodiversity, and sustaining the overall ecological well-being. The EU's strategy includes measures to protect and revitalize soils, ensure their sustainable use and management, and build a framework to improve soil monitoring across the EU (European Commission, 2023b; 2023c).

The adoption of the new Soil Monitoring Law on July 5, 2023, represents a noteworthy advancement in EU soil policy. This legislation aims to create a strong and consistent system for monitoring soil in all member states of the European Union. The primary goals of the law are to address significant soil challenges, including erosion, floods, landslides, depletion of soil organic matter, pollution, and biodiversity loss. The focus is on promoting sustainable soil management and requiring member states to identify and address potentially polluted areas, therefore contributing to the goal of achieving a toxin-free environment by 2050 (European Commission, 2023d).

In addition, the EU Soil Observatory has created a soil health dashboard to evaluate the state of soil health throughout Europe. This tool assesses the magnitude of infertile soils and the underlying mechanisms of their deterioration. Based on this dashboard, a substantial number of soils in the European Union are in a deteriorated condition. Soil deterioration primarily manifests as the reduction of soil organic carbon, the decrease in biodiversity, and the increased vulnerability of peatlands to degradation. According to the European Commission (2023a), this instrument assumes a vital role in highlighting soil health concerns and guiding policy and research initiatives.

Notwithstanding these endeavors, the European Union has significant challenges in upholding soil health, since over 60% of European soils currently exist in an unfavorable condition and are experiencing degradation due to unsustainable activities and environmental limitations. The EU's initiatives encompass the establishment of knowledge repositories, provision of assistance for soil investigation, promotion of awareness on the significance of soil, and active participation in global endeavors to address land and soil deterioration. The EU Soil Observatory functions as a data platform for monitoring advancements towards these goals (European Commission, 2023c). Although there is currently no enforceable comprehensive framework dedicated to soil preservation at the EU level, several existing legislations indirectly promote soil well-being. These encompass the Nature Restoration Law, Environmental Liability Directive, Industrial Emissions Directive, Environmental Impact Assessment Directive, and the Common Agriculture Policy, among other others. These regulations contribute to the reduction of contamination, the mitigation of greenhouse gas emissions, and the prevention of various environmental hazards (European Commission, 2023c).

1.3.4. Lithuanian Soil

The distribution of soil elements in Lithuania is influenced by factors such as soil origin, glacial history, and land use. Albeluvisols (≈30 % of the country), Luvisols (27 %) and Cambisols (13 %) dominate, each distinguished by characteristic textures (from sands to heavy loams), organic-matter content and pH (Buivydaite, 2005).

Arable soils tend to have higher levels of microelements compared to forest soils due to techniques such as fertilization and liming, which increase the concentrations of elements like Ca and Sr, particularly in regions of intensive cultivation (Gregorauskienė, 2012). Acidification (driven by declining liming and acid rain) affects roughly 16 % of arable soils (pH KCl < 5.5) (Buivydaite, 2005). Regional variations in soil composition exist throughout Lithuania, influenced by differences in soil parent material, climate, and historical glaciation patterns. For example, glacial deposits and fluvial sediments in different parts of the country contribute to variability in element concentrations, with certain areas exhibiting higher or lower levels of elements depending on the geological composition (Kadunas et al. 1999; Gregorauskienė, 2012), which includes more weathered and nutrient-depleted sediments (Kadunas et al. 1999).

The eolian sands found in Lithuania are primarily associated with the deposition of clastic materials, while limnoglacial sands contain higher concentrations of lithogenic elements, including Fe, Ti, and Zr. These sands originate from glacial and post-glacial processes, which strongly influence the mineral associations found in the soil.

Forest soils demonstrate strong associations between soil pH levels and the availability of microelements. Acidic conditions in forests encourage the accumulation of heavy metals like Pb, Cd, and Zn in the upper soil horizons (Gregorauskienė, 2012). This highlights the importance of understanding the regional distribution of soil elements, as it can vary significantly depending on the local soil composition, climate, and anthropogenic influences (Kadunas et al. 1999; Gregorauskienė, 2012).

In addition to regional and surface-level influences, the vertical distribution of elements within soil profiles also reveals important trends. Gregorauskienė and Kadnas (2006) examined the vertical distribution of elements in 53 soil profiles across Lithuania. Their study showed that element depletion, especially of Ca and Mg, is widespread due to leaching. The surface A-horizon is enriched in elements such as Pb, Sn, and Mn, reflecting anthropogenic and biogenic influences. In contrast, elements like Fe, Al, Li, and Zn accumulate in the B-horizon, linked to clay minerals and soil-forming processes. Resistant elements such as Zr are also found in higher amounts in upper layers, while Sr, and Na remain relatively stable throughout the profile. These patterns reflect the impact of natural processes and human activity on Lithuanian soils.

Lithuania's geochemical landscape is influenced by air masses that carry pollutants from industrialized regions of Europe. The air masses that bring in dust from industrialized regions of Europe contribute to the increased levels of contaminants such as Pb and Zn in Lithuanian soils. These pollutants are carried by air currents originating from the Atlantic Ocean and other industrialized regions, especially in winter when snowfall plays a crucial role in the deposition of airborne contaminants (Kadunas et al. 2005; Gregorauskienė, 2012). Heavy-metal concentrations in topsoils remain largely below EU threshold values but show enrichment in loamy and clayey textures (e.g. Zn up to 60 $mg \cdot kg^{-1}$, Cu up to 56 $mg \cdot kg^{-1}$), whereas pesticide residues are rarely found above detection limits (Buivydaite, 2005). This atmospheric deposition contributes to the long-term accumulation of heavy metals in both forest and agricultural soils, particularly in the surface layers, which are more vulnerable to contamination (Gregorauskienė, 2012).

1.3.5. Geochemical Profiles and Element Distributions in Lithuanian Soils

The behavior and distribution of elements such as As, Cr, Cu, Pb, Rb, Sc, Sr, Zn, and V in Lithuanian soils show complex relationships with soil composition, pH levels, and human activities. As exhibits mobility in both acidic and neutral-alkaline soils, particularly in soils rich in clay. The concentration of Cr is directly related to the amount of clay present, as well as the presence of Fe-Mn oxides and CaCO3. The distribution of Cu is influenced by the presence of clay, carbonates, and iron hydroxides, whereas Pb is associated with agricultural materials and is evenly distributed across different soil types. The use of potassium fertilizers affects the distribution of Rb, while Sc is linked to Al and Ba in clay fractions. Sr, part of the radioactive carbonate group, is influenced by the amount of carbonate and clay content. Zn distribution is governed by geochemical characteristics and mineral composition, while V distribution is affected by the presence of soil mull and humus (Kadunas et al., 1999; 2005).

Specific areas in Lithuanian soils demonstrate a high concentration of these elements. The Mūša-Nemunėlis and Venta's central plain have elevated levels of arsenic, but the Nemunas lowland, Šūduva, and the Mūša-Nemunėlis lowlands display higher amounts of chromium. The Nemunas and Šūduva lowland soils exhibit elevated median concentrations of copper, whereas the Mūša-Nemunėlis Lowland displays high median amounts of rubidium. The Mūša-Nemunėlis Lowland exhibits the highest concentration of scandium. The Mūša-Nemunėlis and Sudovian lowlands exhibit elevated median quantities of strontium. The Žemaičiai, Eastern Lithuanian Highlands, and Nemunas Lowland exhibit heightened levels of Zn concentrations. Vanadium exhibits the greatest median concentration in the lower Nemunas region, Sudovian plains, and Southern Lithuanian highlands. The distributions of Lithuanian soils are impacted by various factors, such as soil composition, industrial activities, and agricultural methods. These aspects demonstrate the intricate interaction between natural and human-induced elements in defining the geochemical landscape of Lithuanian soils (Kadunas et al., 1999; 2005).

1.3.6. Key Findings from Lithuanian Topsoil Studies

Agricultural practices, including the use of commercial fertilizers and manure from large farms, have contributed to soil pollution, with the use of commercial fertilizers increasing fourfold from 1960 to the late 1980s (Tumas, 2000). Since the 1980s, Lithuania has been facing substantial difficulties concerning soil contamination, predominantly caused by industrial activity and transportation. Research has identified the motor and railway industries as significant contributors to pollution, resulting in the accumulation of toxic metals such as Pb, Cu, Co, Cd, Cr, Ni, and Zn. This pollution affects not only the land but also the air, water, and living organisms (Baltrenas and Kliaugiene, 2003).

The dispersion of contaminants is significantly affected by meteorological conditions, which are strongly influenced by the Baltic Sea. The climate of Lithuania is characterized as a transitional maritime to continental climate, which allows for the easy movement of moist air masses and the transmission of pollutants from industrial areas around Europe (Nikodēmus and Brūmelis, 1994). In Lithuania, higher heavy metal deposition was observed in the western part compared to the eastern region, attributed to variations in air concentration and precipitation. The study observed seasonal fluctuations in heavy metal concentrations, suggesting that long-range air mass transfer plays a significant role in distributing pollutants across the region (Kvietkus et al. 2011). The air quality in Lithuania, even in places known for their exceptional cleanliness, is greatly influenced by the long-distance transportation of many contaminants (Milukaite et al., 1995).

Table 3. Cr, Cu, Pb and Zn concentrations levels from Lithuanian districts (mg/kg) (Baltrenas and Kliaugiene, 2003).

District	Cr	Cu	Pb	Zn
Klaipeda	37.3	9.7	19.8	30.7
Varena	16.2	3.9	14.3	21.8
Vilnius	32.9	8.8	16	30.9
Ignalina	29.7	9.3	13	21.9
Kedainaniai	44.6	10.5	13.6	35
Panevazys	29.8	9	13.4	26.2
Radviliskis	38.4	9.9	15.9	26.8
Svencioniai	25.2	7.7	15.2	25
Telsia	38.3	11.2	15	33.6
Siauliai	34.4	10.5	15.2	26

A study by Vasarevicius and Greicuite (2004) examined the presence and distribution of Zn, Cu, and Pb, which are common components of munitions, in the lower layers of soil (up to a depth of one meter) in Lithuanian military areas. The findings suggest that certain activities, such as military exercises, contribute to the accumulation of heavy metals in the environment. Additionally, Table 1 highlights the variations in Cr, Cu, Pb, and Zn concentrations across different Lithuanian districts, emphasizing the regional disparities in soil contamination. For example, the study shows that districts like Klaipeda and Radviliskis have notably higher concentrations of these metals compared to other regions (Baltrenas and Kliaugiene, 2003).

Specific urban areas like Vilnius and Kaunas have high levels of heavy metals due to industrial and traffic emissions (Nikodēmus and Brūmelis, 1994). The transportation system, including the motor and railway sectors, has been highlighted as a prominent source of pollution, making a substantial contribution to soil contamination. The contamination impacts diverse ecosystems, as toxins possess the capacity to propagate through ecological food chains. The Radviliskis railway station, as demonstrated by Baltrenas and Kliaugiene (2003), has elevated concentrations of Pb in soil samples. in Vilnius city revealed significant heavy metal contamination in soil particles, carried by surface runoff sediments from urban areas, particularly in zones with high traffic (Ignatavicius et al. 2017). This emphasizes the magnitude of soil contamination in the city. A study assessed heavy metal levels in soils and grass located near major roadways. The investigation revealed that the levels of heavy metals, particularly Pb, Cu, and Cd, were most elevated at a distance of 5 meters from the road and observed a decline in these concentrations as the distance from the roads increased. Elevated concentrations of specific metals were detected in proximity to the roadways (Matyžiūtė-Jodkonienė, 2007; Jankaitė et al. 2008; Grigalavičiene et al. 2005).

Soil pollution is worsened by the release of pollutants from local businesses, including cement manufacture and fossil fuel burning. The Mažeikiai oil refinery, located in Northwest Lithuania, is the largest oil refinery in the Eastern Baltic region. It releases a quantity of V emissions that is consistent with other significant sources in Scandinavia. Furthermore, the neighboring Naujoji-Akmene cement plant contributes to the pollution of the region with Cr (Ruhling et al. 1992). The radioactive fallout from the Chernobyl tragedy, which was affected by rainfall, also played a role in exacerbating this problem (Nikodēmus and Brūmelis, 1994). The study conducted by Jankauskaitė et al. (2008) investigated the correlation between the susceptibility of urban landscapes to chemical pollution and the presence of topsoil contamination. The findings indicate that locations with industrial, infrastructural, and old town characteristics are more likely to have polluted and sensitive sites.

During the period from 1995 to 2006, significant factors that contributed to changes in pollution levels in the soil of Vilnius were Ag, Sn, Cu, Zn, Mo, and Pb. Contamination levels diminished around former industrial sites, but they escalated in different public spaces. During the period from 2006 to 2011 in Vilnius, notable changes in the amounts of Mo, Ba, Sn, Cu, Pb, Zn, and Ni elements were recorded according to Vilnius Municipality (Vilnius Aplinka, 2023). A geochemical study conducted in Vilnius' Šnipiškės Eldership analyzed 517 topsoil samples and found extensive contamination. Hazardous amounts of elements such as Zn, Hg, and Pb were detected, particularly in the vicinity of central streets and industrial zones. The pollution has been linked to residential sources, industrial operations, and the manufacture of construction materials, particularly concentrated in locations near metal processors and regions with heavy vehicular activity (Kadunas et al. 1999).

Taraskevicius et al. (2008) and Kumpiene et al. (2011) discovered that industrial and infrastructural sites, together with preschool playgrounds in higher-altitude regions, display increased concentrations of contaminants such as Ag, Cu, Mo, Pb, Sn, and Zn. This is mostly attributed to emissions from urban industries and traffic. These investigations demonstrate a clear relationship between the intensity of urban activity and the levels of soil contamination. In addition to these findings, Gregorauskienė and Kadūnas (2013) examined the distribution of various elements in Lithuanian soil, revealing the complex relationship between natural and anthropogenic influences in determining soil composition. Milukaite et al. (2008) provided more evidence of this influence by establishing a connection between elevated levels of pollutants in particular urban areas and the presence of heavy traffic and commercial operations. Taraškevičius et al. (2013) conducted a study discovered that the most harmful chemical elements in soil were highly concentrated in the upper layers, including in places such as preschools. This highlights the significant impact of surface activities on soil health.

Dundulis (2006) states that the Lithuanian coastline area is predominantly composed of sandy soils, which are essential for construction purposes. The sands found in this area show a wide range of characteristics, including different origins, ages, and grain sizes. This diversity contributes to the formation of a complex geological structure that enables a variety of ecological and human activity.

1.4. Heavy Metals Human Health Effects

The release of heavy metals during mining and smelting operations is a major cause for worry in terms of both the environment and human health. Metals such as lead and arsenic present considerable dangers to human health through many means of exposure, including ingestion, inhalation, and contact with the skin. Children are especially susceptible to harm because of activities such as putting their hands in their mouths and their ability to absorb substances more quickly. Research has demonstrated increased concentrations of lead in the bloodstream of residents living in close proximity to smelting facilities, highlighting the significant consequences of these operations. Although copper and zinc are micronutrients that are necessary for the body, excessive amounts of these metals can have detrimental effects. The identification of soil pollutants and their sources is of utmost importance due to their direct impact on human health. Consequently, there is a necessity for the development of effective remediation methods to tackle these difficulties (Adnan et al. 2022). Briffa et al. (2020) has been noted that metals interact with nuclear proteins and DNA, causing site-specific harm. The two main forms of damage that might occur are "direct" damage and "indirect" damage. The metal causes conformational changes in biomolecules through "direct" damage. On the other hand, exposure to heavy metals causes "indirect" harm by causing the generation of reactive oxygen and nitrogen species, such as hydroxyl and superoxide radicals, hydrogen peroxide, nitric oxide, and other endogenous oxidants. Furthermore, it has been demonstrated that several signaling pathways are activated by HM. Numerous biological and physiological issues may arise as a result of this bioaccumulation. While some heavy metals are necessary for life and are referred to as essential elements, it's important to remember that an excessive buildup can be hazardous (Briffa et al., 2020).

With an average concentration of roughly 150 g/g in the Earth's crust, V is a widely dispersed metal. In regions where fly ash contamination is present, soil concentrations can range from 3-310 g/g and can even be significantly higher (up to 400 g/g). In global circulation, the ocean floor acts as the main long-term V store. V concentrations in coal and crude petroleum oils range widely, from 1 to 1500 mg/kg. V exposure can happen through the air, drinking water, or food (WHO, 2000). Both acute and chronic poisoning have occurred among V-using and industrial production workers. Clinical symptoms that are frequently described by those who have been exposed primarily entail irritating effects on the respiratory system. When vanadium pentoxide dust is inhaled over an extended period of time, it can cause symptoms such rhinitis, pharyngitis, bronchitis, chronic productive coughing, exhaustion, shortness of breath, and pneumonitis. Severe exposure can cause a variety of health problems, such as severe throat, nose, and eye irritation, nosebleeds, throat pain, dizziness, headaches, nausea, rashes, nervous system impairment, liver and kidney hemorrhage, tremors, paralysis, behavioral changes, cardiac disease, gastrointestinal organ inflammation, and tooth and tongue blackening (WHO, 2000; Briffa et al., 2020).

There is a lot of Cr in the crust of the Earth, and depending on its chemical state, it has different toxicities. Only the trivalent and hexavalent versions of the different chemicals demonstrate noticeable biological toxicity. Cr compounds are frequently employed in industrial processes like the extraction of chromite ore, the manufacture of pigments, the tanning of leather, the preservation of wood, and as anti-corrosive cookware additives. Industrial paints commonly include hexavalent Cr, and chromates, which are trivalent and hexavalent chromium salts, are made through mining, smelting, roasting, and extraction. These processes produce hazardous dust that could endanger human health. Workers in the Cr-producing business have a greater incidence of lung cancer, and industrial waste from that industry is a noteworthy source of land and water contamination (Kim et al 2015). According to Zamora-Ledezma et al. (2021), Cr exposure can cause ulcers, ulcerative colitis, liver and kidney damage, pulmonary congestion, and skin inflammation. Symptoms of acute Cr poisoning include gastrointestinal ulceration, nausea, vomiting, fever, diarrhea, vertigo, toxic nephritis, liver damage, coma, and sometimes deadly effects at larger dosages (often at 1-3g). Lung cancer, allergic contact dermatitis, eczema, gingivitis, mucous membrane irritation, bronchitis, liver and kidney illness, sinusitis, pneumonia, and repetitive skin contact are all problems that can be brought on by long-term exposure. Notably, rather than through food or skin contact, chromium is largely carcinogenic through breathing. In addition to additional health problems (Briffa et al., 2020) and hair loss, severe exposure can also cause these (Qin et al., 2021).

Fe is a naturally occurring element that is abundant and is present in many food sources, including soil and water. Its presence in the environment is crucial for maintaining human health since it helps to create hemoglobin when mixed with protein, Cu, and other elements (Albretsen, 2006). Additionally, Fe is essential for the production of myoglobin, which provides oxygen (O2)-rich muscular tissues. The National Academy of Sciences (NAS) advises children to consume 7 mg of iron per day as their dietary Reference Dose (RfD), stressing the significance of iron intake within this range. However, exceeding the recommended daily dose by more than 200 mg can poison people or possibly be fatal, especially in young children. According to Olawoyin et al. (2012), oral dosages of more than 100 mg/kg are often regarded as possibly fatal for all animals.

One of the first metals used by humans was Cu, a metal that occurs naturally in its metallic form as well as in a variety of ores and minerals (Stern et al., 2007). Cu concentrations in the air normally vary from a few nanograms to around 200 ng/m3, but they can reach as high as 5,000 ng/m3 when Cusmelting facilities are nearby. Cuexposure in the form of copper dust is increased for those who live or work close to Cu mines or businesses that use Cu metal (Karim, 2018). Hair loss, anemia, renal damage, and migraines are just a few of the health problems that Cu exposure can cause (Zamora-Ledezma et al., 2021). Cu poisoning symptoms, sometimes known as "metal fever," include flu-like symptoms, diarrhea, vomiting, eye irritation, dizziness, and mouth irritation. Due to necrosis, an initial dosage of Cu salts might result in acute gastroenteritis. According to Briffa et al. (2020), excessive Cu exposure can result in hepatocellular degeneration, necrosis, cytotoxic effects on red blood cells that cause hemolysis, hepatic and kidney disease, insomnia, anxiety, agitation, restlessness, and Wilson's disease, which has a variety of symptoms including lack of appetite, exhaustion, jaundice, speech impairment, and brain damage and can be fatal. The use of fertilizers and pesticides also contributes significantly to the promotion of soil copper buildup. The amount of fertilizers and pesticides used and the Cu concentrations in agricultural soils are clearly correlated, according to data on fertilizer and pesticide consumption. This environmental buildup of copper has been linked to lung cancer risk, weariness, dizziness, and kidney and brain impairment (Qin et al., 2021).

According to Duruibe et al. (2007), excessive Zn exposure can cause serious disturbances in biological processes and impede the ability to develop and reproduce. Pain, skin irritation, fever, vomiting, and anemia are all signs of excessive Cu exposure (Zamora-Ledezma et al., 2021). In addition, elevated Zn levels have been linked to nausea, vomiting, stomach discomfort, a drop in high-density lipoprotein (HDL) cholesterol, pancreatic issues, exhaustion, epigastric pain, Cu insufficiency, anemia, reduced immunological function, and neutropenia (Briffa et al., 2020). In addition, excessive Zn exposure has been connected to lung cancer risk, weariness, dizziness, and renal and brain damage (Qin et al., 2021). The negative effects highlight how crucial it is to control and monitor Cu intake in order to preserve optimal health and avoid related health issues.

In the past, port wine was sweetened with lead acetate to facilitate heating and storage. Over the past century, Pb emissions have had a significant negative influence on the environment, with more than half of these emissions coming from the use of gasoline. The majority of jobs that expose workers to inorganic Pb are those that involve mining, smelting, welding lead-painted metal, and working in battery factories. The danger of low to moderate exposure may also be present in the glass sector. Due to lead emissions, areas adjacent to Pb mines and smelters are particularly vulnerable to high levels of air pollution. Lead that is in the air can settle on water and soil, further contaminating the environment. Headaches, irritability, abdominal pain, and other nervous system-related symptoms are all signs of acute lead poisoning (Järup, 2003). The human body can suffer from a variety of negative outcomes from lead poisoning. This includes impeding hemoglobin synthesis and impairing the functioning of a number of organs, including the kidneys, joints, reproductive, and cardiovascular systems. Both the central nervous system and the peripheral nervous system may be permanently damaged as a result of it. Additionally, gastrointestinal and urinary tract injury may manifest as neurological problems and bloody urine, respectively. Pb poisoning can also have serious and long-lasting effects on the brain (Duruibe et al., 2007). According to Zamora-Ledezma et al. (2021), lead exposure is also linked to kidney and neurological system damage, mental impairment, and a higher risk of cancer. Other negative effects include high blood pressure, miscarriages, low birth weight and premature babies, stillbirths, kidney damage, brain injury, abdominal pain, pica, peripheral nerve damage, sperm damage, encephalopathic symptoms, disruption of hemoglobin synthesis resulting in iron deficiency, and cognitive impairment. Lead exposure in children can affect the growth of the brain and central nervous system, lower intellect, lower academic achievement, and shorten attention span (Briffa et al., 2020).

The generation of fossil fuel energy and the smelting of non-ferrous metals are the two main industrial activities that have a substantial impact on the amount of As in the air, water, and soil. The main anthropogenic cause of air pollution in this context is smelting activity. The production and use of arsenical wood treatments and insecticides are additional sources of contamination. As much as 1000 ng/m3 of arsenic can be found in the air in rural areas near industrial sources (Järup, 2003). Both inorganic and organic compound forms of metalloid arsenic occur, with the inorganic form being more dangerous. Inorganic pentavalent arsenic compounds dissolve in water to produce salts and weak acids known as arsenate (Kim et al., 2015). Although mining, ore smelting, and other industrial operations are frequently linked to human arsenic poisoning, the main natural source of As exposure is contaminated water. Because soil-borne arsenates can dissolve and contaminate groundwater, they can build up in aquatic species and eventually in rice, a common staple meal. Significant health concerns are associated with this intake pathway (Kim et al., 2015). Toxic exposure to inorganic arsenic can cause a number of health problems, such as irritation of the gastrointestinal and respiratory systems, changes to the skin, a decrease in the formation of red and white blood cells, an increased chance of developing cancer, infertility, miscarriages, cardiac troubles, and brain damage. DNA harm could also result from it. Contrarily, organic arsenic mostly damages nerves and disturbs the stomach without harming DNA or causing cancer (Briffa et al., 2020). In addition, it is known to result in significant hair loss (Qin et al., 2021). The brain is a target organ for the harmful effects of Arsenic and induces neurological problems (Bharti and Sharma, 2022). The negative health consequences highlight the urgent need to address As poisoning and its implications on the ecosystem as well as human health.

The study papers give a variety of viewpoints on the possible consequences of Zr and its compounds on human health. Zr and its compounds can be cytotoxic, resulting in immune system dysfunctions and pathological alterations in the skin and lungs, as stated by Bi (2015). In his discussion of the use of zirconium in biomedicine, Lee et al. (2010) emphasized the metal's biocompatibility and low toxicity, with only a few known adverse effects. According to Hartwig and Commission (2021), zirconium and its derivatives, with the exception of zirconium dioxide, are not sensitizers, and cutaneous absorption has little effect on systemic toxicity. Hichem et al. (2022) examines the possible toxicity of zirconia nanoparticles, pointing out contradictory results on their safety and their contribution to the production of reactive oxygen species. Finally, Chasapis et al. (2020), while not directly relating to Zr, focuses on the significance of Zn in human health and provides important background data on critical minerals. Overall, although zirconium and its compounds may have toxic effects on the skin and lungs as well as the immune system and other organs, the nuanced results and contrasting viewpoints highlight the need for more in-depth research to determine the true extent of zirconium's effects on human health.

Polyakova (2012) study highlights the existence of stable Sr in Arkhangelsk region water sources and its possible impact on human bone tissue. Similarly, Mao-jiang (2012) study focuses on the role of Sr in bone calcification, provided insight on its impacts on salt absorption and cardiovascular activity. Höllriegl and München (2011) study investigates the prevalence of Sr in the environment, both stable and radioactive, and its potential health impacts, notably on bone abnormalities and diseases. According to Zeneli and Daci (2014) research, persistent, moderate strontium exposure may lower levels of vital elements like Cu, Co, and molybdenum in human blood and serum, which may have a negative impact on health. In conclusion, these studies emphasize how important it is to understand the presence of strontium and its consequences on human health, including the increased risk of severe cardiac events, as noted by Glaspole in 2019.

Ta et al. (1989) and Akinfieva et al. (1989) both investigated the prolonged inhalation effects of Rb on the cardiovascular system but found no cardiotoxic effects. Usuda et al. (2014) conducted a risk assessment study on several Rb compounds and discovered that the biological effects of Rb differed depending on the counter anion, with anhydrous Rb fluoride exhibiting renal and liver toxicity. A study in Israel's Galilee, Coastal Plain, and northern Negev regions investigated the presence of Rb in the environment, revealing high concentrations of Rb in soil, soil solutions, rainwater, throughfall water, and plant litter leachates. This suggests rubidium accumulation in local soil and a semi-closed Rb cycle in the soil-plant system, which could affect local population health (Kot, 2018). If excessive Rb dust is inhaled, it can cause respiratory issues and discomfort. Long-term exposure can cause lung damage and other health problems.

It is especially harmful in the workplace because damps and fumes can be inhaled with air (Rim et al., 2013). It can cause respiratory problems such as coughing, wheezing, and shortness of breath. Scandium dust can also irritate the skin and eyes. Mostly research on scandium is done on animals.

According to Zulaikhah et al. (2020), mercury can cause glutathione depletion and oxidative stress, which can lead to cell damage, DNA damage, and disruption of antioxidant metabolism. Khatoon-Abadi and Khatoon-Abadi (2008) examine mercury bioaccumulation in the environment, particularly in fish, which can pose major health concerns to humans through eating. Man-tin (2012) focuses on atmospheric mercury pollution and its influence on human health, emphasizing the importance of addressing low-level atmospheric mercury contamination. Mercury, a naturally occurring toxic metal, contaminates the environment through natural processes and human activities. This has led to catastrophic outbreaks of mercury-induced diseases, particularly affecting the nervous systems in children, alongside health issues such as heart problems, kidney damage, and respiratory failures. The review underscores the importance of risk assessment and management in addressing the widespread challenges of mercury poisoning (Tchounwou et al., 2003). Boguszewska and Pasternak (2004) mentioned that mercury is a highly toxic heavy metal that is commonly encountered, and exposure to it can cause a variety of health problems, including immunological, sensory, neurological, motor, and behavioral dysfunctions.

1.5. Dust-Related Health Problems on a Global Scale: In-Depth Studies

Approximately 5 billion Mg of dust are released into the atmosphere annually from drylands (Nieder et al., 2018). The elderly, small children, and people with long-term cardiopulmonary conditions are among the vulnerable groups most at danger from dust health effects (Schweitzer et al., 2018).

In Goudie (2014) study, dust storms, which originate in drylands around the world, have an influence not only on the health of individuals living in these places, but also on downwind areas such as Phoenix, Athens, Beijing, and Tokyo. Dust storms, which commonly originate in the Sahara, central and eastern Asia, the Middle East, and areas of the western United States, can transport particulate matter, contaminants, and allergies across vast distances. These sources have an impact on regions ranging from South America to the United States. Dust from Asia's mid-latitude deserts, particularly Mongolia and the Tarim Basin-Taklamakan Desert, reaches North America, Greenland, and Europe. These sources include the United States' Great Basin, Argentina, southern Africa, and Australia. Former lake basins, such as the Bodele depression in Chad and depressions in Central Asia and Northern China, are major sources of fine, wind-blown mineral dust around the world (Nieder et al., 2018). Due to changes in land use and climate, the frequency of dust storms is altering in some areas, potentially aggravating their health consequences. These events transport a wide range of pollutants, including heavy metals and biological components, with particle loadings that can exceed acceptable levels. Dust storms can cause respiratory and cardiovascular problems, conjunctivitis, skin irritations, infectious infections, meningococcal meningitis, coccidioidomycosis, skin irritation, measles, and even traffic accidents (Goudie, 2014; Goudie, 2020). Smith and Lee (2003) paper cover soil dust and it has a substantial impact on air quality and is the second most significant source of primary particles after sea salt (Nieder et al., 2018). Soil dust is mostly caused by wind and agricultural operations, with construction, unpaved road use, and other causes also playing a role. Inhaling suspended soil particles can cause eye discomfort, respiratory diseases, pulmonary disease, and an elevated risk of lung and skin cancer. Workers in dusty environments, such as agriculture, construction, and mining, are particularly vulnerable. Increased soil dust in the air, on the other hand, can have an impact on the health of the general population. Soil dust generation, exposure, dust properties, and potential health effects are discussed, with the emphasis on fine particles posing a greater risk because they can reach the alveolar region of the respiratory system and cause harm (Smith and Lee, 2003).

Wang et al. (2020) and Karanasiou et al. (2012) research emphasizes the importance of analyzing the health implications of Sahara dust in Southern European countries such as Spain and Italy. Wang et al. (2020) investigated the impact of Sahara dust on air quality and human health using a global transport model and surface data, indicating variable dust concentrations across Europe. They found that dust exposure causes around 41,884 fatalities per year in 13 European countries, with Spain, Italy, and Portugal bearing the greatest burden due to high PM10 levels. Karanasiou et al. (2012) also mentioned the difficulty in comprehending health impacts associated with varied particle sizes, underlining the need for more research, such as identifying the chemical makeup and potential toxicity of coarse particles. To protect public health, these findings collectively advocate for stronger management of anthropogenic sources of particle pollution in areas significantly impacted by Sahara dust. Mitsakou et al. (2008) study measures the effect of dust transport on the quality of the air in different Greek cities that comes from Sahara. Data on PM10 concentrations from permanent monitoring stations are compared with SKIRON modeling-estimated dust concentrations for the years 2003–2006. The results show that, depending on the region, natural dust transport contributes more than 20% to annual exceedances of EU PM10 standards and more than 10% to PM10 levels overall. The study's secondary component determines the inhaled lung dose for inhabitants of different Greek regions. It demonstrates that although the upper respiratory tract is where mineral dust particles primarily settle, they can still have a major impact, much like when exposed to smoke or extremely polluted metropolitan environments during dust episodes.

Natural and man-made dust represent a severe health concern to rural inhabitants in Asia's arid regions. Agriculture, mining, and rapid industrialization all worsen the burden of dust and dust-borne diseases in the Central and Inner Asian drylands. The region's limited research suggests health dangers but does not quantify them. Dust dynamics on the Central Eurasian steppe are altered by anthropogenic influences such as the drying of the Aral Sea. In Khanbogd, Mongolia, a case study investigated the potential health hazards of large-scale mining, demonstrating varying dust exposure among households and town residents. Particulate matter was found in dust traps from sources other than mine. According to research, air dust from numerous sources can increase human exposure. In Central Asia, increased awareness of dust-related health risks reflects community concerns (Sternberg and Edwards, 2017). The presence of metals in attic dusts in Sydney, Australia, provides a historical record of air pollution. The study looks into the content and sources of metals in ceiling dust from homes in various industrial settings and roof types, and it emphasizes the possible health risks, particularly for children, when dust is disturbed and metals are discharged into the indoor environment (Davis and Gulson, 2005).

Ritz et al. (2019) give a comprehensive review of the important negative health effects connected with airborne pollutants, with a particular emphasis on fine dust. Their findings highlight the far-reaching influence of these contaminants on human health, from prenatal development to later stages of life, with a special focus on lung and heart problems. The study highlights major findings, such as a 7% increase in mortality for every 5 g/m3 of long-term PM2.5 exposure and a 25% relative risk increase in type 2 diabetes for every 10 g/m3 of PM2.5 exposure. Ritz et al. advocate for a significant reduction in existing EU regulations for these pollutants to accord with WHO recommendations.

Blondet et al. (2019) study in Cartagena-La Unión, Spain, examined four monitoring sites to explore air pollution and health concerns in an ancient mining district. Total metal content in dust, particularly Zn, Pb, As, and Cd, exceeded European limits in the mining and urban areas, and As exceeded norms along the shore. Health risk assessments that considered both total and bio accessible metal fractions revealed acceptable risks at most sites, with the exception of the mining tailings area, where high As and Pb content resulted in unacceptable cancer and hazard risks, emphasizing the importance of accounting for bio accessible fractions in risk assessment.

Dust storms in the Khuzestan region are caused by particulate matter input from both local and distant sources, posing a severe hazard to both indoor and outdoor quality as well as human health. The NOAA HYSPLIT model and meteorological data were utilized in this work to locate the sources of airborne particulate matter in Ahvaz City. The study discovered that average particulate matter concentrations exceeded the National Ambient Air Quality Standard during the cold season, underlining the importance of activities such as establishing green areas, reducing emissions, and international collaboration to manage dust (Goudarzi et al., 2018).

Susinski et al. (2009) analyze the environmental impact of a 190-hectare sterile landfill in Husnicioara, Mehedinti District. Solid suspension emissions from the dump pollutes the air. Dust emissions from vehicle and conveyor conveyance are a major issue for adjacent settlements and mines. Near coal resources and along the road between the mine and the dump, the Romanian air quality threshold for dust (0.5 mg/m3) is exceeded by a factor of ten. Levels near emission sources reach 4.0-5.7 mg/m3, exceeding the Maximum Allowable Limit (MAL) of 0.5 mg/m3 by 8-18 times, whereas readings at 100-150 meters are within this limit.

Staykova et al., (2022) study investigates the impact of anthropogenic pollution on air quality in the regions of Stara Zagora, Kazanluk Municipality, and Galabovo Municipality. Data from 2017-2018 show that PM10 levels in Stara Zagora, Ruzhena, and Galabovo were elevated, leading to an increase in respiratory disorders, particularly in children.

In Carlsen (2014) The study discovered links between anti-asthmatic medication use and prior exposure to PM10. There are increased links between emergency hospital contacts and volcanic sources of PM10, with PM10 impacts increasing following the Eyjafjallajökull eruption. Sensory organ irritation from volcanic ash but no impairment in lung function, with vulnerability in people with underlying lung illness, the elderly, and children. There is a dose-response pattern for respiratory symptoms and psychological discomfort following the eruption of Eyjafjallajökull, particularly among those reporting numerous physical symptoms in Iceland.

In order to estimate PM10 emission factors, Candeias et al. (2020) study collected road dust from suburban and urban streets in Viana do Castelo, Portugal. The results of the investigation showed that the components found in road dust differed between urban and suburban locations. While suburban areas had elevated levels of As, probably from agriculture, urban streets showed elements connected to transportation, such as Cu, Zn, and Pb. The majority of the minerals in the dust samples were irregularly aggregated quartz, muscovite, albite, kaolinite, and other minerals. Risks to human health were found, including the possibility of non-carcinogenic consequences in children from Zr consumption and a risk to human health from arsenic in suburban street dust. Furthermore, a study conducted to measure the health impacts of dust particles in Vilnius and Kaunas emphasized the potential health hazards linked to exposure to these pollutants, locally, traffic and household heating are regarded as the primary causes that significantly contribute to the deterioration of air quality. (Orru et al. 2012).

1.6. Review of Indoor Air Pollution in Educational Facilities and Its Influence on Child Health

Ensuring a safe and healthy environment for children to study and flourish is crucial, since their well-being is of utmost significance. The quality of the air within schools is an important part of this environment. The health of school-aged children can be significantly impacted by air pollution, both indoors and outdoors, as several studies have consistently demonstrated (Annesi-Maesano et al., 2013; Schwartz, 2004).

Schools are an important site for researching the effects of indoor air quality because kids spend a lot of time there (Fsadni et al., 2018; Zhang et al., 2019; Wallner et al., 2012; Mainka and Zajusz-Zubek, 2015). In classrooms, high concentrations of pollutants have been found, including particulate matter (PM2.5 and PM10), volatile organic compounds (VOCs), carbon dioxide (CO2), carbon monoxide (CO), and a variety of airborne microorganisms. According to Zhang et al. (2019), these pollutants are frequently brought in from outside sources like traffic pollution, and they are made worse by things like burning coal during the heating season (Wallner et al., 2012).

The negative impact of inadequate indoor air quality on children's respiratory health is one of the main worries (Fsadni et al., 2019; Zhang et al., 2019; Bates, 1995). Negative health consequences have been connected to indoor air pollution exposure (Wallner et al., 2012). According to Zhang et al. (2019), these consequences include worsening asthma, a rise in the frequency of respiratory symptoms, a decline in lung capacity, and an increased risk of atopy. Furthermore, research indicates that children who live close to schools have higher rates of wheeze, which emphasizes the connection between indoor and outdoor air quality and respiratory health (Fsadni et al., 2019). High levels of pollution and traffic jams are also major issues in developed countries, especially for children with asthma who do not have regular access to healthcare. This emphasizes how vital it is to address how air pollution affects children's health (Bates, 1995).

The SINPHONIE (School Indoor Pollution and Health: Observatory network in Europe) study, which was carried out in the study by Baloch et al. (2020), gathered information about the distribution of several indoor air pollutants (IAPs), physical and thermal parameters, and their correlation with different health symptoms. 5175 students were involved in this extensive study, which gathered data from 115 schools spread over 54 European cities and 23 nations. The findings showed a strong correlation between schoolchildren's health issues, such as upper and lower respiratory, ocular, and systemic problems, and their exposure to indoor air pollutants such PM2.5, benzene, limonene, ozone, and radon.

A study conducted by Liu et al. (2013) in China on the impact of present ambient air pollution on kindergarten-aged children's health found that children living in different environmental conditions were more likely to experience respiratory problems. 6730 Chinese children, ages 3 to 7, participated in the study, which was carried out in seven Northeast Chinese cities. According to the research, children's respiratory issues were more common among those who lived close to factories or chimneys, used coal-burning appliances, were around smokers, or had just undergone house renovations. A significant contributing factor to the development of respiratory morbidity was shown to be air pollution, especially in girls, who seemed to be more vulnerable than boys.

A 2015 study by Choo and Jalaludin found that comprehending indoor air quality in schools requires an understanding of geographic variances. Studies conducted, for example, in Malaysia have highlighted the effects on children's respiratory health of insufficient indoor air quality factors, such as interior temperature, ventilation rates, and the concentration of indoor pollutants (CO, VOCs, PM). Compared to their rural counterparts, children living in metropolitan areas are more likely to experience respiratory problems due to the concentration of pollution in these locations.

Younger children, such as those attending nursery schools, are particularly exposed to the impacts of indoor air pollution, according to another study by Mainka et al. (2018). Their young bodies and still developing immune systems make kids more vulnerable to the damaging effects of air pollution. Furthermore, younger children's exposure to indoor pollution increases by the amount of time they spend indoors. Negative health impacts, including higher rates of kindergarten and absences from school, have been linked to this exposure.

Pegas et al. (2010) did a study in Lisbon, Portugal, where they simultaneously assessed the levels of several pollutants indoors and outside. The findings of the study highlighted the detrimental effects of indoor sources and building characteristics on air quality, highlighting the necessity for enhanced ventilation rates and the utilization of low-emission materials and by Almeida et al. in 2011, Lisbon, discovered that coarse particle concentrations in classrooms exceeded those seen outside, with indications of student activity contributing to particle resuspension.

In 2015, Mainka and Zajusz-Zubek conducted a study in Gliwice, Poland, to measure the levels of particulate matter (PM) in naturally ventilated nursery schools. The results showed that PM concentrations were higher than should have been. In Korea, schools were constructed in the 1960s and 1970s. Comprehensive school building construction and renovation program attempted to improve teaching environment but unintentionally introduced materials that potentially harm indoor air quality. The researchers tested PM10 levels in schools and discovered that interior levels were higher than outdoor ones. Inadequate ventilation rates were also found to be a factor contributing to poor indoor air quality (Yang et al., 2009).

In a 2018 study, Stranger et al. evaluated the pollutant levels, indoor-to-outdoor ratios, and building characteristics of 27 primary schools in Antwerp, Belgium. According to the study, PM2.5 mass concentrations frequently exceeded outdoor values and did not show a linear association with them. The PM2.5 elemental compositions in classrooms differed from those outside, most likely as a result of dust re-suspension brought on by occupied rooms. The presence of carpets worsens elevated indoor PM2.5 levels in schools, particularly on lower levels, which raises concerns for children's health. The findings suggest that estimating children's personal exposure to these indoor air contaminants by using local outdoor data is not reliable.

Although the health of students has received a lot of attention, it's crucial to remember that adults who work in schools, such as teachers, are also exposed to indoor air pollutants. There isn't much data on this demographic, however, there could be negative health implications. Particularly allergic people seem to be more vulnerable to negative effects on their respiratory health (Annesi-Maesano et al., 2013). Poor indoor air quality in schools might cause illness and lower student performance. The prolonged exposure of children to air pollution, both indoor and outdoor, is a significant problem because of their vulnerability and duration of education. When evaluating the health impacts of human exposure to air pollution on children, indoor air quality in schools is crucial.

1.7. Interactions Between Indoor and Outdoor Air Quality: Correlation of Dust, Particulate Matter, and Pollutant Sources

Indoor air quality is linked to outdoor air quality since air can enter through doors, windows, and ventilation systems. Dust has a wide range of effects on plants, soil, clouds, air pollution, and aquatic and terrestrial nutritional and biogeochemical cycles. Field et al. (1992) investigated atmospheric pollutants at two sites during a recent air pollution incident in central London: one site was in a naturally ventilated office, while the other site was five meters from the roadside. Pollutant levels outside were up to five times greater than the average, and inside levels also increased. Volatile organic compounds (VOCs) on the other hand exhibited distinct behavior indoors, probably as a result of fog limiting their entry by acting as a scavenger. Six distinct buildings are examined in the study of Haug (2019), which is being carried out in Bergen, Norway, and measures variables like temperature, humidity, PM2.5, and PM10. Overall, the results show that these buildings' air quality complies with suggested guidelines. In older mechanically ventilated buildings, the study also reveals a more pronounced relationship between interior air quality and outdoor pollution, offering insights into the dynamics of indoor air quality in various situations.

Fly ash, building, and other factors all have an impact on atmospheric particles. Road dust was mostly caused by fuel-burning automotive exhaust and other sources. Indoor dust particle sources were definitely more complex, including the secondary reaction of fly ash, which could be because coal-burning fly ash persisted indoors for a long time (Zhou et al., 2021). Shilton et al. (2002) discovered that wind speed and direction had an effect on the penetration of outdoor particulates indoors, with higher wind speeds resulting in higher indoor dust levels. Saraga et al. (2017) discovered that particle infiltration and penetration from the outer environment, primarily through the ventilation system, influenced inside dust levels and to a lesser extent, through windows or cracks in the building.

Dust storms increase particulate matter (PM2.5 and PM10) concentrations. This problem has an impact on the dust stream's downstream ecosystems and population, as well as the spread of numerous diseases (Sorkheh et al., 2022; Krasnov et al., 2015) and dust storms, which are common in arid locations such as the Middle East and portions of Asia, are connected with elevated levels of Particulate Matter (PM) in the atmosphere. These outdoor PM levels are strongly related to inside PM concentrations (Kalua et al., 2020). According to the research of Shen and Kuo (2010), Asian dust storms are a well-known springtime meteorological phenomenon in East Asia, with their origins in arid parts of China and Mongolia. They discovered that during Asian dust storms, the concentrations of PM2.5 and PM10 in both indoor and outdoor air significantly increased by three times. Because ventilation systems in tall buildings pull in outside air, they are particularly vulnerable to external air contaminants, especially during dust storms. This effect is present in the interior environment at both fine and coarse particle levels. In ten homes in the Kuwait City metropolitan region, Yuan et al.'s (2020) study examined the amounts of 19 trace elements indoors and outdoors during dust and non-dust occurrences. The results showed that compared to non-dust events (40–60%), particle penetration efficiencies were lower during dust storms (less than 20–30%). Because the infiltration rates and settling velocities of the two size fractions differed, coarse particles had lower penetration effectiveness than fine particles.

An examination of snow dust in Lithuania revealed elevated concentrations of zinc and lead, indicating the potential impact of industrial operations and air dispersion, as seen by different regions. The results of analysis identified two primary clusters of elements: geogenic, which are influenced by soil composition and major businesses such as AB "Akmenės cementas" and AB "Lifosa," and technogenic, which are influenced by regional and transregional atmospheric movement, highlighting the contribution of pollution sources to the distribution of these elements (Kadunas et al. 1999).

Górny et al. (1995) study looked at aerosol concentration and size distribution inside and outside several housing locations in Upper Silesia, a heavily industrialized region in southern Poland, from December 1992 to April 1994. According to the analysis, tobacco smoke and dirty outside air are the main sources of indoor particulate matter in this location, which is why interior levels are typically lower than outdoor levels. On the other hand, the addition of new sources, like coal stove emissions or tobacco smoke, significantly altered the makeup of indoor aerosols. The results of the study highlight how important ambient air migration is in contaminating interior air. Bucur and Danet (2019) found that there were no reliable correlations between indoor and outdoor concentrations of PM2.5, furthermore natural ventilation and the generally low contribution of indoor sources are the major determinants of a strong correlation between indoor and outdoor air quality. In study of Zhou et al. (2021) observed that correlation between indoor dust and atmospheric particles > correlation between indoor dust and airborne dust > correlation between indoor dust and road dust. As a result, the most common interchange occurred between indoor dust and atmospheric particles, because the smaller the particle, the simpler it was to travel. The second, it was between indoor dust and airborne dust because the distance was near enough that they were at a similar height. The relationship between inside dust and road dust was weak. Because the road dust particles were rather big, movement would be limited. Outdoor carcinogenic risks were higher than indoor carcinogenic risks for adults, while indoor carcinogenic risks were higher than outdoor for children, indicating that children were more likely to be exposed to indoor dust by mouth-hand intake. Study in Jordan, the primary source of V, Ti, Mn, Pb concentrations were similar to outdoor concentration, and it was clearly vehicular emission, with the exception of Ni, Cr, and Cu, which were identified in higher concentrations in indoor dust samples than outside dust samples. The results show that heavy metal distribution is different in the commercial area versus residential area due to increased traffic density (Madanat et al., 2017). Othman et al.'s (2019) study focused on PM2.5 levels and dust composition in school classrooms in Kuala Lumpur City Centre. Indoor and outdoor 24-hour PM2.5 concentrations were determined to be around 11 micrograms per cubic meter. Student activities influenced indoor dust composition, whereas soil, industrial, and traffic sources shaped outdoor and indoor dust composition. Dingle et al. (2021) study examined the amounts of metals in household dust in Fort McMurray, Alberta. Entryway metal concentrations were found to be comparable to those of outside soils, especially those with natural components. Nevertheless, the amounts of copper, zinc, and lead in indoor living spaces were much greater, suggesting indoor sources. The concrete in basements had noticeably higher levels of lead. Surprisingly, interior dust metal concentrations were not significantly affected by the wildfires of 2016. In conclusion, there are differences in the composition of house dust in different parts of the house due to influences from both indoor and external sources. In Mohammed et al. (2023) data collection for a study was done in model rooms with various window arrangements in the northeastern part of Nigeria. Dust accumulation on surfaces was assessed through the application of mathematical correlations. The results showed that the amount of surface dust was significantly influenced by the layout and positioning of building apertures.

Study of Rodríguez-Chávez et al. (2021), in a high school close to mine tailings and a copper smelter in Hayden, Arizona, samples of particles were collected for both indoor and outdoor samples. The study emphasizes the significance of particle size in evaluating indoor air quality because, whereas bigger particles over one micrometer continue to be an important contributor of indoor pollutants, finer particles smaller than one micron are significantly decreased by filtration.

Each source of pollution has a unique isotopic signature. Therefore, by examining the ratio of carbon and sulfur isotopes in air particles, the origins and processes of elemental transformation may be identified. Indoor activities, particularly cooking, had an effect on particles in the atmosphere. Sulfur in air particles is mostly caused by coal burning. The sources of sulfur in indoor dust were even more complicated. (Zhou et al., 2021).

1.8. Assessing the Role of HVAC Systems in Minimizing Dust Pollution

HVAC filter dust analysis can be used for indoor environmental investigation. Pollutants in ducts mainly consist of sand, soil, fibers, hair, smoke, and carbon particles. Inorganic pollutants like crystalline structures are found in supply ducts, while organic fibers are mostly found in ventilation ducts (Jung et al., 2014; Nõu and Viljasoo, 2011). Dust, comprised of solid particles with diameters smaller than 75 micrometers, is a common contaminant found in homes, offices, and various indoor spaces (Nõu and Viljasoo, 2011). 70% of US homes have central air systems with built-in filters, making them widely accessible. HVAC filters capture particles over long periods of time and can provide a significant amount of information about indoor air quality (Noris et al., 2009; 2011). Therefore, understanding and controlling dust levels are vital for maintaining a healthy indoor environment.

HVAC systems are essential to maintaining indoor air quality. These devices have the ability to affect how dust and other pollutants are distributed inside buildings. Dust control devices in HVAC systems were assessed in a survey carried out in Seoul's subway, emphasizing panel-type filters, electrostatic precipitators, and demisters. Panel-type filters have collection efficiencies ranging from 20% to 90%, depending on particle size and filter life (Jang et al., 2009).

It's critical to assess pollutants and emissions from HVAC systems. Diverse concentrations of culturable fungi, bacteria, and heavy metals were discovered in studies on microbiological pollutants and metals collected on HVAC filters in residential and light-commercial buildings (Noris et al., 2009). An investigation into the effects of different heating systems on dust levels revealed that the choice of heating systems also affects interior air quality. The findings indicated that various heating techniques had an impact on the amount of dust present. Specifically, ground source heat pumps produced greater dust levels, possibly because the warm air moved upward and mixed-up dust from the floor (Nõu and Viljasoo, 2011).

In order to lower airborne particulate matter and bioaerosols in homes, HVAC filters are crucial. It was evaluated how well HVAC filters worked to lower the amounts of particles found indoors. Higher-efficiency filters have been shown to dramatically lower indoor proportions of outdoor particles, but the results vary depending on the age of the house, the climate zone, and the ventilation design (Azimi et al., 2016).

Controlling indoor particle concentrations requires an understanding of the rates of source emissions and material movement between surfaces and indoor air (Schneider, 2008). Keeping doors and windows closed indoors results in much less exposure to external dust (Alzona et al., 1979). Dust tends to collect more around doors and windows, mostly from outside sources. How important it is to adjust building opening designs in order to efficiently control interior dust levels (Mohammed et al., 2023). Also, during dust storm events, reducing interior exposure to crustal particles could be accomplished by enhancing home insulation (Yuan et al., 2020). Lastly, according to Nafchi et al. (2021), rooms with a central HVAC system responded faster to an internal source of pollution, than rooms with only fan coil units. The complicated and varied task of maintaining a healthy indoor environment includes knowledge of HVAC systems, heating techniques, and the dynamics of dust and pollutants.

1.9. Dust and Data: An Introductory Overview of PMF in Environmental Studies

A well-liked statistical method that may determine the primary contributing components without any prior knowledge of the sources is called positive matrix factorization (PMF) (Srivastava et al., 2021). Dr. Pentti Paatero developed the PMF receptor model in the 1990s to examine multivariate data. Without rejecting any values, it can manage non-representative data, such as missing data and outliers, and gets around the drawbacks of conventional factor analysis models. As a result, the original data set is not reduced (Comero et al., 2009). A few studies have used PMF in their investigations.

Utilizing PMF on data gathered during two field campaigns in 2016 and 2017, a study conducted in China explores the sources of PM2.5 in Beijing's urban and rural areas. Seven factors—traffic emissions, biomass burning, road dust, soil dust, coal combustion, oil combustion, and secondary inorganics—are identified by the PMF study as contributing to PM2.5 mass. Summertime PM2.5 mass at the urban site is dominated by soil dust and secondary inorganics, while wintertime PM2.5 mass at both locations is mostly influenced by biomass burning, coal combustion, and secondary inorganics (Srivastava et al., 2021). In a different investigation, PMF was used to pinpoint the sources of contamination in samples of street dust collected from Zhengzhou, China. The heavy metal industry, anthropogenic activities, coal combustion, and vehicle exhaust were shown to be the four main contributors to heavy metal pollution. This study's application of PMF demonstrates how well it can locate pollution sources in complex environmental matrices (Faisal et al., 2021). Apart from the previously mentioned research, PMF Modeling was employed to pinpoint five sources of pollution that are responsible for dust storms in nine different regions of Khuzestan, Iran. The origins of interior dust were found using these sources. The findings demonstrated that dust events in mid-summer to early autumn and mid-winter to early spring were primarily caused by internal sources, such as mud-salt zones and evaporitic deposits, while dust events in mid-October to early winter had external origins, primarily in Iraq and Saudi Arabia (Akbari et al., 2022). Another study investigated the sources of PM10 and their contribution to measured PM concentrations at three receptors in Western Macedonia, NW Greece, using the multivariate EPA PMF 5.0 receptor model on elemental data obtained from PM10 samples (Garas et al., 2020) and A study investigated ambient PM at Shinjung station in New Taipei City, Taiwan, using a dichotomous sampler and a MOUDI sampler. The samples were analyzed for metallic trace elements and ionic compounds. PMF was used to identify PM sources, and a total of five source types were identified, including soil dust, vehicle emissions, sea salt, industrial emissions, and secondary aerosols (Gugamsetty etl al., 2012). Finally, a Madrid investigation employed PMF to identify the sources of PM10 in a heavily trafficked road in Spain. Vehicle emissions, road dust, secondary aerosols, and soil were found to be significant contributors to PM10 (Karanasiou et al., 2011).

In their work, Srivastava et al. (2021) compared PMF results to other receptor models and data to determine their accuracy for source attribution. The results show that PMF is a valuable tool, but it should be used with caution because it failed to properly resolve some sources and exaggerated dust sources.

1.10. Unsolved Questions and Research Focus

While significant research has been conducted on dust pollution and heavy metal contamination in various environments, particularly in urban areas and industrial settings, several important questions remain unresolved:

- **Limited focus on schools:** Although many studies examine heavy metal contamination in outdoor environments and its impact on health, few have specifically focused on educational facilities, particularly in Lithuania and Eastern Europe. There is a lack of detailed research on the sources and composition of indoor dust in schools and how it correlates with outdoor environmental pollution.
- **Heavy metal accumulation in long-term dust deposits:** Previous studies often focus on short-term dust monitoring, but there is limited understanding of how heavy metals accumulate in dust over an extended period in school environments. The long-term accumulation of dust poses unique risks that have not been fully explored, particularly in educational settings.
- **Connection between soil pollutants and the presence of contaminants in indoor dust:** There is limited understanding of the direct relationship between heavy metal contamination in outdoor soil and the subsequent accumulation of these metals in indoor dust within school environments.
- **Impact on vulnerable populations:** While the health impacts of heavy metal contamination are well-documented for children, few studies have explored the broader impacts on adults, such as teachers and staff, who spend prolonged periods in polluted school environments.

In this dissertation, these unsolved questions will be addressed by:

- Investigating the concentration and sources of heavy metals in long-term accumulated indoor dust in schools.
- Establishing the link between outdoor soil and indoor dust contamination.
- Assessing the potential health risks for both children and adults in these environments.

1.11. Chapter Conclusion

The literature review confirms that heavy metal contamination in urban environments is a significant health concern, particularly in school settings where children are vulnerable. This chapter underlines the importance of further investigating the link between outdoor pollution and indoor dust contamination, especially in the context of schools in Vilnius.

- Soil acts as a key environmental interface, and its contamination with heavy metals (HMs) directly affects air quality and ecosystem functioning.
- Urban dust, especially street and indoor dust, is a significant carrier of pollutants like HMs and poses serious health risks, particularly in schools.
- Children are especially vulnerable to indoor dust contamination due to their behavior and physiology, increasing their exposure to toxic metals.
- Indoor dust pollution results from both internal sources (e.g. building materials, activities) and external ones (e.g. traffic emissions, outdoor soil and air).
- Although global research on HM contamination in indoor dust exists, there is a lack of studies in Lithuanian educational environments.

2. Materials and Methods

2.1. Description of the Area of Study

Vilnius, the capital and largest city of Lithuania, is located approximately 312 kilometers from the Baltic Sea and covers an area of 401 square kilometers with a population of about 550,000 (Misiune et al., 2021). These environmental and demographic characteristics are significant because they can influence local dust accumulation patterns and pollutant dispersion. Vilnius is located at an elevation of 112 meters above sea level and has coordinates of 54.687157 degrees north and 25.279652 degrees east. The city has a moderate climate, with annual rainfall averaging 80mm in July and 30mm in February. In Vilnius, the average annual humidity level is roughly 78 percent. Throughout the year, the wind direction in the city varies, but it is primarily from the south and west (Misiune et al., 2021).

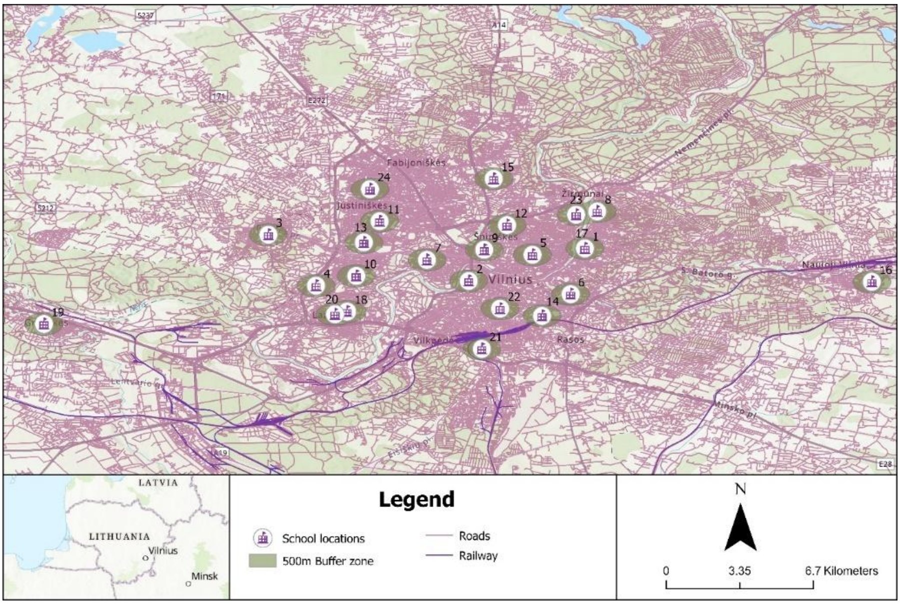


Figure 1. The study's focus is on the geographical location of the Vilnius area and sampled schools.

2.2. Determination of Collection Areas

The process of identifying collection areas involves analyzing various factors, including the years that schools were built and renovated, as well as the locations of those schools. It is crucial to examine historical data on school buildings and renovations as well as the geographic distribution of these educational establishments in order to guarantee precise and efficient collecting area designation. The methodical identification of regions where collection efforts should be focused is made easier with the help of this data-driven strategy. Historical industrial activity in Vilnius (1938-1940), including metalworking, printing, and chemical industries, likely contributed to long-term HM accumulation in urban dust (Dagys, 1968). Many schools built before 1980 (Žverynas Gymnasium (1964), Karoliniškės Gymnasium (1974)) may have accumulated pollutants from past industrial emissions, leaded gasoline, and older construction materials.

Table 4. Built and renovated years of studied schools in Vilnius.

Code	Schools	Built Year	Renovation Year
S1	Vilnius Antakalnio Gymnasium	1929-1930	2012
S2	Vilnius Simonas Daukantas Gymnasium	1930	2008
S3	Vilnius Pilaitė Gymnasium	1992	2012
S4	John Paul II Gymnasium	1992	2004
S5	Vilnius St. Christopher Gymnasium	1976	-
S6	Petro Vileišis Progymnasium	1969	2013
S7	Žverynas Gymnasium	1964	2010
S8	VGTU Engineering Lyceum	1963	2015
S9	Vilnius Vladislav Sirokomla Gymnasium	1950-1951	2017
S10	Karoliniskes Gymnasium	1974	2010
S11	Vilnius Viršuliškiai School	1974	2011
S12	Senvagė Gymnasium	1976	2011
S13	Vilnius Levo Karsavinas School	1978	-
S14	Vytes Nemunėlis Elementary School	2002	-
S15	Vilnius Baltupiai Progymnasium	1979	2009
S16	Vilnius Žaros Gymnasium	1980	-
S17	Vilnius Antakalnis Progymnasium	2012	-
S18	Vilnius Lazdynai School	1971	2009-2010

S19	Grigiškės Gymnasium	1952	2015
S20	Vilnius Žėručio Elementary School	2001	-
S21	Vilnius Naujininkai Progymnasium	1973	2012-2013
S22	Vilnius Simonas Daukantas Progymnasium	1993	2001
S23	Vilnius Pranas Mašiotas Elementary School	2001	-
S24	Medeinos Elementary School	1993	-

2.3. Sample Collection and Preparation

Multiple methods have been used in numerous articles to gather dust samples from diverse sites. Some investigations collected data using vacuum cleaners and their bags (Olujimi et al., 2015; Kurt-Karakus, 2012; Doyi et al., 2019; Naimabadi et al., 2021). Other authors collected dust samples from a variety of locations, including classroom floors, windowsills, playgrounds, balconies, doorsteps, stairs, entryways, fans, air conditioner filters, bookshelves, wall corners, desks, and chairs (Muhamad-Darus et al., 2017; Radhi et al., 2020; Shi and Wang, 2021; Latif et al., 2013).

In this study, dust samples were collected from locations that are typically challenging for cleaners to access and clean, including areas behind radiators, on top of bookshelves, in corners, higher windowsills, and gymnasium areas that are difficult to reach by human hands for cleaning purposes.

In 2022 and 2023, dust samples were gathered from 24 educational institutions selected according to specific criteria, including their geographical location within Vilnius and proximity to recognized pollution sources such as highways and frequent transit routes. We deliberately selected schools constructed from 1930 to 2012 in order to encompass a wide variety of building ages and histories, thereby capturing potential variations in dust contamination related to building materials and environmental exposure over time. Most of these establishments have been renovated at various times, potentially affecting the composition of indoor dust. The selection criteria were formulated to encompass a thorough portrayal of dust accumulation throughout Vilnius, considering different architectural periods and maintenance approaches. The samples were collected with brush techniques from often neglected areas by cleaners, such as the area behind radiators, the upper part of bookcases, corners, windowsills, and hard-to-reach sections of gymnasiums. Our focus was on the gradual accumulation of dust over a long period of time in samples places. Each sample consisted of approximately 10 grams of dust. At each site, one composite sample was collected from designated hard-to-clean locations (e.g., behind radiators, upper shelves). While this approach ensured consistency, future studies may consider collecting replicate samples at each site to evaluate within-site variability. The dust samples stored in sterilized, airtight containers at room temperature in a contamination-free lab environment. Afterward, each sample was separated into capsules (max. 6 capsules) for examination in the lab.

Figure 2. Where samples collected

Preparation and Analysis of Dust Samples. This work utilizes pre-existing datasets from previous research (Kumpienė et al. 2011; Kadūnas et al. 1999; DGE Baltic Soil, 2021;2023) instead of collecting new field samples for the evaluation of topsoil samples. The selection of this approach was based on the thorough and wide nature of the available data, which sufficiently encompasses the geographic and chronological aspects relevant to our investigation. By leveraging these existing datasets that enhanced our research by incorporating a broader historical context and cross validating our dust sample results with previous soil data (Kumpienė et al. 2011; Kadūnas et al. 1999; DGE Baltic Soil, 2021;2023).

The dust samples were examined using Niton XL2 XRF Analyzer spectroscopy by Thermo Fisher Scientific (USA) (Aguilera et al. 2021; Zacco et al. 2009), Prior to analysis, the samples were prepared by breaking them into smaller pieces and mounting them on a sample holder. To ensure the integrity of our analysis, it is crucial that the capsules should be clean and free from any contaminants that could interfere with the results. Each time the device is activated, calibration and system checks are conducted using standard samples with established concentrations. The apparatus was exclusively operated within a laboratory stand, setting the analysis duration to 600 seconds to maximize accuracy through exposure to three distinct characteristic energy lines. The accuracy of chemical element analysis varies, ranging from 10% for elements such as Cr, Cu, Zn, Zr, Sr, Rb, Mn, Fe, Ti, to 20% for As, Pb, Cd. Additionally, the device underwent inter-calibration with the SPECTRO XEPOS (SPECTRO Analytical Instruments GmbH, Kleve, Germany) energy dispersive X-ray fluorescence (ED-XRF) spectrometer at the Lithuanian Geological Survey, ensuring high measurement accuracy. XRF's non-destructive nature (no acid treatment) allows for the reuse of samples in multiple devices. XRF is a commonly used method for analyzing the elemental composition of samples (Hou et al. 2004). X-ray fluorescence (XRF) instrument used in mining, manufacturing, and environmental monitoring for quick, non-destructive elemental analysis. These samples were carefully prepared before the analysis by breaking them up into smaller pieces and placing them on a sample holder. The samples must be pure, and free of any contaminants that could interfere with the analysis procedure (Zacco et al. 2009).

XRF spectrometry offers advantages such as element-specific detection and eliminates the need for pre-treatment of the samples (Zacco et al. 2009) and this technology, when used with adequate sample preparation and understanding of its limitations, can provide precise and reliable results as a valuable complement to laboratory analyses in the field of geochemical and environmental analysis. Such as, sample preparation differences, moisture content, matrix effects, and analytical interferences (Mantler & Schreiner, 2001; Gianoncelli & Kourousias, 2007; Laperche & Lemière, 2020; Kadachi et al., 2012). Mercury concentrations analyzed with the Niton XL2 Analyzer and SPECTRO XEPOS were undetectable, hence, analyses were limited to elements consistently detected by both instruments.

Figure 3. Samples and used capsules for the analysis.

2.4. Pollution Assessment

2.4.1. Geo-accumulation index

Müller first proposed (Müller, 1969) the Geoaccumulation Index (I_{geo}) to assess metal concentrations in sediments' 2-micron fraction. International standard shale values are used as a baseline for this indicator (Barbieri, 2016).

Equation 1

$$I_{geo} = log_{2} \frac{Cn}{Bn\ x\ 1.5}$$

The concentration of a specific element in dust is denoted as Cn. The constant value of 1.5 is used to account for natural variations in element content and to detect even minimal anthropogenic influences. The geochemical background value is represented as Bn. Müller classified the Geoaccumulation Index into seven classes, ranging from class 0 to class 6. The highest class, class 6, signifies an enrichment factor of at least 100 times higher than the background values (Barbieri, 2016).

2.4.2. Contamination Factor

The contamination factor (CF) is a method used to assess the level of contamination of indoor dust by a particular metal. It is calculated using the following equation:

Equation 2

$$CF = \frac{C_{Sample}}{C_{Background}}$$

The CF provides a quantitative measure of the extent to which the concentration of a specific metal in indoor dust deviates from the background concentration. The background value of trace elements in the Earth's crust is denoted as CBackground, while the concentration of the elements found in the samples is represented by CSample. Background values for indoor and outdoor dust have not been established in this investigation. Alternatively, Vilnius and global background values for soils have been used (Aguilera et al. 2022). The contamination factor (CF) can be classified as follows: CF < 1 indicates low contamination, CF between 1 and 3 represents moderate contamination, CF between 3 and 6 indicates considerable contamination, and CF greater than 6 suggests very high contamination (Gope et al. 2017). According to studies (Gope et al. 2017; Aguilera et al. 2022), these categories are founded on accepted environmental research standards and offer a methodical way to assess the degree of contamination in urban dust.

2.4.3. Modified Degree of Contamination

Modified degree of contamination (mCd) The mCd is a global contamination index that evaluates the degree of contamination of sediments, integrating all the toxic metals evaluated in the ecosystem (Custodio et al., 2022). mCd quantifies the absolute degree of contamination in a sample by dividing the sum of the contamination factors (Cf) of selected metals by the total number of measured metals (n). Used this approach and implemented it for our dust samples. This approach provides an average total value for various contaminants. The mCd is further classified into seven different classes to categorize the level of contamination (Saha et al., 2022).

The Modified Degree of Contamination (mCd) is categorized into different levels of contamination: uncontaminated to very low (mCd ≤ 1.5), low (1.5 < mCd ≤ 2), moderate (2 < mCd ≤ 4), high (4 < mCd ≤ 8), very high

(8 < mCd ≤ 16), extremely high (16 < mCd ≤ 32), and ultra-high (mCd > 32) (Custodio et al., 2022).

Equation 3

$$mCd = \frac{\sum_{i=1}^{n} Cf}{n}$$

2.4.4. Pollution Load Index

The Pollution Load Index (PLI) is a measure that assesses the overall pollution load resulting from the presence of hazardous metals at a specific site. The PLI is calculated by considering the Contamination Factor (CF) for each element present. CF represents the degree of contamination for each individual element.

By calculating the PLI for a particular location, information about the cumulative pollution load caused by all the hazardous metals can be obtained. When the PLI is less than 1, it indicates that there is no pollution present at the site. A PLI value of 1 suggests that only baseline levels of pollutants are present, implying minimal pollution. However, if the PLI exceeds 1, it indicates that the quality of the site has deteriorated due to pollution. This methodology for assessing pollution load using the PLI, based on CF values, was discussed in a study conducted by Gope et al. in 2017.

Equation 4

$$PLI\ for\ a\ site = (CF_1\ x\ CF_2\ x \dots x\ CF_n)^{\frac{1}{n}}$$

Equation 5

$$PLI\ for\ a\ zone = (PLI_{site\ 1}\ x\ PLI_{site\ 2}\ x \dots x\ PLI_{site\ n})^{\frac{1}{n}}$$

2.4.5. Enrichment Factor

The measurement of anthropogenic pollutant deposition on surface soil is done using the Enrichment Factor (EF). According to Bern et al. (2019), it contrasts the concentration of a metal of interest with that of a stable reference element. For EF calculations Fe used. Numerous writers who have studied marine and estuary sediments have employed Fe (Barbieri, 2016; Bern et al., 2019). Natural weathering processes are suggested by EF values between 1 and 3, however considerable contributions from non-crustal sources such as pollution are indicated by values over 3. According to Barbieri et al. (2016), EF serves as an indicator for dust pollution, with values below 1 signifying no enrichment and values above 50 indicating extremely severe enrichment. EF could change according to background values and small change can increase EF.

Equation 6

$$EF = \frac{[Cx/Cref]_{Sample}}{[Cx/Cref]_{Background}}$$

2.5. Health Risk Assessment Model

2.5.1. Non-Carcinogenic and Carcinogenic Exposure Evaluation

The non-carcinogenic exposure to harmful metals in school dust for both adults and children was evaluated in this study using models derived from the US Environmental Protection Agency. The primary target receptors, including adults working at schools and children attending them, are exposed through four main pathways: direct ingestion of dust (D_ing, Equation 7), inhalation of dust particles through the mouth and nose (D_inh, Equation 8), dermal absorption through skin contact (D_dermal, Equation 9), and inhalation of vapors, particularly from mercury (D_vap, Equation 10) (Li et al., 2014).

To assess the health risks, exposure parameters were used to calculate the potential risks associated with different exposure pathways in Table 5. These parameters include the concentration of the elements in dust (C), ingestion rate (IngR), inhalation rate (InhR), body weight (BW), and other factors such as the skin surface area exposed to dust and the particle emission factor (PEF). The exposure values differ for children and adults, reflecting variations in body weight, surface area, and inhalation rates (Table 5), which are essential for determining the risks posed by dust exposure through ingestion, inhalation, and dermal absorption.

Equation 7

$$D_{ing} = C \ x \ \frac{IngR \ x \ EF \ x \ ED}{BW \ x \ AT} \ x \ CF$$

Equation 8

$$D_{inh} = C\ x\ \frac{IngR\ x\ EF\ x\ ED}{PEF\ x\ BW\ x\ AT}$$

Equation 9

$$D_{dermal} = C\ x\ \frac{SL\ x\ SA\ x\ ABS\ x\ EF\ x\ ED}{BW\ x\ AT}\ x\ CF$$

Equation 10

$$D_{vap} = C\ x\ \frac{InhR\ x\ EF\ x\ ED}{VF\ x\ BW\ x\ AT}$$

Table 5. Exposure parameters used for health risk assessment through different exposure pathways for dust (Li et al., 2014; Behrooz et al., 2022; Vosoughi et al., 2020)

	Parameters and Units	**Child**	**Adult**
C	Concentration of the element (mg/kg)		
IngR	the ingestion rate (mg/day)	200	100
SA	the surface area of the skin exposed to heavy metals (cm^2)	2800	5700
AF	the skin adherence factor (mg/cm^2);	0.2	0.7
ABS	dermal absorption factor (unitless)	0.001	0.001
InhR	the inhalation rate (m^3/day);	7.6	20
PEF	the particle emission factor (m^3/kg)	1.4E+09	1.4E+09
EF	the exposure frequency (days/year);	285	285
ED	the exposure duration (year);	6	30
BW	the body weight (kg)	15	70
AT	the average time (days);		
	For carcinogens	25550	25550
	For non-carcinogens	2190	10950
CF	the conversion factor	1E-06	1E-06
VF	volatilization factor m^3/kg	32675.6	32675.6

The hazard quotient (HQ) and hazard index were used to evaluate the non-carcinogenic effects of metals (HI). A heavy metal's HQ is calculated by dividing its ADD by its reference dose (RfD) for the same exposure pathway (s). The reference dose (RfD see in Table 6) (mg/kg day) is the highest daily dose of a metal from a particular exposure pathway, for both adults and children, that is thought not to significantly increase the risk of adverse effects on sensitive people over the course of their lifetime. It is assumed that there won't be any negative health impacts if the ADD is less than the RfD, HQ ≤ 1, but if the ADD surpasses the RfD, HQ ≥ 1, it is expected that there will be negative health effects. The hazard index (HI) is the total risk of a single non-carcinogenic factor through all three paths of exposure. The value of HI ≤ 1 indicates that there is no risk of non-carcinogenic effects, whereas HI ≥ 1 suggested that there is a chance of negative health impacts, and that chance grows as HI values rise (Qing et al., 2015).

Equation 11

$$HI = \sum HQ_i = \sum \frac{ADD_i}{RfD_i}$$

Table 6 Used RfD values.

Element	**RfD Ing**	**RfD Der**	**RfD Inh**	**CSF Ing**	**CSF Der**	**CSF Inh**
As	0.0003	0.000123	0.000301	1.5	1.5	15
Cu	0.04	0.0402	0.012	NA	NA	NA
Zn	0.3	0.3	0.35	NA	NA	NA
Sr	0.6	0.12	0.6	NA	NA	NA
Pb	0.0035	0.00053	0.0035	0.0085	0.0085	0.042
Cr	1.5	0.006	0.00003	0.5	0.5	41
V	0.007	0.007	0.00007	NA	NA	NA
Fe	0.7	0.7	0.8	NA	NA	NA

The hazards associated with carcinogens are quantified as the additional likelihood of an individual acquiring cancer over a lifetime due to exposure to the suspected carcinogen. The formula for determining the increased lifetime cancer risk is (Kamunda et al. 2016):

Equation 12

$$Risk = \sum_{k=1}^{n} ADDi \; x \; CSFi$$

Equation 13

$$RiskTotal = Risk(ing) + Risk(inh) + Risk(Der)$$

Risk is a unitless probability of an individual developing cancer over a lifetime. CSF is cancer slope factor and $(mg/kg/day)^{-1}$. Where Risk (ing), Risk (inh), and Risk (dermal) are risks contributions through ingestion, inhalation and dermal pathways (Kamunda et al. 2016).

The EPA typically seeks to limit lifetime cancer risks from chemical exposures to between 10^{-6} and 10^{-4}. Consequently, there is a significant possibility that extremely high exposures may exceed the EPA's tolerable risk threshold. This series aims to protect both the general populace and at-risk groups (Baynes, 2011).

2.6. Geospatial Mapping, Statistical Analysis and Data Computation

Statistical analysis and data calculations. All statistical analyses and data calculations were conducted using Python 3.12.1 (Some Codes are provided in supplementary content to give better idea for how the calculations are made.), utilizing various scientific libraries to ensure accurate and robust computations.

Data Collection and Preparation. The study focused on comparing heavy metal concentrations—specifically As, Cu, Zn, Pb, and Cr—between soil and dust samples collected from various schools across different years. Data from the years 1999, 2011, 2017, 2018, 2019, 2020, 2021, and 2023 were analyzed. For each year, concentrations of the five metals were extracted for both soil and dust samples. The datasets were carefully cleaned and prepared to ensure accurate and comparable results.

Data Cleaning. To prepare the datasets for statistical analysis, a thorough data cleaning process was implemented:

1. Handling Non-Detects (Values Below Detection Limits):

- Values reported as below detection limits (e.g., "<0.50") were replaced with half the detection limit (e.g., 0.25). This approach is standard in environmental studies for censored data, providing a reasonable estimate without significantly biasing the results (Helsel, 2005).

2. Conversion to Numeric Data:

- All data entries were converted to numeric format. Non-numeric entries and any remaining non-detect symbols were appropriately handled to ensure the datasets contained only numerical values suitable for statistical analysis.

3. Addressing Missing Values:

- Missing values were excluded from the analysis for specific statistical tests. If too few observations were available for a particular metal in a given year (e.g., less than three data points), the results were marked as "not enough data" to ensure statistical validity.

Statistical Analysis. A combination of parametric and non-parametric statistical tests was used to evaluate whether the metal concentrations in soil and dust samples differed significantly for each metal and year, and to assess changes in soil metal concentrations across different years.

Normality Testing. Before performing comparative analyses, the Shapiro-Wilk test was conducted on the log-transformed data to assess normality for each metal in both soil and dust samples (for soil vs. dust comparisons) and for soil samples across different years.

Interpretation of Shapiro-Wilk Test Results:

- p-value > 0.05: The data is assumed to follow a normal distribution.
- p-value ≤ 0.05: The data is considered non-normally distributed.

Comparison of Soil and Dust Samples Within Each Year. Used two-sample tests (such as the Independent t-test (Welch's) or the Mann-Whitney U test) to compare soil and dust concentrations because it were analyzing two groups (soil and dust), focusing on whether their means or distributions were significantly different. For each metal and year, the following steps were performed:

1. **Selection of Appropriate Statistical Tests.** Selection of Appropriate Statistical Test:
   - For each metal and sampling year, first assessed normality using the Shapiro-Wilk test. If both soil and dust datasets for a metal were normally distributed ($p > 0.05$), it has

employed an Independent Two-Sample t-test (Welch's version, which does not assume equal variances) to compare mean concentrations. If either dataset deviated from normality ($p \leq 0.05$), the Mann-Whitney U test was used instead to compare the distribution of concentrations.

2. Determination of Statistical Significance:
   - The threshold for statistical significance was set at $p < 0.05$:
     - p-value < 0.05: Indicates a statistically significant difference between the concentrations of metals in soil and dust for that metal and year.
     - p-value ≥ 0.05: Suggests no significant difference, indicating that metal concentrations in soil and dust are comparable.
3. Effect Size Calculation:
   - For t-tests, Cohen's d was calculated to estimate the effect size, providing insight into the magnitude of differences between soil and dust concentrations.
     - Interpretation of Cohen's d magnitude:
       - Small Effect Size: $d=0.2$
       - Medium Effect Size: $d=0.5$
       - Large Effect Size: $d=0.8$ or greater
     - Sign (Positive or Negative): Indicates the direction of the difference between the two groups.

Software and Libraries Used. All statistical analyses were conducted using Python 3.12.1, utilizing the following libraries:

- Pandas: For data manipulation and cleaning.
- NumPy: For numerical computations and data transformations.
- SciPy: For conducting statistical tests such as Shapiro-Wilk, t-tests, Mann-Whitney U tests, ANOVA, and Kruskal-Wallis tests.
- Statsmodels: For performing Tukey's HSD post-hoc analysis.
- Scikit-Posthocs: For conducting Dunn's test with Holm adjustment.
- Matplotlib/Seaborn (if applicable): For data visualization and plotting results.

Assumptions and Considerations.

- ✓ Sample Size Requirements: Statistical tests were performed only when there were sufficient data points (e.g., at least three observations) in each group to ensure the validity of the test results.
- ✓ Assumption Checks: Assumptions of normality and homogeneity of variances were assessed before selecting the appropriate statistical tests..
- ✓ Data Integrity and Limitations: All data cleaning and transformation procedures, including handling non-detects by substituting half the detection limit, converting entries to numeric values, and excluding insufficient data points—were systematically documented and uniformly applied across all datasets to ensure data integrity and reproducibility.

This methodology provided a systematic and statistically sound approach to evaluate whether there were significant differences in metal concentrations between soil and dust samples within each year and to assess changes in soil metal concentrations across different years.

The geographic mapping software, ArcGIS Pro 3.0, was used to create maps, as it serves as a powerful analytical tool for representing geographical data (Partee and Lindsay, 2017), was used to study and represent the distribution of PM and HMs throughout Vilnius. Data was computed, and statistical analysis was conducted using the Python programming language. The distribution of metals was analyzed and visualized through geospatial mapping software, with metal concentration measurements forming the basis for the maps. Eleven selected metals were analyzed to establish their spatial distribution. While IDW interpolation was used for datasets with a larger number of data points, such as soil and PM data, heat mapping to point out schools, was employed for smaller datasets, such as school dust, to ensure accuracy. This approach allows for a clearer depiction of contamination patterns and high-risk areas without introducing bias from interpolation. Future studies may enhance the understanding of PM and metal distribution in schools by incorporating more data points and additional co-variates, such as traffic density, meteorological conditions, or proximity to industrial areas. This could lead to more comprehensive modeling of pollution patterns and further improve the assessment of environmental risks in educational settings.

Principal Component Analysis (PCA) is a powerful technique used in multivariate research, particularly when dealing with datasets containing interconnected quantitative variables (Abdi, 2010). Its primary purposes are to reduce the dimensionality of a dataset and to identify latent factors. By extracting orthogonal principal components from the original variables, PCA simplifies the dataset while preserving key information (Chen et al., 2016). These components help reveal the underlying structure of the data and provide valuable insights. PCA also enables the visualization of relationships between the original variables, making it easier to understand complex multidimensional systems (Daultrey, 1976). In this study, PCA was employed to identify the primary variables driving metal concentrations in school dust and to reduce the complexity of the dataset before further analysis.

K-means Clustering: K-Means Clustering is an unsupervised learning algorithm that partitions a dataset into K distinct, non-overlapping clusters based on feature similarity. The goal is to group similar data points together while maximizing the differences between clusters.

- Cluster: A group of data points aggregated together because of shared characteristics.
- Centroid: The center point of a cluster. It is the mean position of all the points in the cluster.
- K: The number of clusters the algorithm aims to identify in the data.

Arbitrary Assignment: In K-means clustering, cluster labels (e.g., Cluster 1, Cluster 2, Cluster 3) are arbitrarily assigned. This means that Cluster 1 in one analysis does not inherently correspond to Cluster 1 in another analysis unless explicitly matched based on cluster characteristics.

Independence Across Analyses: When performing K-means clustering separately for each year (2017 to 2023), the algorithm assigns cluster labels based solely on the data for that specific year. Consequently, Cluster 1 in 2017 may represent a different grouping of contaminants compared to Cluster 1 in 2018.

Hierarchical Clustering Analysis (HCA). In contrast to PCA, which focuses on reducing dimensionality, Hierarchical Clustering Analysis (HCA) is used for grouping similar data points. For this study, HCA was applied to the dataset after standardizing the values using z-scores, followed by calculating Euclidean distances between the heavy metal concentrations. Ward's linkage method was chosen as the clustering criterion (Zheng et al., 2012). HCA serves to group the data points based on their similarity, offering a different perspective from PCA by clustering variables or samples with shared properties. Dendrograms were used to visualize the clustering process, helping to identify groups with similar heavy metal concentrations. A positive coefficient indicates a positive linear relationship, while a negative coefficient indicates a negative linear relationship (Radhi et al., 2020)

While PCA allowed us to identify key drivers of metal concentrations, HCA further grouped these concentrations into meaningful clusters, allowing us to understand the relationships between them in greater detail. The combination of PCA and HCA provides complementary insights: PCA focuses on reducing dimensionality and identifying the most important variables, while HCA groups similar observations based on their characteristics, offering a more nuanced understanding of the relationships between data points. Together, these techniques ensure a comprehensive understanding of the dataset before proceeding with further spatial analysis or mapping. This method already used by different researchers (Szczepanik et al., 2021; Jiang et al., 2015; Taşan et al., 2022).

To determine the optimal number of clusters for our dataset, a two-step clustering approach was employed. Hierarchical Clustering Analysis (HCA) was initially used to group the samples based on their similarities, with the results visualized as dendrograms. Following this, K-means clustering was applied to refine the number of clusters identified by HCA. The optimal number of clusters was determined using both the elbow method and silhouette analysis.

The elbow method relies on Within-Cluster Sum of Squares (WCSS), which measures the total variance within each cluster. The elbow method helps visualize the point where adding more clusters no longer significantly reduces the variance, indicating the optimal number of clusters. In contrast, the silhouette method evaluates the cohesion and separation of clusters by measuring how similar each data point is to its own cluster compared to others. The silhouette score ranges from -1 to 1, with higher values indicating more cohesive and well-separated clusters (Belyadi and Haghighat, 2021). This combination of methods ensures that the selected number of clusters is both statistically sound and produces meaningful groupings of the data.

By using this two-step clustering approach, the data were grouped accurately and reliably, providing a solid foundation for subsequent mapping and risk assessment.

Correlation Analysis. To assess the strength and direction of relationships between individual metal concentrations, a correlation analysis was conducted. Given that the data may not adhere strictly to normal distribution assumptions, Spearman's rank correlation coefficient were employed, depending on the normality of each variable.

Selection of Correlation Method:

- Spearman's Rank Correlation Coefficient (ρ): Used when at least one variable in a pair was not normally distributed. Spearman's correlation assesses the strength and direction of the monotonic relationship between two variables, based on the ranked values rather than the raw data.
- Distance Correlation: It is a robust, non-parametric statistic that quantifies the dependence between two random variables or vectors. Unlike traditional measures such as Pearson or Spearman correlation, which are limited to detecting linear or monotonic relationships, distance correlation is capable of capturing both linear and non-linear dependencies. The distance correlation metric produces values between 0 and 1, where a value of 0 indicates complete statistical independence and a value closer to 1 implies a stronger association. Importantly, distance correlation does not indicate the direction of the relationship (i.e., positive or negative), but rather the strength of the association irrespective of its form.

Finding nearest distances with shapefiles and PM sensors. Methodological framework employed to analyze the spatial relationships between air quality measurements (PM2.5 levels) and various infrastructural features (schools, roads, traffic lights, railways, and bus stops) within the study area. The primary datasets utilized in this study comprise PM2.5 concentration measurements and the geographical locations of educational institutions (schools). The data was imported into the analysis environment using the pandas library, which facilitates efficient data manipulation and preprocessing. Supplementary geospatial datasets were acquired in the form of shapefiles, encompassing various infrastructural features: roads, traffic lights, railways and bus stops. These shapefiles were loaded using the `GeoPandas` library, which extends pandas' capabilities to handle geospatial data. Data integrity is paramount; hence, the clean_data function was employed to purge records with missing geographic coordinates. The cleaned datasets were transformed into GeoDataFrames, Both GeoDataFrames were assigned the CRS `EPSG:4326`, which corresponds to the WGS 84 geographic coordinate system. Ensuring the spatial data is suitable for analysis involves several steps:

- 2D Geometry Enforcement: The ensure_2d function strips any Z-axis (altitude) information from geometries, ensuring all spatial features are two-dimensional. This simplification is crucial for planar distance calculations.
- Geometry Validation: The validate_geometries function assesses the validity of geometries within each GeoDataFrame, removing any invalid geometrical entities that could skew spatial analyses.
- CRS Transformation: To facilitate accurate distance measurements, all GeoDataFrames were reprojected to the projected CRS EPSG:3857 (Web Mercator) using the reproject_gdf function. This projection is suitable for distance calculations over regional scales.

Proximity Analysis was conducted to determine the spatial relationships between schools and various infrastructural features, including PM2.5 sensors, roads, traffic lights, railways, and bus stops. For point-based features (PM2.5 sensors, traffic lights, railways, and bus stops), the `scipy.spatial.cKDTree` was utilized to perform efficient nearest neighbor searches:

- KDTree Construction: The build_kdtree function constructs a KDTree from the coordinates of the target features (e.g., PM2.5 sensors).
- Nearest Neighbor Identification: The find_nearest function queries the KDTree to identify the nearest feature for each school, returning both the distance and the index of the nearest feature.
- Distance Calculation: The resultant distances represent the proximity of each school to the nearest feature, which are subsequently appended to the schools GeoDataFrame.

Roads, typically represented as LineStrings or other non-point geometries, required a different approach for distance calculation. The proximity metrics for all features are integrated into the schools GeoDataFrame.

2.7. Positive Matrix Factorization

A multivariate factor analysis tool called positive matrix factorization (PMF) was utilized to identify the sources of heavy metal pollution. This efficient analysis method involved breaking down the matrices of sample concentration figures into factor profile matrices and factor influence matrices. The sources of pollution were determined by analyzing the resulting profiles (Faisal et al., 2021).

The PMF model analyzed using EPA PMF 5.0 and was applied in this study to identify the origins and spatial distribution of metals in the soil. Initially, the model was set up with 3, 4, and 5 factors, and a random seed number was selected from 20 iterations to begin the analysis. The determination of the optimal number of factors was based on finding the lowest and most reliable Q true value. The search for an appropriate residual matrix E involved finding the minimum Q value, which helped establish the suitable number of factors. The PMF analysis resulted in the optimal output, providing the lowest Q value (Saha et al., 2022; Souto-Oliveira et al., 2021; Chavent et al., 2008).

2.7.1. Positive Matrix Factorization Model Analysis

The USEPA used the PMF receptor model 5.0 to identify and distribute pollution sources (Norris et al. 2014). This statistical method has advantages over other methods such as PCA (Saha et al., 2022). The samples were examined and classified into factor outline and factor influence matrices. The purpose of this model is to divide the original matrix X (dimensions i x j) into two computational factor matrices, F (dimensions k x j) and G (dimensions i x k), as well as a residual matrix E (dimensions i x j). The following equation was used to determine the PMF (Saha et al., 2022):

Equation 14

$$Xij = \sum_{k=1}^{p} (Gik \ x \ Fkj) + Eij$$

The PMF model use matrix X (I x j) to represent the jth metal concentration recorded at the ith sample point. The model then factors this matrix into two computational matrices, Gik and Fkj, which represent, respectively, the contribution of the kth source to the ith sample and the concentration amount of the jth metal from the kth source. A residual error matrix, Eij, is also supplied. To normalize each element's prediction error in matrix X, the model takes uncertainty in data quality into account. Individual sources are indicated by the parameter p. The PMF model provides non-negative concentration matrices G and F that are tuned to minimize the objective function Q (Chai et al., 2021; Faisal et al., 2021; Srivastava et al., 2021). The value of Q is determined using the following calculation:

Equation 15

$$Q = \sum_{i=1}^{n}\sum_{j=1}^{m}\left(\frac{Xij - \sum_{k=1}^{p} Gik\ x\ Fkj}{Uij}\right) = \sum_{i=1}^{n}\sum_{j=1}^{m}\left(\frac{Eij}{Uij}\right)^{2}$$

The uncertainty values Uij of the metal concentrations in the samples (Xij) and the residual error matrix Eij are used to calculate the objective function Q. The m and n parameters denote the number of metals and samples, respectively. To utilize the PMF model, it is necessary to have data on both metal concentrations and their related uncertainties (Saha et al., 2022). The uncertainty associated with a specific metal's Method Detection Limit (MDL) can be computed using its concentration and the given error fraction (Norris et al. 2014). If the metal concentration exceeds the MDL value, the uncertainty can be calculated using the formula below:

Equation 16

$$Uij = \sqrt{(\sigma\ x\ Xij)^{2} + (MDL)^{2}}$$

If a metal's concentration exceeds the Method Detection Limit (MDL), the uncertainty is determined using the first equation, where represents the relative standard deviation and Xij represents the metal concentration (Norris et al. 2014; Faisal et al., 2021). If the metal concentration is less than the MDL, the uncertainty is determined using the following equation:

Equation 17

$$Uij = \frac{5}{6}\ x\ MDL$$

This is how the method detection limit (MDL) value is computed (USEPA,2016):

Equation 18

$$MDL = t_{(n=1,1-\infty=0.99)}\ S_{s}$$

The method detection limit (MDL) is calculated by combining the student’s t-value with appropriate values for n and 1-∞ (in this case, 0.99 for a single-tailed 99th percentile t statistic) and the sample standard deviation (Ss) of the replicate spiked sample analyses (USEPA, 2016). The Bootstrap approach was used to assess the uncertainties in the PMF model, which takes into account random errors induced by sample variability or disproportional observations on the PMF outcome. The range between the base factor concentration and the upper uncertainty limits for the Bootstrap was utilized to assess the uncertainties (Saha et al., 2022).

Lastly, key points as below:

- Blue Bars: These represent the concentration (Conc. of Species) of each metal species in the factor. The height of the blue bar indicates the concentration of the metal in that factor.
- Red Dots/Squares: These represent the percentage (% of Species) contribution of each metal species to that specific factor. The higher the red square, the greater the percentage contribution of that particular metal to the factor.

3. Results and Discussion

3.1. Assessment of Heavy Metal Contamination in Dust in Vilnius Schools

The analysis of dust samples from 24 schools in Vilnius revealed diverse concentrations of heavy metals, including As, Cu, Zn, and Pb. These metals are known to originate from various sources such as traffic, construction activities, and atmospheric deposition. Our findings showed significant variation in metal concentrations between the schools, suggesting multiple local pollution sources.

Cu concentrations varied widely, from 51.28 mg/kg to 395.37 mg/kg, suggesting that localized factors such as proximity to high-traffic zones significantly influence its distribution. Conversely, As levels were relatively uniform across all schools, implying a pervasive background source. The significant variability in metal concentrations, as shown in Table 7 and Figure 4, aligns with global trends observed in urban environments where traffic-related emissions and other anthropogenic activities contribute to localized contamination.

These results suggest that while some elements, such as As, may come from broader, more diffuse sources, others, like Cu, are influenced by more localized factors, such as traffic density or nearby construction. This diversity in contamination sources underscores the complexity of urban pollution patterns and the need to consider multiple variables when assessing environmental risks in schools.

The comparative visualization of elements such as As, Cu, Pb, Cr, and Zn in the dust samples revealed distinct distribution patterns, providing insights into the potential sources and types of dust accumulation in the educational environments studied. These variations suggest that certain pollutants may be influenced by proximity to traffic, construction, or other localized activities within the urban setting. The notable differences in element concentrations between dust and soil samples underscore the complexities of environmental sampling, highlighting the importance of careful data interpretation.

A more detailed analysis of these variations is shown in Table 8, which compares metal concentrations in Vilnius schools with data from other global indoor and outdoor environments. For example, As concentrations in our samples ranged from 4.55 mg/kg to 69.96 mg/kg. In comparison, South Africa's School B recorded 0.78 mg/kg, and Sydney measured 17.6 mg/kg. These differences underscore the influence of both geographical and environmental factors on dust contamination, providing important context for interpreting the levels observed in Vilnius schools.

The variability in metal concentrations, particularly when compared to global benchmarks, sets the stage for further comparative and correlational analysis. Such investigations could help clarify the connections between dust contamination in urban school environments and specific sources of pollution, whether local or global.

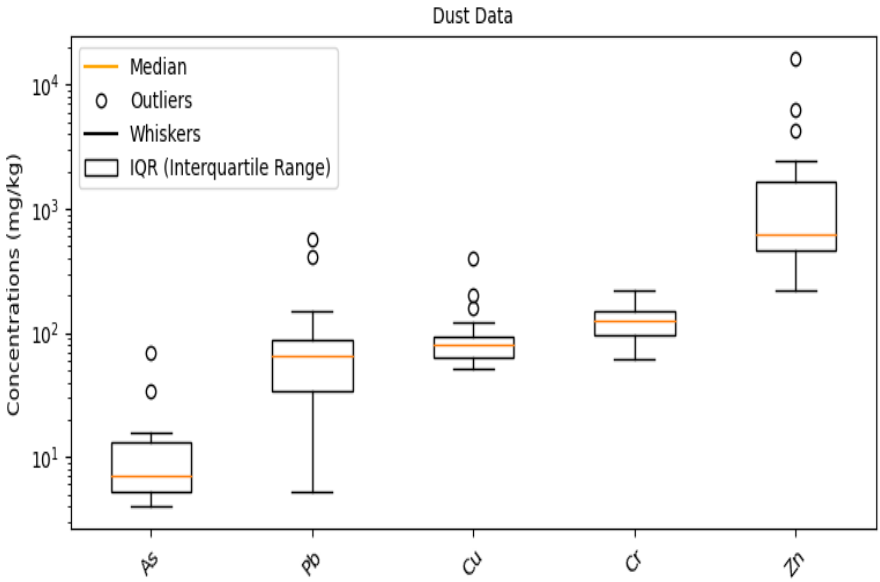


Figure 4. Distribution of As, Cu, Zn, Pb and Cr

3.1.1. Pollution Assessment

Figure 5 illustrates the Contamination Factor (CF) for various elements. Notably, Zn exhibited CF values ranging from 1.82 to 27.98, indicating very high contamination, while Zr and Rb displayed minimal contamination. This suggests that Zn may be a key indicator of localized pollution sources. Significant contamination was found in a number of schools, especially in S2-63.73, S14-26.71, and S23-31.89, according to the modified Contamination Factor (mCF), which is shown in Fig. 6. Fig. Mean value was 11.39 ± 12.99, Min value 4.2 and Max value was 63.73. 7's Pollution Load Index (PLI) additionally identified schools with high pollution levels, specifically S2, S14, and S23, showing deteriorating environmental conditions in these locations. CF quantifies enrichment but cannot distinguish natural versus anthropogenic sources.

Table 7. Mean concentration of sampled schools in mg/kg

mg/kg	As	Cu	Zn	Zr	Sr	Rb	Pb	Cr	V	Sc	Fe
S1	13.11±4.29	66.93±7.51	1873.24±20.23	62.95±4.31	56.7±1.9	14.2±1.12	116.48±5.12	164.35±9.19	33.19±18.93	176.1±45.55	3707.1±55.51
S2	33.93±7.78	91.33±11.49	16131.34±71.26	61.91±5.4	128.65±3.13	9.66±1.29	415.47±10.24	192.09±10.84	150.86±35.58	202.56±69.52	8549.42±101.7
S3	8.34±4.27	80.72±9.46	632.33±15.12	61.2±5.2	143.53±3.13	14.07±1.33	63.3±5.08	76.56±7.18	26.17±15.02	207.88±48.31	8767.56±98.11
S4	4.55±3.72	199.18±9.76	598.57±12.48	38.37±4.32	52.04±1.91	10.25±1.08	72.44±4.48	108.41±8.09	49.77±16.45	174.32±38.48	2504.3±48.28
S5	8.24±2.74	70.31±7.74	1186.7±16.27	25.12±4.14	58.97±1.96	9.61±1.04	39.57±4.12	158.74±8.93	29.54±15.04	226.99±40.52	5574.71±65.43
S6	7.29±3.67	81.37±8.01	420.5±10.96	32.68±4.21	53.76±1.92	13.06±1.12	65.56±4.34	221.18±9.55	29.41±22.25	207.08±48.79	3781.24±58.25
S7	15.42±3.68	51.28±7.3	839.02±14.8	32.1±4.18	52.8±1.88	13.0±1.12	82.94±4.8	145.07±8.73	29.93±16.33	190.16±39.93	5268.82±67.38
S8	12.68±4.13	55.68±7.8	2464.17±25.59	44.08±4.4	68.21±2.12	12.62±1.14	90.84±5.04	135.88±8.7	37.04±21.34	231.05±51.97	3079.35±53.97
S9	13.32±3.74	68.52±8.9	871.25±16.6	57.65±5.04	161.85±3.13	15.0±1.3	107.49±5.38	121.79±8.21	33.7±16.71	336.43±64.65	7588.79±89.57
S10	5.03±2.1	82.82±7.41	611.95±12.26	42.86±3.97	27.85±1.49	14.0±1.07	5.3±3.99	90.99±8.47	37.26±15.51	126.04±35.15	3357.46±51.65
S11	6.64±4.19	159.6±9.5	1522.27±20.06	37.82±4.55	127.58±2.74	11.03±1.14	86.2±4.81	61.78±7.77	25.7±21.85	227.53±56.01	7724.22±81.38

S12	15.64±5.46	64.46±8.3	1595.86±21.04	31.88±4.44	68.24±2.17	9.53±1.1	151.13±6.08	146.11±8.78	34.58±15.29	196.8±44.19	6438.4±77.65
S13	6.61±2.43	58.87±7.27	497.3±11.56	27.92±4.03	29.51±1.57	7.76±0.98	16.41±3.69	132.96±8.47	32.78±14.92	132.76±35.53	1621.9±38.4
S14	69.96±13.05	88.26±14.34	4323.6±54.58	27.65±7.03	124.73±4.26	16.12±2.0	564.25±16.81	175.0±14.1	nan±42.49	220.03±99.47	25649.29±240.61
S15	5.49±3.32	121.6±10.7	470.05±14.36	20.03±5.16	66.24±2.57	10.16±1.32	22.38±4.46	99.16±12.74	nan±29.93	270.63±96.54	3530.75±70.57
S16	7.7±4.67	53.51±11.57	219.5±12.62	37.51±7.52	470.53±7.18	9.22±1.7	40.59±6.21	nan±14.43	68.05±23.12	811.54±154.92	2122.49±67.9
S17	5.25±2.7	121.13±9.91	352.42±11.9	18.97±4.88	71.38±2.5	10.34±1.26	nan±5.44	118.75±13.8	nan±31.46	307.46±85.7	3225.05±63.99
S18	nan±4.69	93.04±9.44	514.97±14.43	34.45±5.15	92.09±2.8	14.58±1.4	20.36±4.25	60.82±14.06	nan±62.68	306.15±84.11	3860.89±71.04
S19	5.65±2.93	95.16±10.75	235.32±11.2	32.85±5.59	56.53±2.59	8.58±1.36	nan±5.87	131.14±12.29	48.06±24.4	372.64±74.36	3022.12±69.74
S20	13.83±4.83	55.36±9.13	2228.31±30.01	33.83±5.33	57.17±2.46	13.42±1.44	66.23±5.64	198.55±14.67	46.68±26.95	176.58±73.75	3780.5±72.47
S21	5.84±2.89	73.2±9.21	409.87±13.35	27.41±5.19	76.72±2.68	14.65±1.43	nan±5.8	104.99±12.96	nan±31.76	155.36±73.91	2942.69±64.1
S22	nan±4.03	56.05±8.77	332.52±12.2	29.78±5.16	38.38±2.15	8.18±1.26	nan±5.54	143.3±13.24	nan±29.57	118.25±57.63	1569.18±48.34
S23	nan±7.48	395.36±19.02	6252.0±53.65	42.52±6.69	162.7±4.36	16.77±1.83	67.91±6.8	120.12±13.75	125.67±44.64	204.96±86.3	4463.35±94.49

S24	nan±5.54	87.28±10.68	599.02±17.14	24.76±5.57	72.72±2.85	12.53±1.49	46.88±5.63	87.28±13.1	42.01±27.28	193.58±68.57	10063.49±118.68

BDL = Below Detection Limit

Table 8. Dust concentrations were found in similar research (mg/kg).

Locations	Cr	Cu	Zn	Pb	Fe	As	References
Malaysia, Indoor dust	16.88	30.19	148.71	31.24	4225.33	-	Muhamad-Darus et al. 2017
Iraq, Indoor dust	65.68	54.28	43.90	51.46	-	-	Radhi et al. 2021
Hong Kong, Indoor and Outdoor dust	-	247.38	2293.56	199.96	-	-	Sulaiman et al. 2017
South Africa, Outdoor dust:							
School A	87.90	38.00	148.70	12.45	-	2.90	
School B	37.13	60.97	107.35	15.86	-	0.78	
School C	57.45	7.78	45.10	8.08	-	1.66	
School D	82.40	28.66	315.10	52.68	-	1.60	Olowoyo et al. 2016
School E	45.60	44.91	9.53	16.49	-	2.07	
School F	34.90	116.80	37.50	24.43	-	0.99	
School G	24.35	66.85	37.90	62.85	-	1.82	
School H	27.15	416.65	41.10	184.20	-	0.96	

Locations	Cr	Cu	Zn	Pb	Fe	As	References
Nigeria, Indoor dust	41.80	12.70	121.00	27.60	13.70	2.04	Olujimi et al. 2015
Ghana, Indoor dust	381.30	-	-	4.82	-	-	Nkansah et al. 2015
Istanbul, Outdoor dust	254.00	513.00	1970.00	192.00	-	-	Kurt-Karakus et al. 2012
Istanbul, Outdoor dust	89.00	200.00	984.00	30.00	-	-	
Warsaw, Outdoor dust	90.00	109.00	1070.00	124.00	-	-	
New Zealand, Indoor dust	-	-	21700.00	724.00	-	-	
Greece, Outdoor dust	87.00	-	1505.00	133.00	-	-	Yaparla et al. 2019
Sydney, Indoor dust	90.00	-	1876.00	299.00	-	17.60	
Ottawa, Indoor dust	86.70	206.00	717.00	406.00	-	7.30	Doyi et al. 2019
Canada, Indoor dust	117.00	279.00	833.00	210.00	-	13.10	
USA, Indoor dust	-	-	876.00	109.00	-	6.30	
Sydney, Indoor dust	65.00	-	372.00	76.00	2790.00	-	
China, Indoor and Outdoor dust	149.20	70.80	461.50	180.90	-	13.20	Chen e al. 2013
Hermosillo, Outdoor dust	-	26.34	387.98	36.15	-	-	Chen et al. 2016
Iran, indoor dust:							

Locations	Cr	Cu	Zn	Pb	Fe	As	References
Cold Season	67.00	158.00	513.00	56.00	-	-	Naimabadi et al. 2020
Warm Season	97.00	127.60	666.70	292.00	-	-	
Tokyo and Hiroshima	67.80	304.00	920.00	57.90	-	-	

Enrichment Factor. To evaluate the level of metal contamination across various schools, It has been calculated the Enrichment Factor (EF). Fig. 8 depicts the enrichment factor of samples for each element across the areas. It has been found that most areas ranged from not being polluted to being extremely polluted. Notably, elements such as Cu, Zn, and Sc exhibited extremely severe enrichment, followed by As, Pb, and which had above severe enrichments. Conversely, Zr and Rb had no enrichment to moderate enrichment. Of all the elements, Zn and Cr had the highest enrichment, as observed across all samples. Interestingly, similarities were observed across all school samples. Elements with low natural background concentrations can exhibit high EF values even with slight increases in concentration, potentially leading to overestimation of anthropogenic impact, when the actual impact might be less significant.

Figure 5. Results of contamination factor

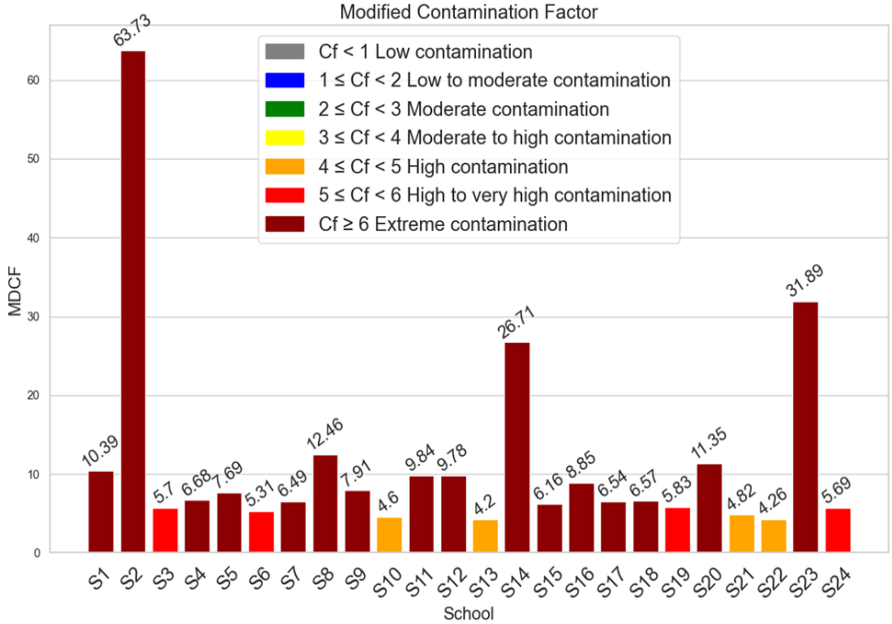


Figure 6. Modified Contamination factor results of sample schools.

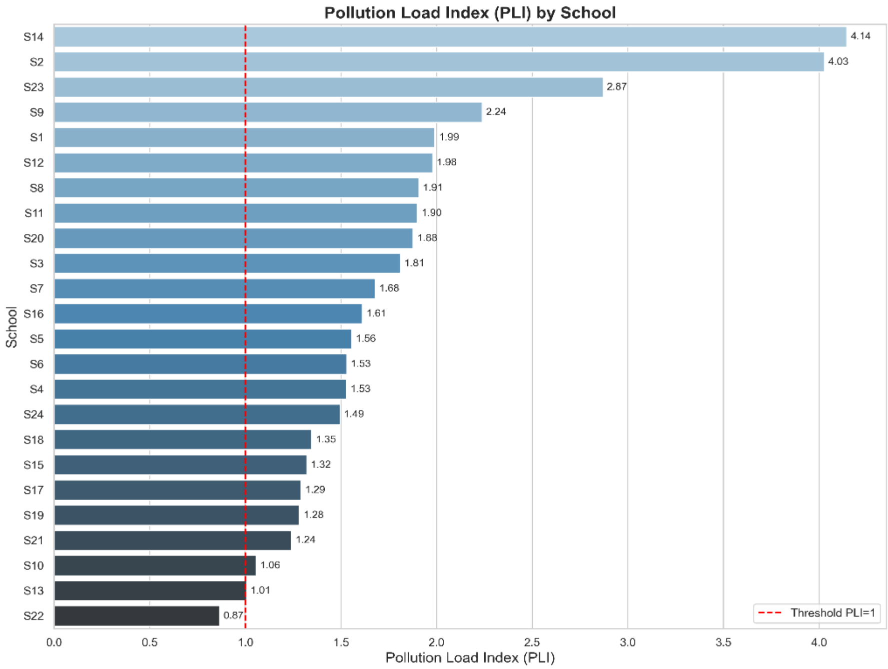


Figure 7. Pollution load index levels for each school

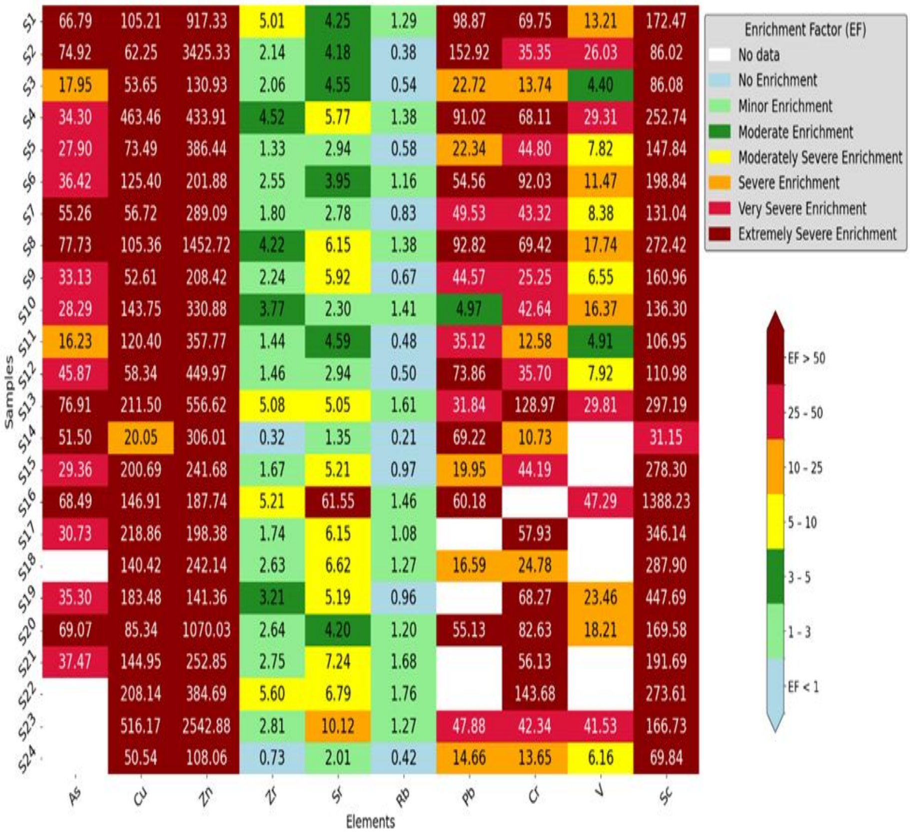


Figure 8. Enrichment values of samples from schools

Geo Accumulation Index. Fig. 9 shows the Geo-accumulation Index (Igeo) heatmap, which shows different amounts of contamination in different samples. Zn was found to be highly contaminated (Igeo > 5) in multiple samples (1, 2, 8, 11, 12, 14, 20, and 23). Igeo readings for Pb, Cr, and As indicated important pollution, although not to an excessive amount as Zn. Samples S23 and S16, however, revealed extreme Cu and Sc contamination. This analysis emphasizes how important it is to assess heavy metal pollution in environments such as schools using the Geo-accumulation Index.

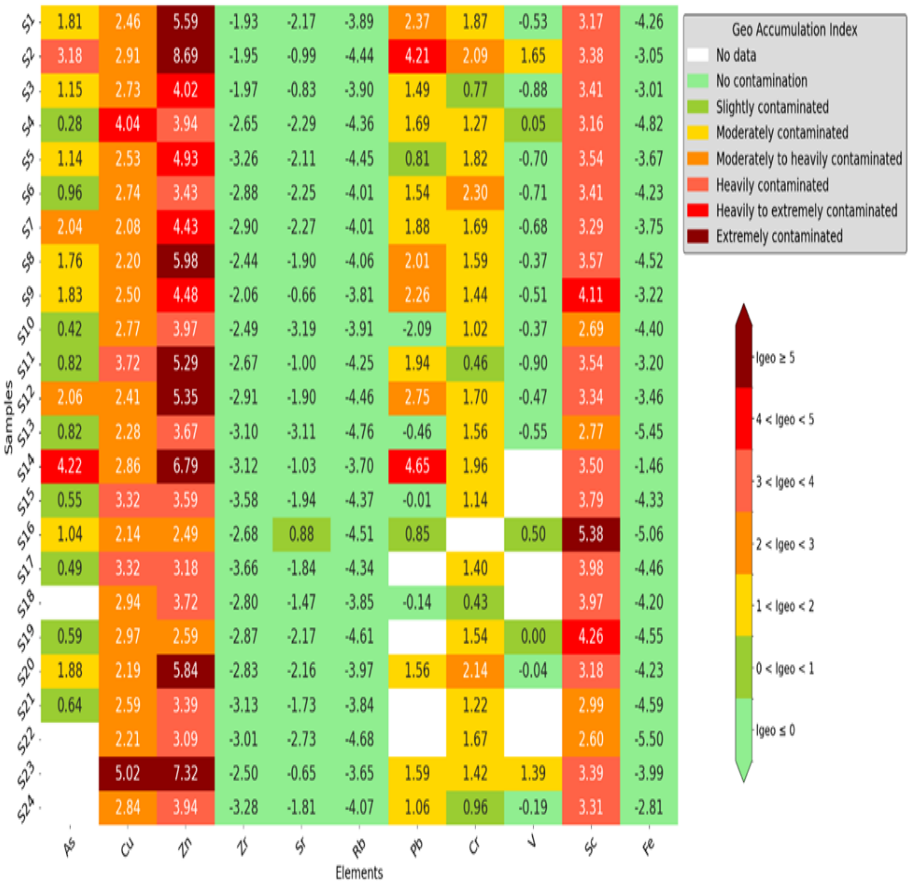


Figure 9. Geo accumulation index results

Principal component analysis. One way to extract information on heavy metals is to use multivariate statistical methods, such as the PCA that Fig. 11 illustrates. The correlation results showed a high complexity among the elements, which required further analysis to classify them and determine their origins (Zheng et al. 2012). PCA was an effective method to identify the pollution sources in this study. PCA has been frequently used to detect pollution sources because it efficiently reduces the number of variables and so facilitates examination of the correlations between the observed variables. Significant correlations between heavy metal pairings, in general, indicate a common or combined origin, whereas weak correlations indicate separate origins (Zheng et al. 2012).

The PCA was initially performed on the full dataset of elemental concentrations. The cumulative explained variance indicated that the first PC1 accounted for 33.62% of the total variance, while the addition of the second PC2 increased the cumulative variance explained to 52.57%. Based on these results, the two most significant principal components were retained for subsequent analyses, as they captured over 50% of the overall variance and provided a simplified yet robust representation of the dataset. The loadings for PC1 and PC2 (see supp. table) revealed distinct patterns in elemental associations. Notably, elements such as As, Pb, Fe and Cr exhibited relatively high positive loadings on PC1. This suggests that PC1 may be strongly influenced by anthropogenic activities such as vehicular traffic, known source of such contaminants. In contrast, PC2 was characterized by higher loadings for Sr and Sc, which are often associated with crustal or geogenic inputs. Thus, the PCA not only reduced the complexity of the dataset but also hinted at the presence of multiple source types contributing to the dust composition.

When the PCA was extended to three components, the cumulative variance explained rose from 52.6% to about 67.8% and loadings were in Cu, Zn, Zr, Rb, V. Combining PCA with clustering techniques effectively revealed the underlying structure of the elemental data. This approach not only reduced the complexity of the dataset but also provided meaningful insights into the potential sources, associations, and geochemical behaviors of the elements studied. The findings suggest that certain elements may originate from similar sources or undergo comparable environmental processes, which is valuable information for interpreting geochemical patterns and assessing environmental impacts in the study area.

Given that all samples were collected from schools, and there were no municipal incineration or industrial sites involved in the city where samples were taken, the similarities become even more notable. To fully understand why particular elements, stand out, more research is required. It can be the result of different sources for these elements or different distributional processes.

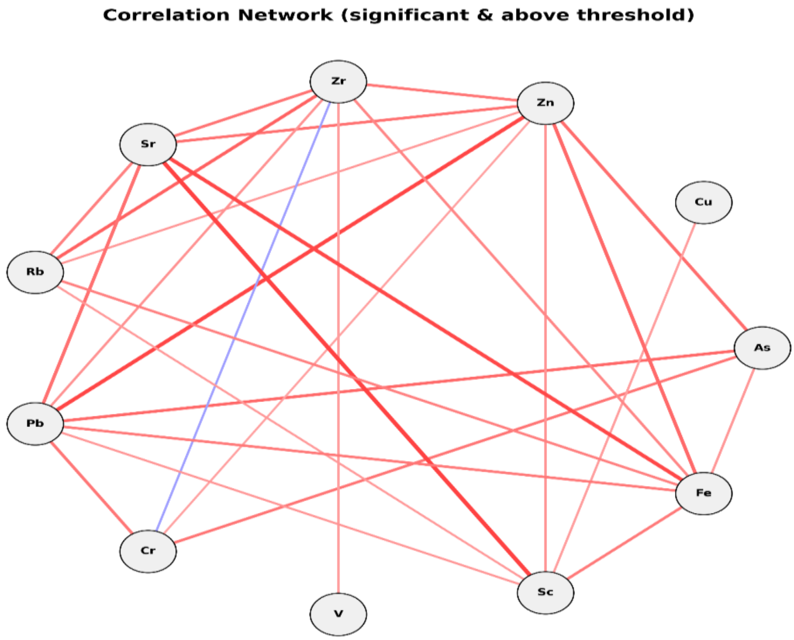
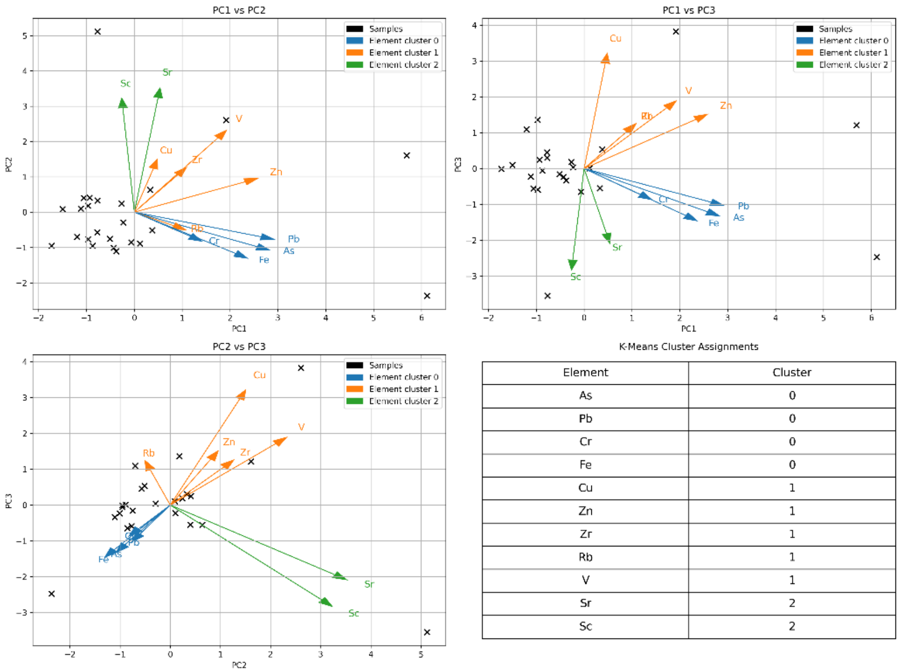


Figure 10. Correlation Network matrix



Element	Cluster
As	0
Pb	0
Cr	0
Fe	0
Cu	1
Zn	1
Zr	1
Rb	1
V	1
Sr	2
Sc	2

Figure 11. Principal Components Analysis with Biplot and K-means clustering

Hierarchical clustering analysis.

Complementing the PCA, Hierarchical Cluster Analysis (HCA) was carried out using Ward's linkage method, which minimizes the total within-cluster variance and verifies the robustness of PCA K means clustering. The HCA dendrogram (Figure 12) alongside the elbow method based on the within-cluster sum-of-squares (WCSS) suggested a potential grouping structure in the data, this information used for K-means clustering in PCA as well. However, a more refined determination using silhouette scores across various cluster solutions revealed that a three-cluster solution (silhouette score = 0.69) provided the most robust partitioning of the dataset. The three clusters derived from the analysis were interpreted as follows as similar to PCA:

- **Cluster 1** included elements such as Sr and Sc, which may be indicative of a dominant contribution from natural, crustal materials.
- **Cluster 2** comprised elements like As, Pb, Cr, and Fe. The strong loadings of these elements on PC1, coupled with their known associations with industrial and vehicular emissions, suggest an anthropogenic influence.
- **Cluster 3** was characterized by elements such as Cu, Zn, Zr, Rb, and V, which could represent a mix of urban-related activities distinct from those dominating Cluster 2.

These groupings, when cross-referenced with the spatial distribution of the schools, revealed that the schools could be classified into three distinct clusters based on their dust elemental profiles. Dust metals were probably well mixed through transportation by winds (Chang et al. 2009). The mean feature values computed for each cluster further reinforced these differences, with one cluster exhibiting relatively higher concentrations of potential toxicants, thus flagging it for further investigation in the context of human exposure and health risk assessments.

The combined PCA and HCA analyses provide a compelling picture of the multi-elemental dust composition around the sampled schools. The explanation of three clusters suggests that dust contamination in these environments is not uniform but instead reflects a mixture of source contributions. The clustering of anthropogenically associated elements (As, Pb, Cr, and Fe) in one group raises concerns regarding localized industrial or traffic-related emissions, which could have direct implications for the indoor air quality and health of school occupants. Conversely, the grouping of elements typically derived from natural sources (Sr and Sc) in a separate cluster implies that background, geogenic dust is also a significant component of the overall exposure. By comparing these groups with where the schools are located, our study showed that there are three main types of dust profiles across the schools.

Road dust contamination is largely attributed to tire wear, brake lining and road surface abrasion, leading to the presence of various heavy metals (Kummer et al. 2009; Adamiec et al. 2016). Metals stemming from vehicles, such as Cu, Zn, Cd, and Pb, primarily originate from wear and tear rather than combustion processes (Sternbeck et al. 2002). Especially, asphalt and sandpaper-like effects contribute to As levels, As also can increase in the environment through use of arsenic-based pesticides. The heavy metal concentrations in road dust are significantly affected by vehicle operation and road type. Brake dust, containing elements like Fe, Zn, Pb, Cu, and Cr, further adds to the contamination, with Cu and Cr being key tracers of non-exhaust brake and tire wear emissions (Adamiec et al. 2016; Sternbeck et al. 2002). Road travel has historically been a major source of Pb emissions, notably in Europe, where leaded gasoline was widely used. The 2010 leaded gasoline ban is expected to result in a considerable reduction in airborne Pb emissions (Zheng et al. 2012; Kummer et al. 2009).

The absence of proper ventilation, combined with the increased use of products like batteries, cell phones, and lights, worsens indoor air quality by limiting pollutant dilution (Jha et al. 2020) Many batteries exceed EU limits, and frequently, those containing excessive levels of Pb are not properly labeled (Recknagel et al. 2014). Furthermore, compact fluorescent lamps (CFLs) and LED bulbs are found to have high concentrations of Cu, Pb, and other heavy metals, posing risks to indoor air quality (Lim et al. 2013).

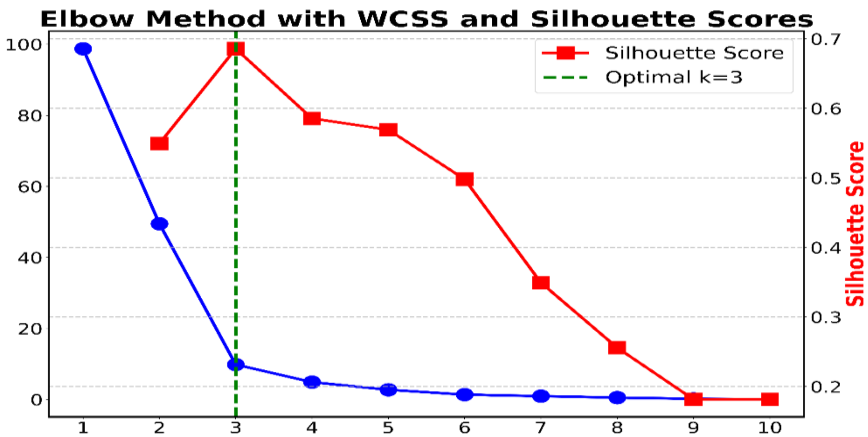


Figure 12. Optimal clusters for all sampled schools

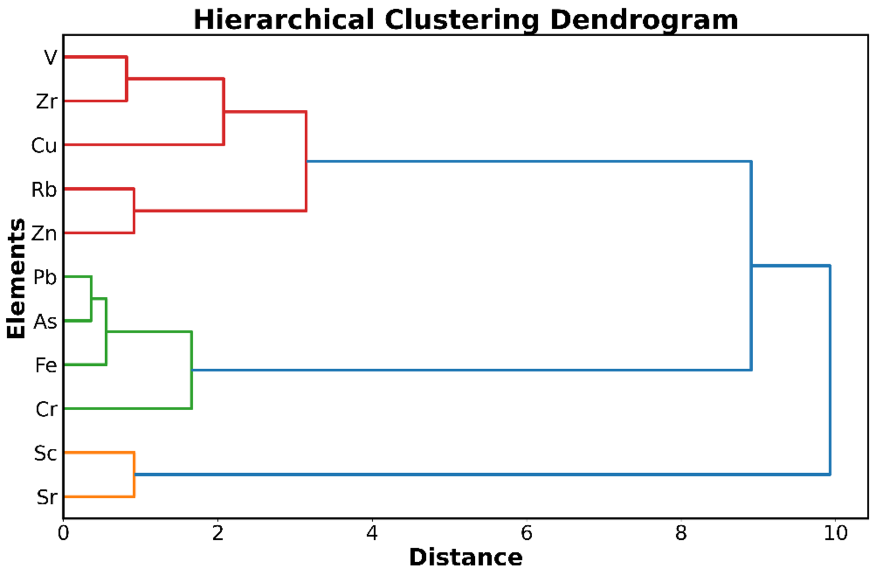


Figure 13. Hierarchical dendrograms for all sampled elements

Spearman Correlation Analysis. To better understand the relationships among various dust elements, a Spearman correlation analysis was performed (Fig. 10 and Supp Fig.). This non-parametric approach evaluates the strength and direction of monotonic associations between pairs of elements, thereby offering insights into common sources or similar geochemical behaviors. It is important to note that, in the Vilnius area, there is no significant industrial activity; thus, the observed correlations should be interpreted within the context of other urban and natural sources.

- **As and Zn:** A moderate positive correlation ($\rho = 0.461$, $p = 3.70\times10^{-6}$) was observed between As and Zn. Although such an association is often linked to industrial emissions in heavily industrialized areas, in Vilnius it likely reflects other anthropogenic inputs such as traffic emissions or domestic heating.
- **As and Pb:** A similar moderate correlation exists between As and Pb ($\rho = 0.495$, $p = 5.40\times10^{-7}$). In the absence of significant industry, this co-occurrence may point to vehicular emissions or even long-range atmospheric transport of these elements from other regions.
- **As and Cr:** A positive correlation ($\rho = 0.433$, $p = 1.59\times10^{-5}$) links As with Cr, another element commonly present in urban dust.
- **As and Fe:** A significant though somewhat lower correlation ($\rho = 0.257$, $p = 0.0136$) implies that Fe may also contribute to the same source signatures as As.

Anthropogenic elements Zn, As, Pb, are found for Kanpur, Beijing, Dhaka, and Hanoi; these abundances are in alignment with the enrichments seen for these HMs at these sites, and with the higher PM2.5 levels at these sites (McNeill et al. 2020).

- **Zn and Fe:** A moderate positive correlation ($\rho = 0.481$, $p = 1.18\times10^{-6}$) between Zn and Fe suggests that these elements might share common sources. Given the limited local industrial activities, this relationship is likely influenced by a mix of urban activities and natural processes such as soil dust resuspension. Supporting this, analysis of PM2.5 in the indoor air of urban schools reported pooled concentrations of 17.32 ng/m^3 for Zn and 14.49 ng/m^3 for Fe, observed across different countries, indicating that both elements are consistently present (Fakhri et al. 2022). With **Zr** ($\rho = 0.437$, $p = 1.35\times10^{-5}$), **Sr** ($\rho = 0.455$, $p = 5.18\times10^{-6}$), **Rb** ($\rho = 0.272$, $p = 0.00862$), **Pb** ($\rho = 0.657$, $p = 1.11\times10^{-12}$), **Cr** ($\rho = 0.210$, $p = 0.0444$), **Sc** ($\rho = 0.270$, $p = 0.00927$)
- **Sr, Sc, and Fe:** A strong positive correlation is observed between Sr and Sc ($\rho = 0.670$, $p = 2.77\times10^{-13}$), and Sr also shows a robust correlation with Fe ($\rho = 0.643$, $p = 4.65\times10^{-12}$) as well as between Sc and Fe ($\rho = 0.40$, $p = 2.1\times10^{-}5$). These relationships are typically indicative of natural contributions. These strong correlations reinforce the role of geogenic sources, such as soil and mineral dust, in contributing to the overall dust composition (Bradford et al. 1996).
- **Cu's Weak Associations:** Cu generally exhibits weak and statistically non-significant correlations with other elements (for example, Cu with As has $\rho = -0.10$, $p = 0.363$). This divergence suggests that copper's presence in the dust may be governed by more localized or specific processes that differ from those affecting the other elements. In an urban setting like Vilnius, this could be due to particular sources such as corrosion of building materials or localized vehicular wear rather than broad-scale anthropogenic emissions. Urban dust, with its varying particle-size distribution and large surface area that facilitates heavy metal deposition and transport, is mainly derived from industrial activities and coal combustion, while traffic emissions serve as an important, albeit secondary, source of pollution (Kaonga et al. 2021).
- **Zr and Rb:** A significant positive correlation between Zr and Rb ($\rho = 0.455$, $p = 5.30\times10^{-6}$) indicates that these elements might share similar mineralogical or geochemical characteristics. In the context of Vilnius, this further points to the influence of natural soil dust rather

than industrial inputs. Umeobi et al. (2024) found similar correlations in their study.

- **Cr and As:** A moderate positive correlation between Cr and As ($\rho = 0.325$, $p = 0.00156$) may reflect certain shared sources. This relationship might also be influenced by traffic or other urban-related emissions rather than industrial processes.

A study published in 2021 by Fiala and Hwang in Houston, Texas, reveals that pavement wear is a major source of heavy metals in road dust, with both asphalt and concrete surfaces contributing significantly to V, and Cr levels, while concrete pavement, likely due to its fly ash content, accounts for about 45% of the total Pb in concrete road dust compared to only 27% from asphalt pavement. The association between Sr and Fe could also indicate involvement in iron oxide formation or adsorption processes. Iron oxides are known for their ability to adsorb various ions and pollutants, and Sr's potential association with iron oxides may enhance these adsorption processes. Studies have shown that iron oxide surfaces facilitate ion exchange and metal sorption, making them vital in environmental remediation and geochemical cycling (Gareev, 2023; Boruah et al., 2023; Song et al., 2022).

While the Spearman correlation analysis may reveal similar patterns to those observed in Principal Component Analysis (PCA) and Hierarchical Cluster Analysis (HCA), it provides additional insights into the strength and direction of monotonic relationships between specific elements. This analysis refines our understanding of the interactions between these elements by quantifying pairwise relationships, which multivariate techniques may not fully capture. Spearman correlation coefficients and their statistical significance offer detailed information on how closely two metals are associated, enhancing the interpretation of the multivariate analyses.

Generally, these analytical methods serve different purposes:

- PCA is designed to reduce dimensionality by identifying the directions (principal components) in which the data vary the most. While it reveals overall structure and variance patterns, it does not provide direct information about the relationship between any two individual variables.
- HCA organizes variables (or observations) into clusters based on a defined similarity or distance measure. The resulting dendrogram visually represents these groupings and the relative distances between clusters, making it useful for exploring the inherent structure in the data.
- Spearman's rank correlation coefficient assesses monotonic relationships (whether linear or not) by comparing the ranks of the data rather than their raw values. This makes it particularly useful when the data is non-

normal or when outliers are present, as it does not require the strict assumptions of Pearson's correlation.

Source apportionment of metals using PMF. In our PMF (Figure. 14) analysis of ambient dust in Vilnius, six factors were resolved. Each factor exhibits a distinct elemental “fingerprint” that can be linked to known sources. In the discussion below, it has interpreted each factor based on its absolute concentrations, its percentage contribution relative to each species (i.e., “% of species sum”), and its comparison with published source profiles. In particular, Factor 6 is highlighted because its characteristics are consistent with traffic-related non-exhaust emissions.

The numbers in a factor’s profile do not reflect the direct measurements from the samples; instead, they are model-derived “loadings” that indicate the relative contribution of each element from that source. Think of them as a “fingerprint” for that source.

Factor 1:

Concentrations: As: 0.24; Cu: 0; Zn: 41.26; Zr: 4.28; Sr: 55.82; Rb: 0.61; Pb: 0; Cr: 0; V: 1.41; Sc: 72.32; Fe: 141.48.

These “concentrations” are not the same as the concentrations measured in the raw samples. Rather, they are the estimated amounts that each source (or factor) contributes to the samples. They allow us to compare different factors and understand which elements are most strongly associated with each source.

Contributions: Sr and Sc constitute 67.9% and 35.7% of their species, respectively, while virtually no Cu, Pb, or Cr is present.

Interpretation: The dominance of natural crustal tracers (Sr, and Sc) and the near absence of metals commonly associated with anthropogenic activities suggest that Factor 1 represents resuspended natural soil or road dust. This component is primarily of geologic origin rather than from vehicular or industrial activities. Additionally, Elevated Sr levels may result from the production and disposal of electronic devices, particularly fluorescent lamps commonly used in schools, where breakage can release high concentrations of Sr (Khalifa et al. 2022) Sc may be associated with its use as an alloying element with Al, Mg, and Zr (Ghosh et al. 2022).

Factor 2:

Concentrations: As: 1.28; Cu: 12.37; Zn: 87.5; Zr: 0.82; Sr: 0.41; Rb: 1.86; Pb: 0; Cr: 11.56; V: 0; Sc: 17.32; Fe: 3667.1.

Contributions: Approximately 69% of the total Fe is associated with this factor, with moderate contributions from Cu (18.7%) and Cr (8.5%).

Interpretation: While Fe is a common component of natural dust, its exceptionally high concentration in Factor 2 (coupled with moderate Cu and Cr) suggests an anthropogenic influence. In urban settings, high Fe can arise from the abrasion of metallic components on vehicles or road surfaces. This factor may therefore represent a mixed source where both natural and traffic-related abrasion contribute. Fe likely originates from automotive components (e.g., brake pads, discs, exhaust systems) and emissions as well as industrial processes such as steel production, metalworking, and welding (Souto-Oliveira et al. 2021; Straffelini et al. 2021). Cu may be emitted from automotive parts (such as Cu-brass radiators and lubricants) and traffic emissions and industrial gases (Wang et al. 2016; Souto-Oliveira et al. 2021).

Factor 3:

Concentrations: As: 0.087; Cu: 0.892; Zn: 1602.2; Zr: 0; Sr: 4.73; Rb: 0.049; Pb: 0; Cr: 0; V: 1.305; Sc: 0.709; Fe: 285.17.

Contributions: Zn dominates with 73.1% of its total signal coming from this factor.

Interpretation: A very high relative contribution of Zn is characteristic of tire wear and friction materials. The absence of significant signals for other metals further isolates the Zn-rich source, leading to the conclusion that Zn is released from sources such as vehicular exhaust, tire wear, construction activities, road dust, and fossil fuel burning (Faisal et al. 2021; Souto-Oliveira et al. 2021). In addition, Zn-based coatings—widely used to protect steel structures against corrosion—may elevate Zn levels in schools (Saarimaa et al. 2022).

Factor 4:

Concentrations: As: 6.40; Cu: 1.42; Zn: 47.74; Zr: 4.75; Sr: 6.26; Rb: 0; Pb: 90.36; Cr: 11.56; V: 8.61; Sc: 5.69; Fe: 57.35.

Contributions: All the measured Pb (100%) is associated with Factor 4, and it also contains moderate amounts of As (52.3%), and V (23.2%).

Interpretation: A factor that "captures" nearly all Pb is strongly indicative of legacy sources. In urban environments, residual Pb from past use of leaded gasoline or other historical industrial activities is often observed. Hence, Factor 4 is interpreted as a legacy or lead-dominated source. These elements likely stem from industrial activities such as metal processing, coal-fired power generation, and the use of wood preservatives, as well as from the degradation of Pb-based paints in older school buildings. Atmospheric deposition from industrial emissions and heavy traffic also contributes to Pb and As levels (Faisal et al. 2021; Tarvainen etl. 2020). Notably, Lithuania was among the largest exporters of paints by volume in 2008–2009, which may have influenced historical contamination (O'Connor et al. 2018). Pb has also been identified in heavy traffic road dust (Murakami et al. 2007). In Lithuania Pb high overall from energy production with a shift toward non-exhaust sources in Lithuania. V is likely derived from domestic heating and automotive traffic (Soldi et al. 1996)

Factor 5:

Concentrations: As: 1.18; Cu: 20.56; Zn: 293.15; Zr: 30.96; Sr: 8.00; Rb: 6.00; Pb: 0; Cr: 0; V: 13.34; Sc: 37.17; Fe: 187.25.

Contributions: Zr accounts for 75.86% and Rb for 50.92% of their totals, a typical signature of natural soil.

Interpretation: Although there is a moderate Cu signal (31.05%), the strong crustal fingerprint (dominated by Zr and Rb) indicates that Factor 5 represents another portion of the natural, resuspended soil dust. This factor may capture dust that has been mechanically disturbed (for example, from road surfaces) but is of natural origin. Furthermore, Zr and Rb originate from the manufacturing of Zr-based products (paints, ceramics, alloys, catalytic converters, refractory bricks) (Naher et al. 2006) and from industrial production and coal burning (Tóth et al. 2019).

Factor 6:

Concentrations: As: 3.06; Cu: 30.97; Zn: 118.62; Zr: 0; Sr: 6.97; Rb: 3.27; Pb: 0; Cr: 112.86; V: 12.52; Sc: 69.18; Fe: 976.99.

Contributions: Cu: 46.78% of total Cu, Cr: 82.99% of total Cr, V: 33.67% of total V, Sc: 34.18% of total Sc, Zn is present but with only 5.42% of its total, suggesting that its dominant source is captured in Factor 3.

Interpretation: Factor 6 has a clear traffic-related signature. Its high relative contributions of Cu and especially Cr are well documented in the literature as markers for brake wear and other non-exhaust vehicular processes as well as combustion of coal or oil (Faisal et al. 2021; Murakami et al. 2007). The moderate V and Sc levels further support the interpretation that this factor represents metals emitted by traffic. Importantly, the absence of Pb in Factor 6 distinguishes it from legacy sources. Taken together, these attributes indicate that Factor 6 is best interpreted as representing traffic-related emissions.

Overall, the PMF analysis indicates that heavy metal contamination in school dust arises from a combination of vehicular emissions, construction materials, and the degradation of older building components.

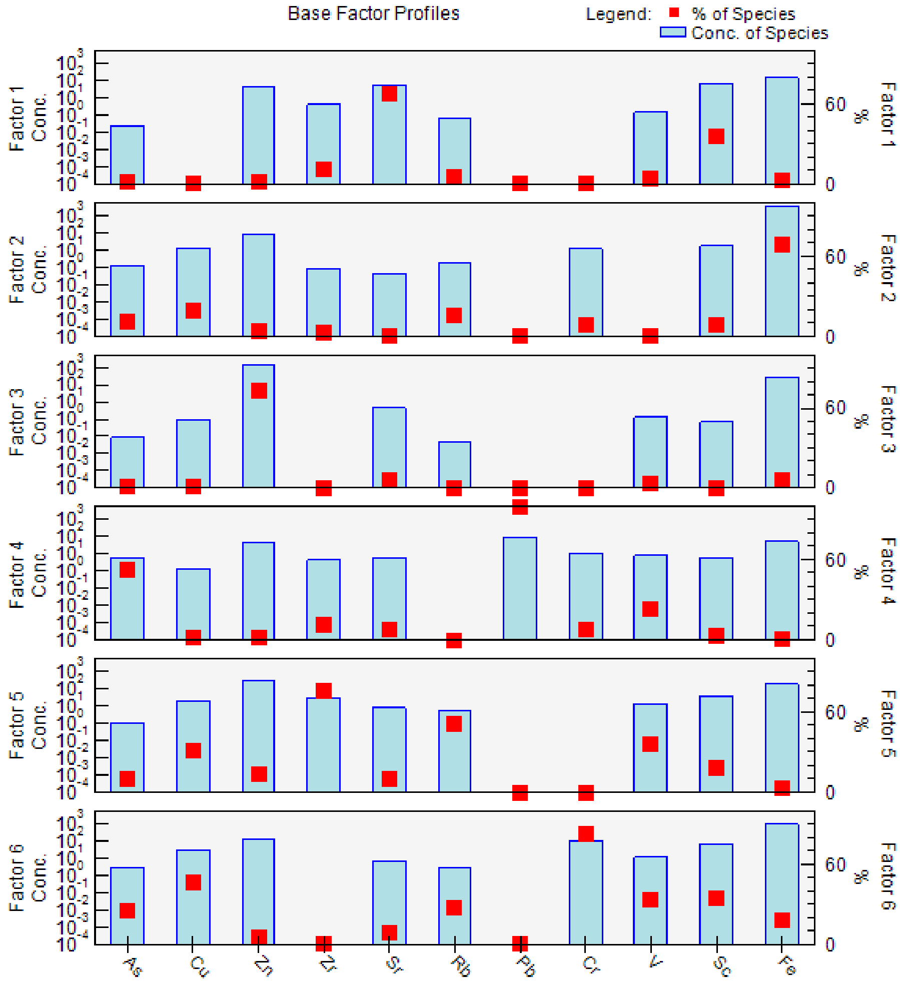


Figure 14. PMF Factors from EPA application

Limitations. This study examining the presence of heavy metal contamination in dust within schools in Vilnius provides valuable insights, however it is subject to various limitations. A limitation of this study is that the 24 selected schools, while varied, may not represent all educational environments in Vilnius. Additionally, the single composite sampling approach may not capture within-site variability. Future research should include a larger sample size, replicate sampling, and seasonal analyses to improve the robustness of the findings. The methodology for collecting and analyzing dust samples, despite its rigor, faces limitations in sample contamination risks and the sensitivity of analytical methods, which could impact the accuracy of metal concentration measurements. The main aim of this study was to evaluate the long-term accumulated of dust and heavy metal pollution in the school environment. However, it did not particularly investigate any changes that occur with the seasons. Gaining comprehension of these seasonal variations could offer supplementary understanding of the temporal patterns of dust and heavy metal accumulation. Hence, although our discoveries provide significant insights into the amounts of pollution over an extended period, further studies that incorporate a seasonal analysis could augment our comprehension of these environmental elements. Health risk assessment, relying on specific assumptions and models, may not comprehensively represent complex real-world exposure scenarios, thereby making the projected risks indicative rather than definitive. Furthermore, it is important to use caution when applying these findings to other situations, taking into account the distinct geographical, environmental, and socio-economic aspects of the schools that were studied. Recognizing these constraints is essential for presenting a balanced perspective on the possible presence of heavy metal pollution in educational environments and its impact on children's health. Further investigation is required to overcome these limitations and gain a more thorough understanding of the situation.

3.1.2 Chapter Conclusion

The comprehensive assessment of heavy metal contamination in dust samples from 24 schools in Vilnius has unveiled significant insights into the environmental quality and potential health risks within educational settings. The following key conclusions encapsulate the primary findings of this chapter:

Elevated Heavy Metal Concentrations in School Dust. Dust samples from Vilnius schools exhibited substantial concentrations of heavy metals, notably As, Cu, Zn, and Pb. The wide range of Cu concentrations (51.28 mg/kg to 395.37 mg/kg) underscores the impact of localized pollution sources, such as proximity to high-traffic areas and urban activities. In contrast, As levels remained relatively consistent across different schools, suggesting a pervasive background source contributing to its presence in dust.

Significant Variability in Metal Concentrations Across Schools. The study revealed considerable variability in heavy metal concentrations between schools, as detailed in Table 7 and Figure 4. This variation indicates the presence of multiple local pollution sources, including traffic emissions, construction activities, and atmospheric deposition. For instance, schools S2, S14, and S23 showed exceptionally high levels of Zn, Cu, and Pb, highlighting the influence of specific environmental factors unique to these locations.

Comparative Analysis with Global Data. When compared to global benchmarks (Table 8), Vilnius schools exhibited higher concentrations of certain heavy metals, particularly Zn and Pb, reflecting the unique environmental and geographical factors influencing pollution levels in the area. This comparative perspective underscores the necessity for localized studies to accurately assess and address heavy metal contamination.

Pollution Assessment Using Contamination Factors and Pollution Load Index. The Contamination Factor (CF) analysis (Figure 5) identified Zn as the most heavily contaminated metal, followed by Cu and As. The modified Contamination Factor (mCF) further pinpointed schools S2, S14, and S23 as hotspots of heavy metal pollution. The Pollution Load Index (PLI) corroborated these findings, indicating deteriorating environmental conditions in the schools, which necessitates urgent remedial actions.

Enrichment and Geo-Accumulation Insights. Enrichment Factor (EF) analysis (Figure 8) revealed extreme enrichment of Cu, Zn, and Sc across most samples, with Zn and Cr displaying the highest levels of enrichment. The Geo-Accumulation Index (Igeo) heatmap (Figure 9) highlighted Zn as highly contaminated in multiple schools, particularly in S1, S2, S8, S11, S12, S14, S20, and S23. These indices collectively emphasize the severe contamination levels and the need for targeted mitigation strategies.

Multivariate Techniques. PCA, HCA, and Spearman correlations effectively reduced data complexity and revealed distinct elemental groupings.

Principal Component Analysis (PCA):

- Two principal components captured over 50% of the total variance.
- PC1: Dominated by anthropogenic markers (As, Pb, Zn).
- PC2: Influenced by geogenic inputs (Sr, Sc).

Hierarchical Cluster Analysis (HCA): Revealed three robust clusters:

- Cluster 1: Natural crustal elements (Sr, Sc).
- Cluster 2: Anthropogenic elements (As, Pb, Cr, Fe).
- Cluster 3: Mixed urban-related elements (Cu, Zn, Zr, Rb, V).

Spearman Correlation Analysis: Confirmed significant pairwise relationships among key elements. Reinforced the grouping patterns observed in PCA and HCA.

Source Apportionment Using Positive Matrix Factorization (PMF). was employed to identify the primary sources of heavy metals in the dust samples. Six factors were identified:

Factor 1: Natural soil dust (high Sr and Sc).

Factor 2: Mixed source with strong Fe, indicating natural and traffic abrasion.
Factor 3: Zn-dominated, linked to tire wear and construction activities, including Zn-based coatings.
Factor 4: Legacy source, capturing nearly all Pb; reflects historical emissions and degradation of older building components.
Factor 5: Natural, resuspended soil dust (high Zr and Rb).
Factor 6: Traffic-related emissions (high Cu and Cr), distinct from legacy sources.

Overall Findings:

- HM contamination in school dust originates from vehicular emissions, construction materials, and the degradation of older building components.
- Spatial patterns indicate heterogeneous exposure among schools, with some sites showing higher concentrations of potential toxicants.

3.2. Assessing the Impact of Outdoor Particulate Matter on Indoor Dust Heavy Metal Contamination

Particulate matter ratio. Figures 15–17 display a clear, temporal trend in the PM2.5/PM10 ratio in Vilnius. A low PM2.5/PM10 ratio indicates a dominance of coarse particles (typically from natural sources like dust or pollen), while a high ratio suggests that fine particles from anthropogenic sources (e.g., traffic emissions, railway activities, burning) prevail (Fan et al. 2021). In Europe, these ratios typically range from 0.39 to 0.74—with Southern Europe on the lower end and Eastern Europe on the higher end (Eeftens et al. 2012). For comparison, São Paulo exhibits ratios between 0.33 and 0.47, and in Saudi Arabia, ratios range from 0.25 to 0.52 (average ≈0.3) (Fan et al. 2021).

Figure 15 also reveals temporal fluctuations, likely influenced by seasonality, vehicular emissions, and residential heating—with weather conditions further modifying pollution levels (Zhao et al. 2019). Ratios greater than 0.5 are considered high and indicative of poor air quality, which can adversely affect respiratory health.

Because PM2.5 particles infiltrate indoor environments more readily, rising PM2.5/PM10 ratios suggest that fine, harmful particulates may increasingly contribute to indoor dust contamination in schools. This is especially concerning for schools located near roads or industrial areas.

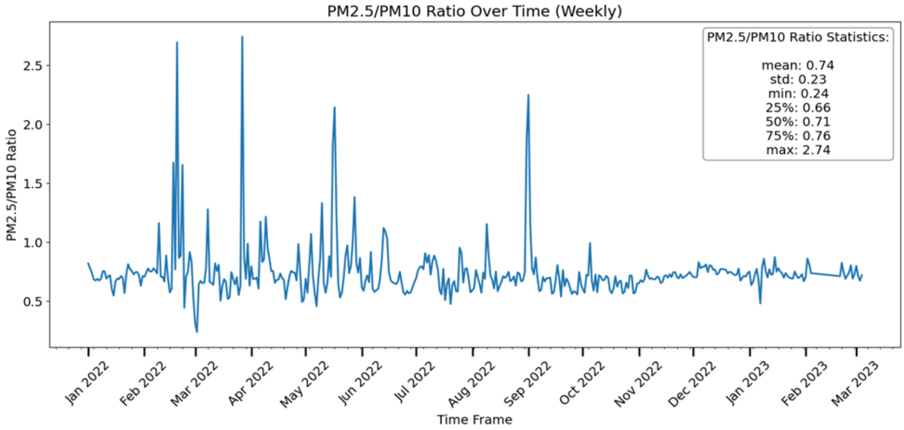


Figure 15. 13 PM 2.5 / PM10 Ratio over a year in Vilnius city (Vilnius Municipality, 2023)

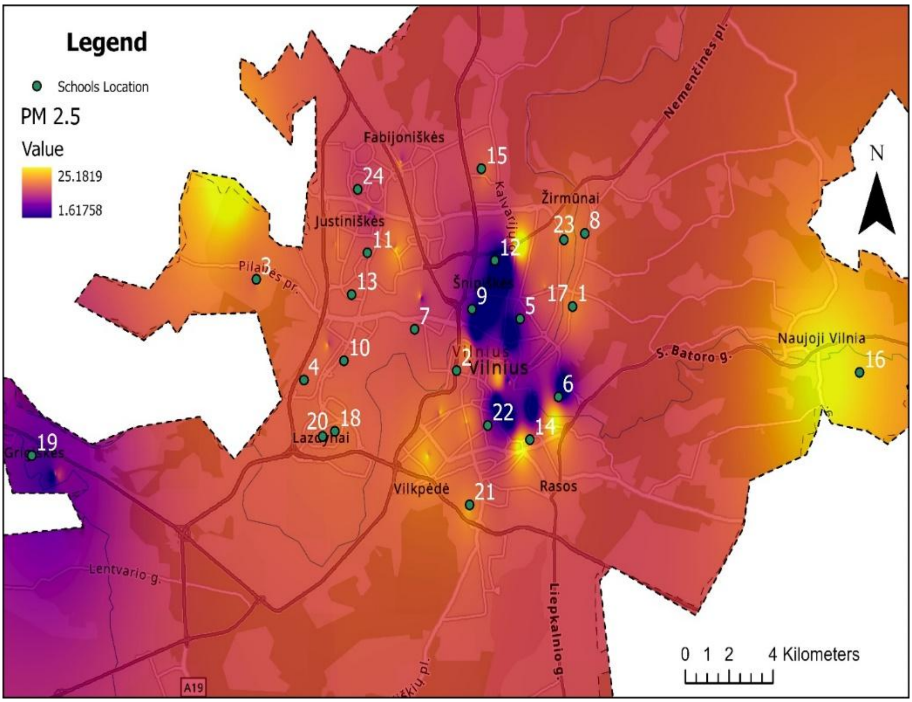


Figure 16. Higher PM2.5 and School locations

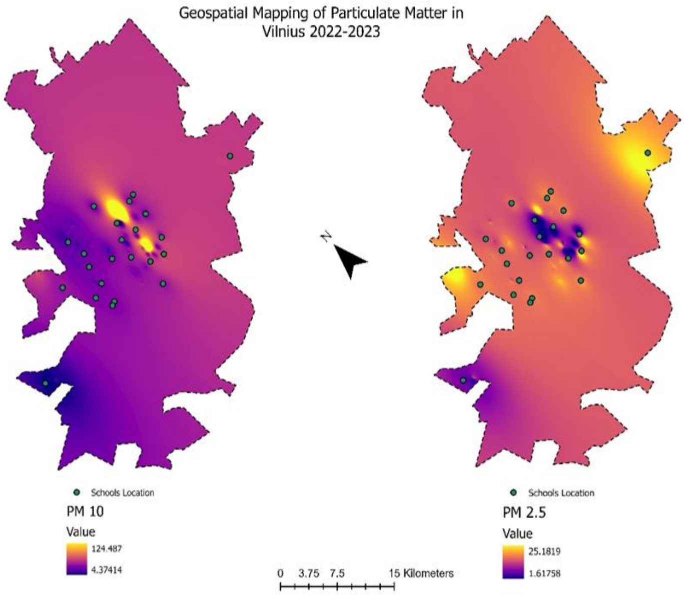


Figure 17. PM 2.5 and PM10 between 2022–2023 in Vilnius city (Vilnius Municipality, 2023).

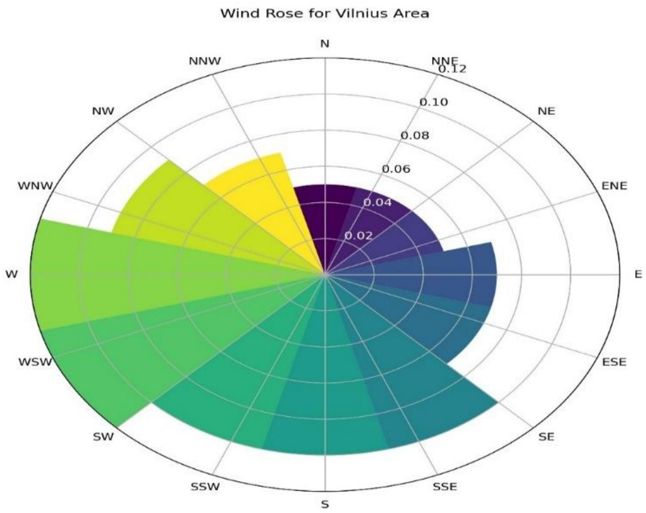


Figure 18. Wind Rose for Vilnius Area (Global Wind Atlas, 2023).

Fig. 18 illustrates the wind rose for Vilnius, indicating potential routes through which winds could transport pollution. Therefore, the PM2.5/PM10 ratios are influenced by multiple factors such as human activities, traffic emissions, and potentially long-range transport (Fan et al. 2021).

Figure 19 illustrates the distribution of distances from each school to the nearest PM sensor, road, traffic light, railway, and bus stop. This analysis provides a comprehensive understanding of the spatial relationships between schools and key environmental and infrastructural features.

Proximity to PM Sensors.

The distance from schools to the nearest PM2.5 sensor varies widely—from 23.54 m (S18) to 2169.92 m (S13), with a median distance of approximately 730 m.

Schools located within 500 m (e.g., S1, S3, S6, S13, S14, S15, S17, S18, S21) are expected to have PM measurements that more accurately represent local outdoor air quality. This proximity is critical because more representative PM data can better explain the potential infiltration of outdoor pollutants into indoor environments. In contrast, schools farther than 500 m might experience spatial variability that could lead to under- or overestimation of local PM levels, potentially affecting the interpretation of associated heavy metal accumulation in indoor dust. Energy Production Sector (including stationary fuel combustion), this sector contributes significantly (~72.8%-82.9%) to national emissions in Lithuania (AAA, 2025).

Proximity to Roads.

The distances between schools and the nearest roads range from as little as 0.07 m (S16) to 60.73 m (S4), with a median distance of approximately 31 m.

Schools extremely close to roads (<10 m, such as S16 and S21) are likely exposed to higher concentrations of vehicular emissions, including particulate matter rich in heavy metals like Zn, Cu, and Pb. Given that vehicles are a primary source of combustion-related pollutants, the short distances suggest that even the majority of schools (with a median distance of 31 m) may be influenced by traffic emissions. This close proximity can enhance the deposition of these pollutants on both outdoor surfaces and subsequently indoors through infiltration. Automobile brake/tire wear and road abrasion: This specific source increased substantially (+83.0% vs. 2005 in 2023) due to increasing vehicle usage despite overall emission reductions (AAA, 2025).

Proximity to Traffic Lights.

The distances from schools to the nearest traffic lights vary between 28.31 m (S13) and 338.29 m (S21), with a median of approximately 145 m.

Schools within 50 m (e.g., S6, S8, S16, S22) are particularly vulnerable to intermittent spikes in emissions caused by stop-and-go traffic at intersections. These transient spikes can result in short-term peaks in particulate matter levels, potentially increasing the deposition rates of heavy metals. Such episodic increases may contribute to the overall variability observed in indoor dust contamination.

Proximity to Railways.

The distances between schools and railways range from 339.55 m (S14) to 10,672.38 m (S24), with a median of approximately 3610 m.

Although the majority of schools are located more than 1000 m from railways—suggesting a lower direct impact—schools within 1000 m (e.g., S14, S16, S21, S22) could be influenced by emissions from trains. Railway emissions may contribute elements like Fe and Cu due to brake wear and engine exhaust. The relatively small number of schools in close proximity means that while the overall impact on indoor dust might be limited, these specific locations require special attention when assessing local air quality influences. Downward trend observed, achieving a 63.5% reduction by 2023 compared to 2005 (AAA, 2025)

Proximity to Bus Stops.

Distances to the nearest bus stops range from 52.94 m (S3) to 678.93 m (S4), with a median of approximately 215 m.

Schools within 100 m of a bus stop (e.g., S3, S6, S7, S8, S10, S13, S14, S16, S18, S19, S21, S22) may experience increased exposure to particulate emissions due to frequent deceleration and acceleration of buses. Bus stops can be hotspots for localized PM emissions, which, when combined with higher pedestrian traffic, may enhance the local concentration of heavy metals in the air. These elevated levels might subsequently contribute to the accumulation of contaminants in indoor dust.

$PM_{2.5}$ and PM_{10} emissions in Lithuania have shifted significantly over the years. While emissions from transport and energy production have declined, household fuel combustion remains a major contributor. At the same time, non-exhaust traffic sources (e.g., brake and tire wear, wear of road surfaces) are becoming an increasingly significant factor (AAA, 2025). Jakimavičius (2011) reports that motorization nearly doubled between 1999 and 2010—from 265 to 560 private cars per 1,000 inhabitants—leading to increased congestion, slower travel times, and higher fuel consumption. Žiliūtė et al. (2010) show that between 2007 and 2009, traffic volumes on key streets varied considerably—T. Narbuto Street saw a 36.29% increase, while other streets experienced moderate increases (6.52–10.18%) or declines of up to 23.13%. In 2009, Geležinio Vilko Street (223,987 vehicles) and Savanorių Avenue (367,991 vehicles) were among the busiest. Notably, heavy trucks—often carrying loads up to 20 tonnes compared to the legal limit of 11.5 tonnes—accelerate pavement wear and generate dust that may carry heavy metals. This traffic analysis underscores the importance of considering not only the proximity of schools to roads but also the intensity of traffic on those roads, as higher traffic volumes can lead to increased emissions of fine particulates and HMs (Supplementary Figure 27).

This information can be useful for identifying which schools have better local data coverage and for assessing the reliability of PM2.5 data at each location (e.g., schools farther from sensors may have less accurate representations of local air pollution). This variability in road proximity may contribute to differences in the levels of metal contamination found in indoor dust at each school, especially for those schools situated closer to major roads where vehicular activity is higher. it is important to note that this PM data covers only one year and was included to provide a broader environmental context rather than being directly tied to the assessment of heavy metal accumulation in indoor dust. The proximity of the sensors may offer additional insight into potential outdoor sources of particulate matter around the schools, but it remains supplementary to the main findings.

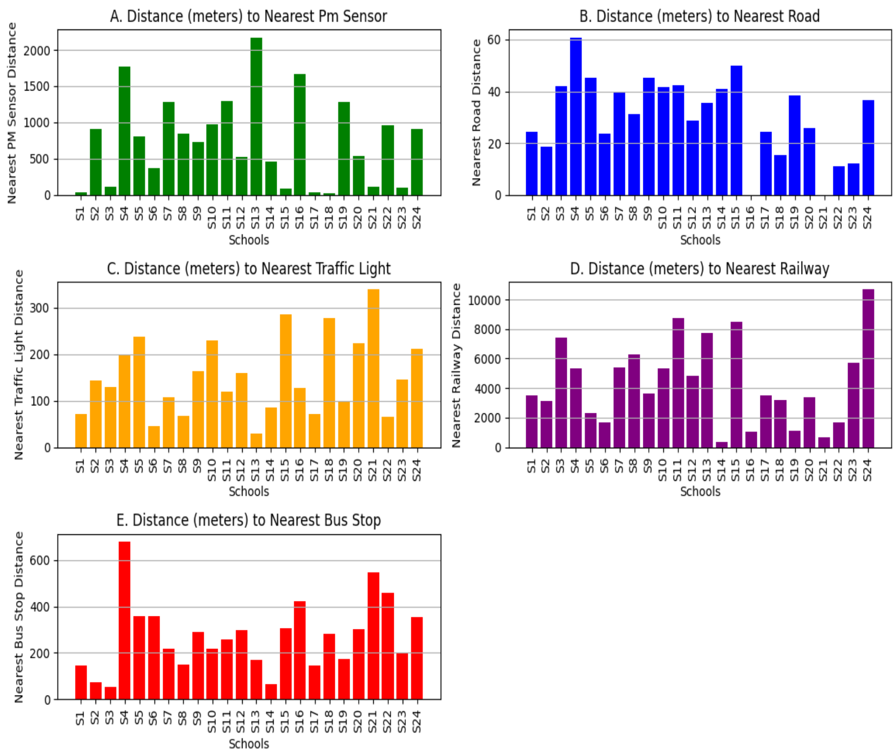


Figure 19. Distance from schools to A- Nearest PM sensors, B- Nearest road, C- Nearest Traffic lights, D- Nearest Railway, E- Nearest Bus stop.

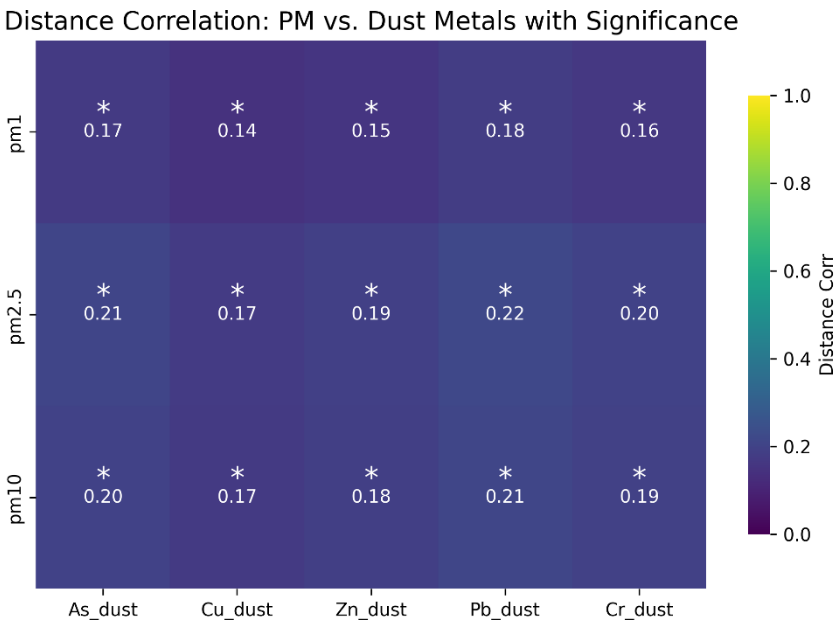


Figure 20. Distance Correlation between PM and Dust Metal (* means $p<0.05$)

Distance correlation analysis revealed weak-to-moderate but statistically significant associations ($p < 0.05$) between particulate matter (pm1, pm2.5, pm10) and dust metal concentrations (As, Cu, Zn, Pb, Cr), with coefficients ranging from ~0.14 to 0.22. These findings indicate that, despite the modest magnitude of correlations, there is a consistent non-linear dependency between PM and dust composition. Overall, our study confirms that dust HM concentrations are significantly related to ambient PM levels, emphasizing the importance of considering both linear and non-linear relationships in environmental assessments. These statistically significant associations suggest that outdoor emissions, likely stemming from traffic and urban activities, are penetrating indoor environments, thereby contributing to the accumulation of toxic metals in school dust. Although based on a single year of data, these findings suggest that outdoor emissions, particularly near dense road networks, contribute to indoor dust contamination, raising concerns about children's exposure to toxic metals in school environments.

3.2.1 Chapter Conclusion

Particulate Matter Trends and Environmental Influences. The increasing PM2.5/PM10 ratios indicate a shift toward finer particles. Finer particulates are more harmful because they can penetrate deep into the respiratory system, posing significant health risks—especially in schools where children spend considerable time indoors.

The outdoor PM data (Figures 15–17) suggest that elevated PM2.5 levels contribute to the heavy metal content in indoor dust. Statistically significant correlations were found between outdoor PM levels and indoor concentrations of As, Pb, Cu, and Zn. Distance correlation analyses reveal statistically significant associations between outdoor PM levels and indoor concentrations of As, Pb, Cu, and Zn, with distance correlation coefficients ranging from approximately 0.14 to 0.22. Although these relationships confirm that outdoor pollution influences indoor dust contamination, other factors (such as indoor sources or building materials) likely also play a role, as indicated by the lack of a significant correlation for Cr. While the initial correlation analysis provides some insights, it's essential to approach the results with caution due to limitations in the data.

Proximity to PM Sensors and Sources. The proximity analysis (Figure 19) reveals that schools closer to PM sensors, roads, traffic lights, railways, and bus stops tend to have higher indoor levels of heavy metals. For example, schools within 500 m of a PM sensor or within 10 m of a major road exhibit higher contamination levels, underscoring the importance of local traffic and infrastructural factors.

The integration of traffic studies with spatial proximity analysis offers valuable insights into the environmental challenges faced by schools in Vilnius. The rapid increase in vehicle numbers and the corresponding variability in traffic volume across different streets—as reported by Jakimavičius (2011) and Žiliūtė et al. (2010)—directly contribute to variations in vehicular emissions. These emissions, when combined with the spatial positioning of schools relative to PM sensors, roads, traffic lights, railways, and bus stops, can significantly influence local outdoor air quality and the subsequent accumulation of heavy metals in indoor dust. Overall, while outdoor PM trends and proximity to pollution sources play a role in shaping indoor dust contamination, other factors, such as indoor activities and building materials, also contribute. It is important to note that the PM data analyzed here covers only one year. Future studies should incorporate detailed spatial-temporal assessments to better understand these relationships.

3.3. Correlating Dust and Surface Soil Contamination in Vilnius Schools

Analysis of soil contamination in Vilnius schools, based on datasets from Kumpienė et al. (2011) and DGE Baltic Soil Monitoring reports (2021, 2023), reveals notable variability in heavy metal concentrations. Zn and Pb emerged as key contaminants. Zn shows significant fluctuations over time, particularly in urban areas affected by traffic and industrial activity (Kumpienė et al., 2011), while Pb levels often exceed permissible limits, indicating strong anthropogenic influence (DGE Baltic Soil Monitoring, 2021, 2023; Kumpienė et al., 2011). In contrast, Cu and Cr are relatively stable, though they occasionally peak due to localized pollution events, and As displays a similar but lower variability pattern. Geochemical mapping by Kadūnas et al. (1999) further indicates that factors such as soil type, altitude, and proximity to urban infrastructure significantly influence metal distribution.

Statistical comparisons between soil and dust samples from 2011 to 2023 (see Table 10) employed non-parametric (Mann-Whitney U) and parametric (Independent t-test, Welch's) tests, based on data normality. To answer; "Do indoor dust samples exhibit significantly different HM concentrations compared to soil, indicating that dust accumulates metals through distinct processes?". The analysis focused on the accumulation of As, Cu, Zn, Pb, and Cr. Key findings include:

- **As:** Dust consistently shows significantly higher As levels from 2017 to 2023, suggesting more efficient accumulation from airborne sources.
- **Cu:** Dust concentrations are significantly higher than soil levels throughout 2011–2023, likely due to traffic emissions.
- **Zn:** Dust exhibits higher Zn levels every year, possibly reflecting inputs from tire wear, industrial processes, and galvanized materials.
- **Pb:** While no significant difference was observed in 2011, dust levels of Pb become significantly higher from 2017 onward, indicating increasing accumulation over time.
- **Cr:** Dust consistently has higher Cr concentrations, with corrosion of Cr-containing materials as likely sources.

For several entries in Table 10, the effect size is listed as "N/A." This notation reflects the following considerations in our analysis: **Non-Parametric Tests:** For comparisons using the Mann-Whitney U test (due to non-normal data), traditional effect size measures like Cohen's d are not directly applicable. **Data Limitations:** In some cases, sample sizes were too small, or the data did not meet the criteria needed for reliable effect size calculations. **Consistency:**

Cohen's d was calculated only for t-tests (Welch's) where its assumptions were met.

The soil metal concentrations in Vilnius schools vary over the years, reflecting a dynamic interplay between pollution control measures, traffic patterns, and urban development. For instance, significant increases in certain metals during specific years may indicate pollution events or shifts in environmental policies. Analysis of soil samples in Šnipiškės reveals pervasive pollution, especially near major streets, where residential and industrial activities appear to correlate with higher contamination levels (Kadūnas et al., 1999).

According to Ahmad et al. (2021), the soil's Cr concentration—which should not exceed the EU standard of 100–150 mg/kg—primarily originates from the migration of landfill leachate. These landfills, containing waste from tanneries, chemical dyeing companies, and industrial products (e.g., cables, steel equipment), contribute significantly to Cr levels. The concentration of Cu in soil has occasionally surpassed the EU limit (50–140 mg/kg), whereas As in both soil and water remains below the EU threshold of 20 mg/kg. Pb levels are generally within the EU standard (50–300 mg/kg), yet its presence is a strong indicator of anthropogenic pollution. Variability in Zn may partly stem from historical agricultural practices; notably, the use of commercial fertilizers—key contributors to soil pollution—increased fourfold from 1960 to the late 1980s (Tumas, 2000).

Differences between soil and dust accumulation may be partly attributed to particle size. Soil contains a broad spectrum of particle sizes, while dust is dominated by finer particles with greater specific surface area, higher organic carbon content, and enhanced cation exchange capacity, which facilitate metal adsorption (Gunawardana et al., 2014; Cao et al., 2012; Lanzerstorfer et al., 2017). Beamer et al. (2012) found that the finest dust particles often have the highest metal concentrations, with levels declining as particle size increases. These characteristics may explain why dust often exhibits higher heavy metal concentrations than soil. Moreover, local factors such as nearby car repair operations, presence of metal acquisition and storage operations in the vicinity and inadequate waste disposal practices can further elevate metal levels in soil (Vilnius Municipality, 2023).

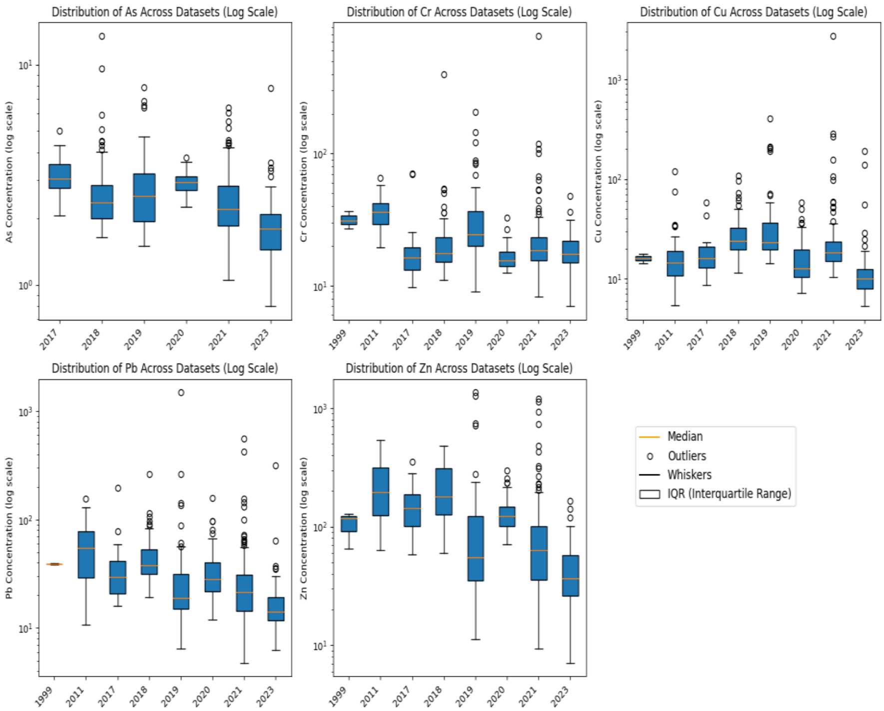


Figure 21. Descriptive analysis of soil datasets from 1999, 2011, 2017, 2018, 2019, 2020, 2021, 2023

Table 9. 1999, 2011, 2017-2021 and 2023 mean Concentration levels of soil studies (mg/kg) (Kumpienė et al. 2011; Kadūnas et al. 1999; DGE Baltic Soil, 2021;2023)

Year	As	Cr	Cu	Zn	Pb
1999	2.5	32.9	8.8	30.9	16
2011	NA	36.22 ± 10.75	18.40 ± 18.58	216.82 ± 122.19	57.97 ± 31.84
2017	3.20 ± 0.68	20.31 ± 15.51	18.41 ± 10.72	150.69 ± 71.07	38.98 ± 35.88
2018	2.91 ± 1.92	26.60 ± 47.71	29.82 ± 18.36	219.50 ± 103.61	48.64 ± 34.39
2019	2.86 ± 1.40	36.35 ± 34.53	45.59 ± 66.75	141.19 ± 259.64	57.00 ± 194.79
2020	2.93 ± 0.32	16.61 ± 3.65	16.65 ± 9.88	131.25 ± 43.00	34.90±23.06
2021	2.42 ± 0.89	18.55 ± 6.30	22.97 ± 16.96	94.84 ± 114.93	26.42 ± 18.83
2023	1.97 ± 1.00	18.58 ± 7.10	16.76 ± 29.24	44.33 ± 32.12	21.58±40.51

Spearman Correlation (Fig. 22). Given the non-normally distributed nature of the data, Spearman's rank correlation coefficient was employed to evaluate monotonic relationships between heavy metal concentrations in paired soil and dust samples across locations and years. This non-parametric method is robust to deviations from normality and effectively assesses the strength and direction of associations between variables.

The analysis identified significant positive correlations for specific metals in certain years, suggesting that elevated soil metal concentrations often coincided with higher dust concentrations. This pattern implies shared contamination sources or intermedia transfer of metals between soil and dust. Key findings include:

- **Zn:** A consistent association was observed across multiple years. In 2017, Zn exhibited a strong positive correlation (Spearman's $\rho = 0.50, p = 0.010$), which persisted in 2018 ($\rho = 0.26, p = 0.034$) and 2020 ($\rho = 0.24, p = 0.043$), indicating a stable soil-dust relationship for this metal.
- **Pb:** A moderate but significant correlation was detected in 2011 ($\rho = 0.48, p < 0.05$), pointing to localized co-occurrence in soil and dust during this period.
- **As and Cr:** Both metals showed significant correlations in 2019 (As: $\rho = 0.30, p = 0.036$; Cr: $\rho = 0.31, p = 0.017$), with weaker but notable associations in 2018 and 2020 ($\rho = 0.22$–$0.29, p < 0.05$).

In contrast, no significant correlations were observed for other media in certain years. This variability may arise from:

1. **Site-Specific Heterogeneity:** Differences in local geology, land use, or anthropogenic inputs (e.g., industrial emissions, agricultural practices) could decouple soil-dust relationships spatially.
2. **Metal-Specific Behavior:** Variations in metal mobility, bioavailability, and partitioning due to chemical properties (e.g., solubility, binding affinity) may influence transport between media.
3. **Environmental Dynamics:** Meteorological factors such as precipitation (affecting leaching) and wind (driving particle resuspension) could transiently alter soil-dust interactions.
4. **Temporal Source Variability:** Distinct contamination sources (e.g., historical soil pollution vs. recent atmospheric deposition) may dominate in different years, weakening cross-media associations.
5. **Lack of Data:** 2011's small dataset.

The persistent correlation of Zn across years suggests a steady anthropogenic source, such as vehicular traffic (tire wear, brake emissions) or industrial effluents, coupled with its natural abundance in regional geology. Conversely, sporadic correlations for As and Cr may reflect intermittent contamination events or seasonally mediated processes.

These findings underscore the complexity of metal distribution in environmental matrices and highlight the necessity of long-term, metal-specific monitoring to disentangle source apportionment and transport mechanisms. The results advocate for integrated soil-dust risk assessments in contaminated ecosystems and indicating that soil and dust may accumulate heavy metals through different mechanisms or rates.

Table 10. The statistical comparison between soil and dust samples.

Year	**Metal**	**Shapiro-Wilk Soil (p-value)**	**Shapiro-Wilk Dust (p-value)**	**Chosen Test**	**p-value**	**Effect Size**	**Significance**
2011	**As**	N / A	N / A	N / A	N / A	N/A	N / A
	Cu	0.010340516	2.84355E-11	Mann-Whitney U test	6.63251E-20	N/A	S D
	Zn	0.192117661	8.12179E-07	Mann-Whitney U test	2.9945E-19	N/A	S D
	Pb	0.651006103	0.904107153	Independent t-test (Welch's)	0.322180527	-0.1648298	NSD
	Cr	0.462614477	4.48803E-05	Mann-Whitney U test	7.17177E-21	N/A	S D
2017	**As**	0.563832164	2.40773E-05	Mann-Whitney U test	2.65376E-13	N/A	S D
	Cu	0.031626713	2.84355E-11	Mann-Whitney U test	1.3873E-13	N/A	S D
	Zn	0.511833549	8.12179E-07	Mann-Whitney U test	3.91615E-14	N/A	S D
	Pb	0.011793762	0.904107153	Mann-Whitney U test	0.008009068	N/A	S D

	Cr	9.81144E-05	4.48803E-05	Mann-Whitney U test	2.15369E-13	N/A	S D
2018	**As**	1.53163E-06	2.40773E-05	Mann-Whitney U test	1.07147E-19	N/A	S D
	Cu	0.008251579	2.84355E-11	Mann-Whitney U test	1.96673E-20	N/A	S D
	Zn	0.000271359	8.12179E-07	Mann-Whitney U test	8.39841E-24	N/A	S D
	Pb	0.004295428	0.904107153	Mann-Whitney U test	0.045259795	N/A	S D
	Cr	1.1123E-09	4.48803E-05	Mann-Whitney U test	9.79019E-24	N/A	S D
2019	**As**	0.005494823	2.40773E-05	Mann-Whitney U test	6.41984E-20	N/A	S D
	Cu	1.77052E-08	2.84355E-11	Mann-Whitney U test	4.23885E-15	N/A	S D
	Zn	0.003882198	8.12179E-07	Mann-Whitney U test	5.30838E-21	N/A	S D
	Pb	3.11788E-06	0.904107153	Mann-Whitney U test	1.07204E-06	N/A	S D
	Cr	0.001547987	4.48803E-05	Mann-Whitney U test	1.00161E-19	N/A	S D
2020	**As**	0.610730767	2.40773E-05	Mann-Whitney U test	1.23221E-24	N/A	S D
	Cu	0.000236045	2.84355E-11	Mann-Whitney U test	5.79468E-27	N/A	S D
	Zn	0.042288542	8.12179E-07	Mann-Whitney U test	1.16026E-27	N/A	S D
	Pb	0.053022694	0.904107153	Independent t-test (Welch's)	2.56981E-05	-0.7407869	S D
	Cr	0.000348318	4.48803E-05	Mann-Whitney U test	1.50477E-26	N/A	S D
2021	**As**	0.007514515	2.40773E-05	Mann-Whitney U test	2.44897E-30	N/A	S D
	Cu	5.02475E-16	2.84355E-11	Mann-Whitney U test	8.84771E-30	N/A	S D
	Zn	0.000518413	8.12179E-07	Mann-Whitney U test	1.3999E-33	N/A	S D

	Pb	1.057E-05	0.904107153	Mann-Whitney U test	1.10828E-09	N/A	SD
	Cr	7.62121E-14	4.48803E-05	Mann-Whitney U test	9.03581E-33	N/A	SD
2023	As	0.034834404	2.40773E-05	Mann-Whitney U test	7.89622E-22	N/A	SD
	Cu	2.10465E-08	2.84355E-11	Mann-Whitney U test	9.87252E-22	N/A	SD
	Zn	0.000705102	8.12179E-07	Mann-Whitney U test	4.24913E-25	N/A	SD
	Pb	3.23148E-06	0.904107153	Mann-Whitney U test	1.36917E-11	N/A	SD
	Cr	0.201838389	4.48803E-05	Mann-Whitney U test	3.90818E-24	N/A	SD

SD= Significant Difference
NSD= Not Significant Difference

Various anthropogenic sources such as infrastructure deterioration, and transportation—are frequently responsible for the presence of heavy metals in urban soil and dust across different schools (He et al., 2022). While these sources are common, the concentrations of HMs they release can vary significantly among individual schools. This variability is influenced by factors including each school's proximity to traffic density in surrounding areas, and the age and condition of school infrastructure. According to Peng et al. (2016) and Mahanta et al. (2010), HMs have a tendency to attach to soil particles, which makes the soil a sink for these metals and causes their levels to fluctuate slowly. On the other hand, the levels of HMs in urban dust are significantly impacted by both natural and human disturbances. Road dust is renewed by rainfall, strong winds, and road cleaning (Li et al. 2015). However, indoor dust, especially in areas with limited air flow, can accumulate for extended periods of time and can be different than outdoor dust.

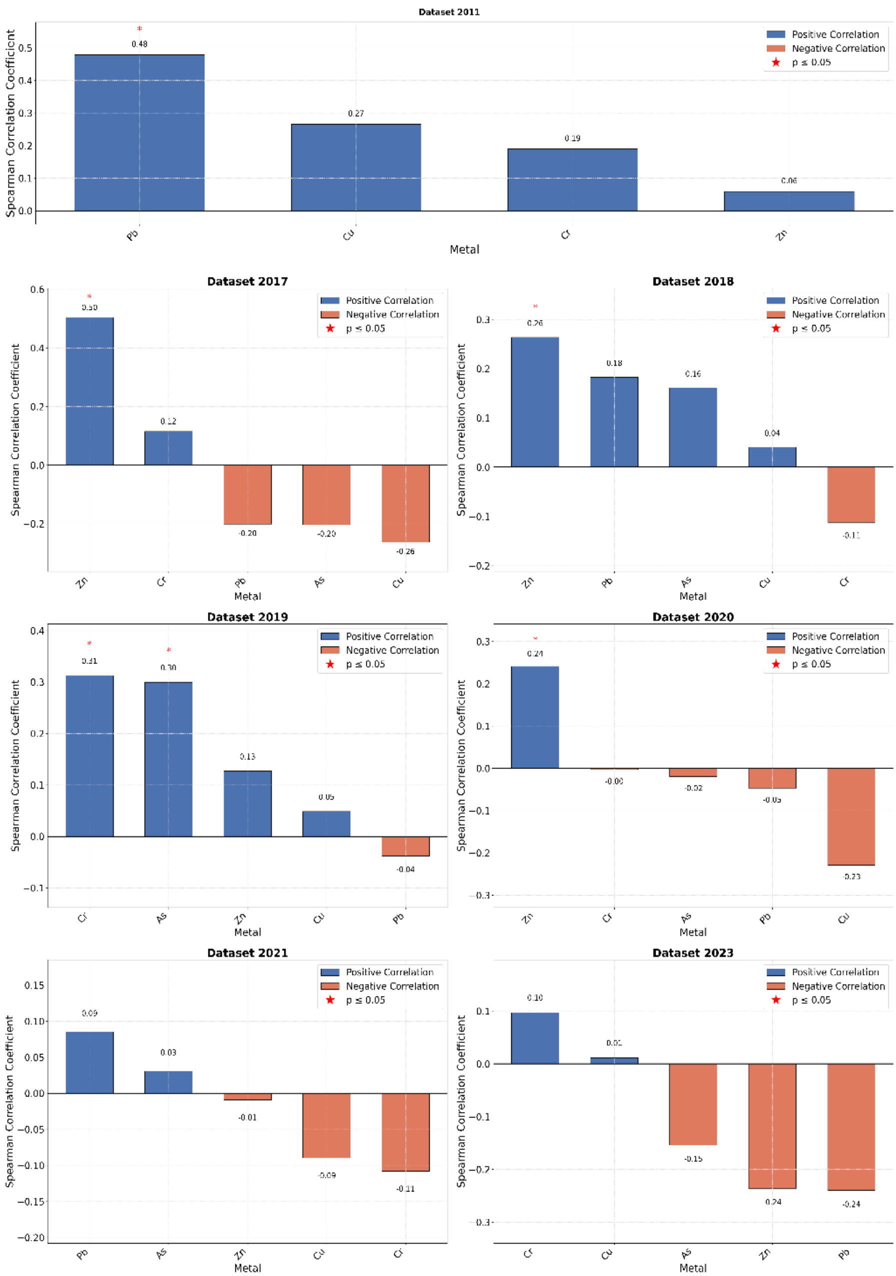


Figure 22. Spearman correlation between dust and soil elements from 2011, 2017, 2018, 2019, 2020, 2021, 2023.

Environmental Context and Contamination Sources. Pb's strong capacity for long-distance air transport contributes to both local and global pollution levels. The mining and smelting industries are primary sources of Pb pollution (Li et al., 2016). Additionally, historical usage of Pb in solvent-based paints, which produce smaller particle sizes and higher concentrations of dangerous metals like As, Cu, Pb, and Zn, contributes to ongoing pollution, particularly compared to water-based paints (Huang et al., 2010). Pb is frequently found in urban soils due to vehicle exhaust, tire wear, bearing wear, and its former use in gasoline.

Meteorological conditions, heavily influenced by the Baltic Sea, play a crucial role in the dispersion of contaminants across Lithuania. This dynamic greatly affects the transmission of industrial pollutants into urban areas, including Vilnius (Nikodemus et al., 1994). The impact of military activities on environmental pollution should also be considered, as studies suggest they contribute significantly to HMs levels in the soil (Vasarevicius et al., 2004).

The high concentrations of Cu detected near bus stops, railroad stations, and a parking lot on a commercial road could be attributed to the long history of Cu usage in brake friction materials since the 1930s (Li et al., 2015). The proximity of schools to these areas, where dust samples were collected, may also have contributed to the elevated Cu levels found in the school dust samples.

Additionally, studies, such as those by Jankauskaite et al. (2008), have illustrated the complex relationship between urban landscapes and topsoil contamination. Infrastructural, and historical areas in Vilnius are more susceptible to pollution, further linking urban development with soil quality.

Although most heavy metals in urban soil and dust originate from similar sources; such as transportation, vehicle emissions, air depositions, power plants burning fossil fuels, infrastructure development or renovation, windstorms, cooking, and even dust carried by shoes, the distribution of these metals in urban soil and indoor dust varies by year and study. This variability highlights the complexity of urban pollution and for each school.

Vilnius and Kaunas, like many urban centers, exhibit high levels of heavy metal contamination due to industrial activities and traffic emissions. The transportation sector, particularly motor vehicles and railways has been a significant source of soil contamination in these areas (Baltrenas and Kliaugiene, 2003; Nikodemus et al., 1994). High levels of heavy metal contamination have also been observed in soil particles from urban areas of Vilnius, especially in high-traffic zones. This contamination, carried by surface runoff sediments, underscores the magnitude of soil pollution in the city (Ignatavicius et al., 2017).

3.3.1 Chapter Conclusion

Elevated Metal Concentrations in Dust:

- Dust samples consistently show significantly higher concentrations of heavy metals than soil samples. For instance, from 2017 to 2023, As levels in dust were significantly higher (p-values: 1.07×10^{-19} to 7.90×10^{-22}).
- Similarly, Cu Zn, Pb, and Cr were elevated in dust compared to soil (e.g., Cu p-values from 6.63×10^{-20} to 9.87×10^{-22}; Zn from 2.99×10^{-19} to 4.25×10^{-25}; Pb from 0.008 to 1.37×10^{-11}; Cr from 7.17×10^{-21} to 3.91×10^{-24}). This indicates that dust serves as a more efficient reservoir for HM, likely due to its prolonged exposure to various pollution sources such as industrial emissions, traffic, and indoor activities.

Soil–Dust Relationships:

- Statistical analysis showed that indoor dust and soil exhibit significantly different HM concentrations, Spearman correlation analysis indicates that, while some metals (notably Zn, and to a lesser extent As and Cr in certain years) exhibit significant positive correlations between soil and dust, most correlations are not statistically significant. This suggests that soil and dust, despite sharing some common sources, accumulate HMs through different mechanisms or at different rates.
- These findings emphasize the need to consider dust as an independent and more hazardous pollutant carrier than soil in school environments.

3.4. Health Risk Assessment of Heavy Metals in Indoor Dust

Hazard Index for health risk. fig. 23 presents the Total Hazard Index (HI) for both adults and children. A value above 1 signifies a non-carcinogenic health risk. Graph A displays the results for adults, with all values falling below 1. Additionally, As and Pb present health risks for children in schools S2 and S14. All other elements have values below 1, indicating no non-carcinogenic risk.

However, Fig. 23 - Graph B, which represents the child HI, shows that for adults on in S2 for Pb cause non-carcinogenic hazard and for children situation is more complicated. Twelve schools have exceeded for As, nine schools for Pb, and one school for Zn and V exceeded 1 which shows that can cause health problems.

Cancerogenic risk on the other hand needs more attention. According to EPA limits, for adults, As, Cr are unacceptable and Pb somewhat in acceptable limits. For children, As and Pb is unacceptable and negligible but for Cr mostly in acceptable and some of them passes threshold which need carefully investigated by the health organizations.

Metal concentrations in various schools were found to vary significantly, suggesting that factors such as proximity to major roads, industrial areas, and the presence of different construction materials could influence metal deposition. Schools located closer to high-traffic areas, for instance, are more likely to be exposed to emissions from vehicles, including Pb and other heavy metals, which settle in dust particles. Additionally, older buildings may have construction materials that contain higher levels of metals like Zn or Cu, which can contribute to indoor dust contamination. Environmental conditions, such as wind patterns and proximity to vegetation, may also play a role in the dispersal and deposition of metals on school grounds. The presence of large amounts of metals, particularly As and Pb, poses significant health risks to children, as reflected in the elevated Hazard Index values. This could be cause because of soil interaction (see Fig. 24 and 25) Therefore, understanding the local environment and building materials is essential in evaluating the sources of metal pollution and their impact on indoor air quality and student health. The presence of large amounts of metals on the school premises raises issues about children's expected safety and health standards.

According to the study findings, many of the schools are located just a few meters away from frequently used roadways and some are near train stations, suggesting that motor vehicle emissions, lubricating oil, grease, and the abrasive wear of rail tracks may contribute to the presence of metals in the school environment. While these factors could likely play a role in metal contamination, it is important to note that this hypothesis has not been directly measured or assessed within this study. To fully understand the sources of metal pollution affecting the schools, future research should focus on collecting data from specific locations, including air quality monitoring and soil analysis, to confirm the contribution of vehicular and rail-related emissions. Furthermore, potential pollution from surrounding restaurants, residential areas, and nearby power plants should also be investigated, as emissions from fuel combustion may further impact the concentrations of metals in indoor dust.

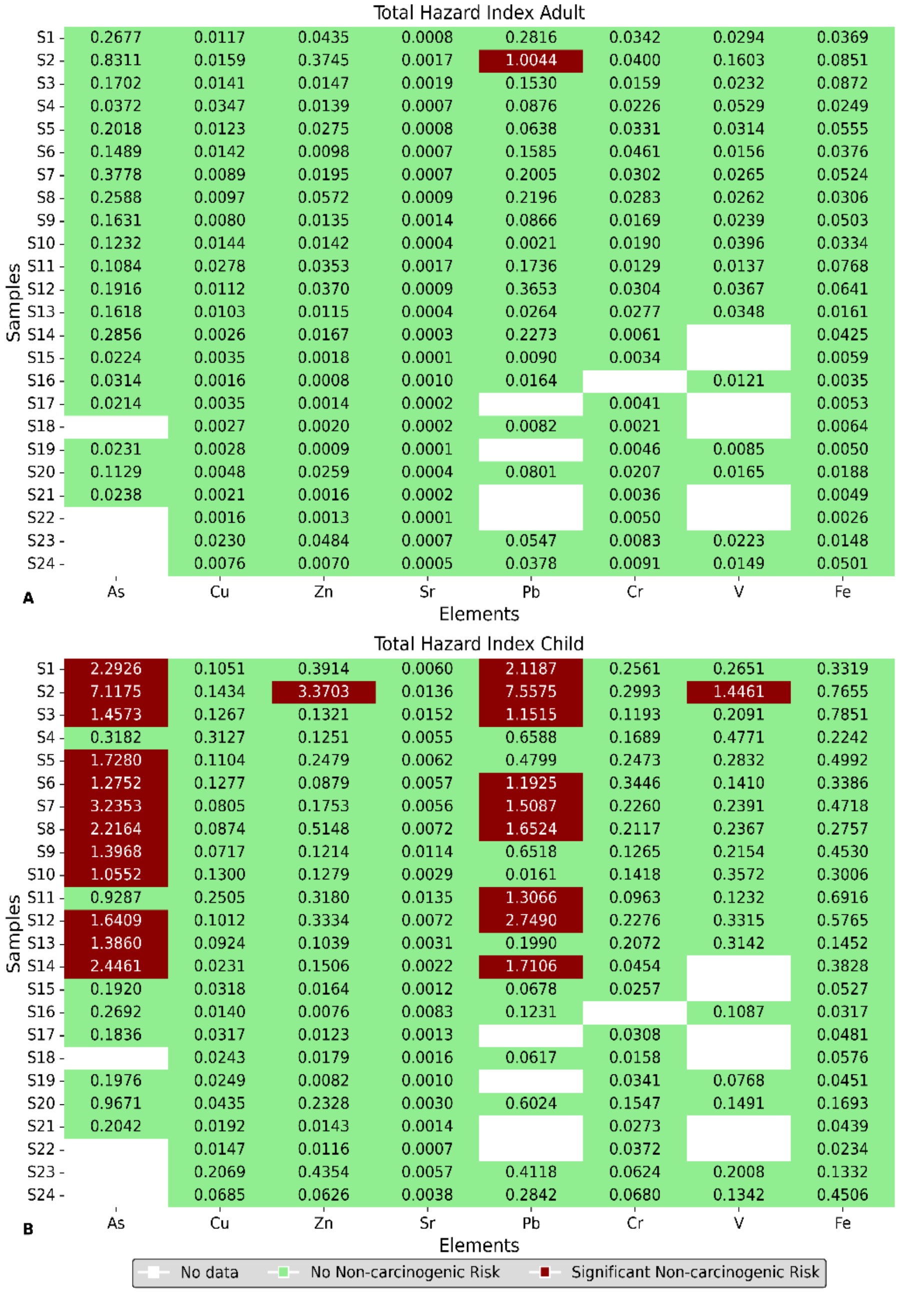


Figure 23. Hazard index values for children and adults in sampled schools for each element.

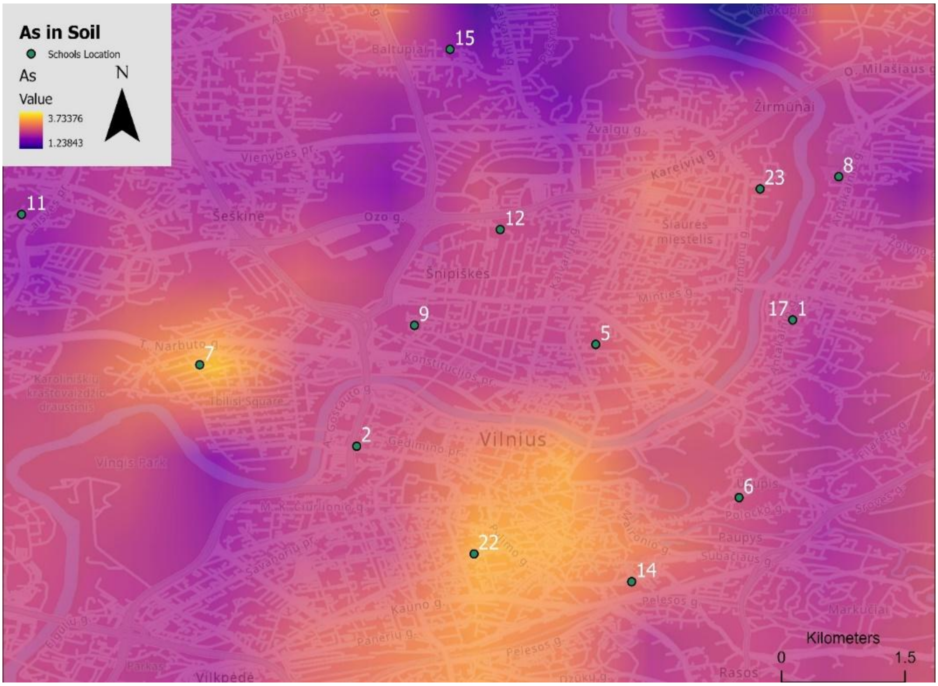


Figure 24. Higher As in soil locations compared to schools' location

Figure 25. Higher Pb in soil locations compared to schools' location

Potential Health Implications. Children are particularly vulnerable to heavy metal toxicity because their immature organ systems make them more susceptible to its harmful effects. Exposure to these pollutants can result in significant health implications, such as intellectual disability, neurocognitive impairments, behavioral abnormalities, respiratory ailments, cancer, and cardiovascular illnesses (Capitão et al. 2022). Moreover, the negative consequences of inorganic pollutants on newborns and children might result in anemia, kidney damage, developmental and reproductive harm, decreased intelligence quotient (IQ), and different harmful effects on the nervous system (Witkowska et al. 2021). Xenobiotic metals have been found to be hazardous and can lead to multiple illnesses such as gastrointestinal, respiratory, cardiovascular, reproductive, renal, and neurological problems. In addition, certain heavy metals can worsen the development of tumors and decrease their responsiveness to treatment (Bair et al. 2022). It is crucial to prioritize the understanding of health risks associated with HM in order to effectively manage the safety of school environments, considering that children spend considerable time outside and have eating habits that may increase their exposure to such metals (Al Osman et al. 2019). The article offers crucial insights into the process of renovating old Soviet-era buildings. It emphasizes the necessity of thoroughly cleaning and removing dust from structures like schools, because children are particularly vulnerable to the harmful effects of pollution. Governments and local authorities should implement routine environmental monitoring in urban schools. This will help in early detection and management of heavy metal contamination, ensuring the safety of children. Schools need to implement systematic dust cleaning routines and contemplate the installation of air filters. Implementing these steps can effectively decrease the level of airborne contaminants in classrooms, therefore protecting the health of children.

3.4.1 Chapter Conclusion

- Figure 20 shows that adults face no non-carcinogenic health risks, as all Hazard Index (HI) values are below 1.
- Children are at significantly higher risk, with 12 schools showing elevated As levels and 9 schools showing high lead (Pb) levels, requiring urgent action to reduce exposure.
- According to EPA guidelines, all adult samples fall into the "Acceptable" or "Negligible" risk categories, with no "Unacceptable" cases observed.

- In contrast, 6 samples for children (2.7%) are classified as "Unacceptable," highlighting their increased vulnerability to heavy metal exposure.

4. General Discussion

This dissertation investigates the dynamics of dust pollution in educational facilities, focusing specifically on the analysis of heavy metals in indoor dust from Vilnius schools and their potential health impacts on children. By employing a range of analytical techniques, this study provides a detailed examination of indoor dust contamination, enhancing our understanding of its implications for child health and contributing significantly to the field of environmental health.

The comprehensive assessment of heavy metal contamination in Vilnius school dust paints a complex environmental picture that integrates multiple lines of evidence. Consistently high concentrations of metals like Zn and Pb, and their variability between schools, emphasize localized pollution pressures, potentially linked to traffic, emissions, and urban infrastructure. Comparative analyses reveal contamination exceeding global benchmarks, reinforcing the significance of site-specific conditions. Pollution metrics such as CF, PLI, EF, and Igeo highlight not only severe contamination at certain "hotspot" schools but also the necessity for targeted remediation strategies. Correlation studies, PCA, and HCA uncover distinct groupings of metals and schools, indicating shared sources and underlying geochemical processes, while PMF analysis attributes observed contamination to a diverse mix of anthropogenic and natural factors. These findings collectively demonstrate that the indoor environment is shaped by the intricate interplay of outdoor pollution, building characteristics, and localized anthropogenic activities. As a result, effective mitigation will require coordinated efforts, informed by comprehensive source identification, to safeguard student health and improve environmental conditions within these educational settings.

The combined evidence from PM2.5/PM10 ratio trends, spatial proximity analyses, and correlation results highlights a multifaceted relationship between outdoor air quality and the composition of indoor dust. Increasing PM2.5/PM10 ratios, indicative of finer and more combustion-related particulate matter, suggest that anthropogenic sources—such as traffic emissions or industrial activities—may facilitate the infiltration of metals into indoor environments. Correlations between PM and indoor dust concentrations of As, Pb, Zn, and (to a lesser extent) Cu further support this link, though Cr's negative association hints at the complexity of source contributions and particle behaviors. Proximity data reveal that schools closer to roads, traffic lights, and other urban infrastructure are potentially more exposed to vehicle-related pollutants, providing context for the observed metal accumulations. Although these relationships are statistically significant, their modest strength underscores that outdoor PM infiltration is just one component of a broader set of factors, encompassing building characteristics, ventilation, indoor sources, and seasonal variations, that jointly shape the indoor dust metal profile. In sum, these findings reinforce the importance of considering both external pollution dynamics and local conditions to better understand and manage indoor air quality in educational settings.

The comprehensive analysis of soil contamination patterns across multiple datasets and years, coupled with comparative examinations of dust samples, provides a nuanced understanding of heavy metal accumulation in urban school environments. Notably, the soil data from various time periods reflect evolving pollution scenarios influenced by industrial, vehicular, and infrastructural sources, resulting in fluctuations in metals like Zn, Pb, Cu, and Cr. The consistent overrepresentation of metals in dust compared to soil underscores the unique capacity of finer particles to adsorb and retain pollutants, effectively serving as both short- and long-term repositories of contaminants. Furthermore, the temporal and spatial variability in soil and dust concentrations, revealed through statistical tests and correlation analyses, highlights the dynamic interplay of anthropogenic activities, environmental policies, and localized conditions. Common pollutant pathways, ranging from traffic emissions and industrial operations to material degradation, converge differently in soil and dust, indicating that indoor dust, with its finer particle size and reduced mobility, can accumulate metals more persistently than soil. This distinction is critical given the heightened vulnerability of schoolchildren to metal exposure. Overall, the integrative approach, incorporating soil and dust assessments, temporal and spatial analyses, as well as advanced statistical and multivariate techniques, demonstrates the complexity of urban environmental contamination.

No specific heavy metal limits for settled indoor dust are proposed here given the dataset's limited scope. Indoor environments differ substantially, in building age, ventilation, occupancy patterns and proximity to emission sources, so a single benchmark is currently impractical. Furthermore, a one-season survey of just 24 schools in Vilnius offers too narrow a data set to support robust guidelines. Establishing meaningful threshold values will demand further research, including multi-season sampling and expanded coverage of diverse building types and regions.

This research emphasizes the necessity of specific policies and interventions in schools to mitigate heavy metal exposure risks and improve indoor air quality, essential for protecting children's health. It also suggests the need for future studies to expand the sample size, investigate temporal pollution variations, conduct extended health monitoring of children exposed to high levels of heavy metals, and include a wider range of educational settings and geographical areas. Examining the specific origins of heavy metals in dust and their entry methods into indoor spaces would provide valuable knowledge for hazard management and reduction.

In summary, this thesis highlights critical environmental health issues in educational environments and underlines the urgent need for improved indoor air quality in schools. The findings lay the groundwork for future research and policy development aimed at creating safer and healthier learning environments for children. The unique contribution of this study lies in its detailed examination of heavy metal concentrations in indoor dust and the combination of health risk assessments with source identification, filling a knowledge gap and contributing significantly to our understanding of environmental health risks in educational facilities.

5. Conclusion

1. Heavy metal concentrations in indoor dust from Vilnius schools varied widely, with Pb levels ranging from 5.3±3.99 mg/kg to 564.25±16.81 mg/kg. Despite this variability, the contamination metrics were uniformly high. all contamination metrics (e.g. enrichment factors and Geoaccumulation Index), Additionally, the mean Pollution Load Index was 1.82 ± 0.82, indicating overall deteriorated environmental quality.
2. PCA, HCA, K-Means clustering and together with correlation analysis revealed distinct elemental clusters in the indoor dust: PC1 contained As, Pb, Cr and Fe. Statistically significant loadings of these elements with PC1, together with their known correlations with vehicular emissions, suggests an anthropogenic influence. PC2 consisted of elements such as Sr and Sc, which may indicate a predominance of natural, lithogenic materials. PC3 was characterized by elements such as Cu, Zn, Zr, Rb and V, which are associated with mixed natural - anthropogenic influences, which are different from those of PC1.
3. The PM2.5/PM10 ratio averaged 0.74, with occasional spikes exceeding 2.7, suggesting substantial indoor infiltration of fine PM. Distance correlation analyses between PM1, PM2.5, and PM10 and indoor dust concentrations of As, Cu, Zn, Pb, and Cr revealed consistently positive coefficients ranging from 0.14 to 0.22 (all $p < 0.05$), indicating weak but statistically significant associations suggest that outdoor emissions, likely originating from traffic and urban activities, are penetrating indoor environments.
4. Long-term monitoring shows that indoor dust accumulates heavy metals more intensely than surrounding soil. Over the years, soil metal levels in the vicinity have fluctuated, but indoor dust consistently holds much higher concentrations of heavy metals such as As, Cu, Zn, Pb, and Cr. Statistical and correlation analysis between soil and dust contamination suggest common pollution sources, even as indoor dust serves as a more concentrated reservoir of these contaminants.

   The health risk assessment using the Total Hazard Index (THI) shows that non-carcinogenic risks differ by element and population. In 50% of the schools, As levels exceed safe limits (THI > 1, up to 7.1), and 37.5% of schools have similar concerns for Pb up to 7.55. Under EPA guidelines, the carcinogenic risk for adults is within acceptable or negligible levels in all schools. For children, most schools fall within safe limits, but 16.7% exceed them, indicating that children are more vulnerable.

Recommendations-Strategies for Reducing Exposures and Mitigating Heavy Metal Concentrations.

- Enhanced Cleaning Protocols involve implementing strict cleaning routines to constantly eliminate dust and particle debris that may accumulate heavy metals. Areas with strong student activity require special care. for policy reforms that mandate the implementation of best practices in cleaning and maintenance within schools to minimize exposure to heavy metals. This could include guidelines for cleaning methods that reduce the resuspension of dust particles and the use of cleaning products that do not contribute to indoor pollution.
- To mitigate the inhalation risks associated with contaminated dust, regulations or guidelines could be developed to mandate the installation and maintenance of high-efficiency particulate air (HEPA) filters in school ventilation systems. Installing high-efficiency filters in school ventilation systems can effectively collect airborne particles and limit the risk of inhaling contaminated dust (Martenies and Batterman, 2018; Lee et al. 2015). These policies could outline specific performance standards for filters based on the local environmental context and the unique needs of educational facilities.
- Green Infrastructure involves using green spaces like gardens and green roofs around school buildings to serve as natural filters for air pollutants, decreasing the infiltration of outdoor pollution into inside areas (Jennings et al. 2021; Bermúdez et al. 2023).
- Developing educational programs to teach students and staff about environmental health risks and preventive activities to promote a culture of safety and awareness.
- Policies promoting collaboration between schools, municipal authorities, environmental agencies, and community organizations can lead to comprehensive approaches to tackle environmental pollution sources. Such policies could establish frameworks for shared responsibility and action, including pollution monitoring, community awareness programs, and the implementation of local pollution control measures.
- Mandatory health and safety audits, including environmental health as-assessments in schools, can identify and manage heavy metal pollution sources. Policies could require regular audits by certified environmental health professionals to assess the levels of heavy metals in school

environments and recommend mitigation measures. These audits could be supported by a central database managed by educational or environmental health authorities to track pollution levels and mitigation efforts over time. Establishing clear guidelines for these assessments, including frequency, methods, and follow-up actions, will be crucial for their success.

- Countries planning school renovations should adopt regulations for the effective removal and management of accumulated dust, leveraging insights from this study to minimize heavy metal exposure risks. Sharing best practices on heavy metal dust mitigation across borders can guide the implementation of safer renovation protocols, ensuring educational environments worldwide are protected from contamination.

These findings emphasize an urgent need for ongoing research, regular monitoring of indoor air quality, and targeted pollution control strategies in educational buildings, particularly in older facilities, to improve indoor environmental quality and safeguard occupant health.

6. References

A. K. Krishna and P. K. Govil, "Heavy metal distribution and contamination in soils of Thane-Belapur industrial development area, Mumbai, Western India," Environmental Geology, vol. 47, no. 8, pp. 1054–1061, 2005.

A. Kashem and B. R. Singh, "Heavy metal contamination of soil and vegetation in the vicinity of industries in Bangladesh," Water, Air, & Soil Pollution, vol. 115, no. 1–4, pp. 347–361, 1999.

A. Soriano, S. Pallares, F. Pardo, A. B. Vicente, T. Sanfeliu, ´ and J. Bech, "Deposition of heavy metals from particulate settleable matter in soils of an industrialised area," Journal of Geochemical Exploration, vol. 113, pp. 36–44, 2012.

Abdi, H., & Williams, L.J. 2010. "Principal component analysis." *WIREs Computational Statistics, 2,* 433-459. DOI: 10.1002/wics.101.

Abdulraheem, M.O., Adeniran, J.A., Ameen, H.A., Odediran, E.T., Yusuf, M.O., & Abdulraheem, K.A. 2022. Source identification and health risk assessments of heavy metals in indoor dusts of Ilorin, North central Nigeria. J. Environ. Health Sci. Eng., 20, 315–330.

Adamiec, E., Jarosz-Krzemińska, E., Wieszała, R. 2016. Heavy metals from non-exhaust vehicle emissions in urban and motorway road dusts. Environ. Monit. Assess, 188, 369. DOI: 10.1007/s10661-016-5377-1

Addo, D.E., Gordon, C., Nyarko, B.J., & Gbadago, J.K. 2012. "Heavy Metal Concentrations in Road Deposited Dust at Ketu-South District, Ghana." DOI: 10.12691/jephh-2-4-1.

Adnan, M., Xiao, B., Xiao, P., Zhao, P., Li, R., & Bibi, S. 2022. "Research Progress on Heavy Metals Pollution in the Soil of Smelting Sites in China." Toxics 10 (5): 231. https://doi.org/10.3390/toxics10050231.

Adnan, M., Xiao, B., Xiao, P., Zhao, P., Li, R., & Bibi, S. 2022. "Research Progress on Heavy Metals Pollution in the Soil of Smelting Sites in China." Toxics 10(5): 231. https://doi.org/10.3390/toxics10050231.

Aguilera, A., Bautista, F., Goguitchaichvili, A., & Garcia-Oliva, F. 2021. "Health risk of heavy metals in street dust." *Frontiers in Bioscience, 26,* 327–345. DOI: 10.2741/4896.

Aguilera, A., Cortés, J.L., Delgado, C., Aguilar, Y., Aguilar, D., Cejudo, R., Quintana, P., Goguitchaichvili, A., & Bautista, F. 2022. "Heavy Metal Contamination (Cu, Pb, Zn, Fe, and Mn) in Urban Dust and its Possible Ecological and Human Health Risk in Mexican Cities." *Frontiers in Environmental Science, 10,* 854460. DOI: 10.3389/fenvs.2022.854460.

Ahmad, S.F., Kumar, P.S., Rozbu, M.R., Chowdhury, A.T., Nuzhat, S., Rafa, N., Mahlia, T.M.I., Ong, H.C., & Mofijur, M. 2022. "Heavy Metal Toxicity, Sources, and Remediation Techniques for Contaminated Water and Soil." Environmental Technology & Innovation 25: 102114. https://doi.org/10.1016/j.eti.2021.102114.

Ahmad, W., Alharthy, R.D., Zubair, M. 2021. "Toxic and Heavy Metals Contamination Assessment in Soil and Water to Evaluate Human Health Risk." Sci Rep 11: 17006. https://doi.org/10.1038/s41598-021-94616-4.

Akbari, Z., Kakuee, O., Shahbazi, R., Khatooni, J. D., & Mashal, M. 2022. "Application of positive matrix factorization and pollutants tracing for identification of dust sources: A case study in Khuzestan, Iran." *Environmental Engineering Research* 27 (6), Article 210365. https://doi.org/10.4491/eer.2021.365.

Akinfieva, T.A., Khamidulina, K.K., Tezikov, E.B., & Ivanov, A.I. 1989. "[Experimental study of the effect of rubidium on the cardiovascular system]." Gigiena truda i professional'nye zabolevaniia, no. 3: 16-20.

Al Osman, M., Yang, F., Massey, I.Y. 2019. Exposure routes and health effects of heavy metals on children. *Biometals, 32,* 563–573. DOI: 10.1007/s10534-019-00193-5

Albretsen, J. 2006. "The toxicity of iron, an essential element." Veterinary Medicine

Alengebawy, A., Abdelkhalek, S.T., Qureshi, S.R., & Wang, M.Q. 2021. "Heavy Metals and Pesticides Toxicity in Agricultural Soil and Plants: Ecological Risks and Human Health Implications." Toxics 9 (3): 42. https://doi.org/10.3390/toxics9030042.

Alengebawy, A., Abdelkhalek, S.T., Qureshi, S.R., & Wang, M.Q. 2021. "Heavy Metals and Pesticides Toxicity in Agricultural Soil and Plants: Ecological Risks and Human Health Implications." Toxics 9(3): 42. https://doi.org/10.3390/toxics9030042.

Ali, S., Ullah, S., Umar, H., Saghir, A., Nasir, S., Aslam, Z., Jabbar, H. M., Aabdeen, Z. ul, & Zain, R. 2022. "Effects of Heavy Metals on Soil Properties and their Biological Remediation." Ind. J. Pure App. Biosci. 10(1): 40-46. http://dx.doi.org/10.18782/2582-2845.8856.

Al-Madanat, O., Jiries, A., Batarseh, M., & Al-Nasir, F. 2017. "Indoor and outdoor pollution with heavy metals in Al-Karak City, Jordan." Journal of International Environmental Application & Science 12 (2): 131-139.

ALZONA, J., COHEN, B.L., RUDOLPH, H., JOW, H.N., & FROHLIGER, J.O. 1979. "Indoor - Outdoor Relationships for Airborne Particulate Matter of Outdoor Origin." *Atmospheric Environment* 13, 55.

Andersson, M., Ottesen, R.T., & Langedal, M. 2010. "Geochemistry of Urban Surface Soils - Monitoring in Trondheim, Norway." Geoderma 156: 112-118. DOI: 10.1016/J.GEODERMA.2010.02.005.

Annesi-Maesano, I., Baiz, N., Banerjee, S., Rudnai, P., Rive, S., & SINPHONIE Group. 2013. "Indoor Air Quality and Sources in Schools and Related Health Effects." Journal of Toxicology and Environmental Health, Part B 16 (8): 491-550. DOI: 10.1080/10937404.2013.853609.

Aplinkos Apsaugos Agentūra. (2025, January 24). Oro taršos ir kokybės švieslentės. Available on: https://aaa.lrv.lt/lt/veiklos-sritys/oras/oro-monitoringas/

Aradhi, K.K., Dasari, B.M., Banothu, D., et al. 2023. "Spatial Distribution, Sources and Health Risk Assessment of Heavy Metals in Topsoil Around Oil and Natural Gas Drilling Sites, Andhra Pradesh, India." Scientific Reports 13: 10614. https://doi.org/10.1038/s41598-023-36580-9.

Awokunmi, E.E., Asaolu, S.S., & Ipinmoroti, K.O. 2010. "Effect of leaching on heavy metals concentration of soil in some dumpsites." African Journal of Environmental Science and Technology 4: 495-499. https://doi.org/10.4314/AJEST.V4I8.71302.

Azimi, P., Zhao, D., & Stephens, B. 2016. "Modeling the impact of residential HVAC filtration on indoor particles of outdoor origin (RP-1691)." *Science and Technology for the Built Environment* 22 (4): 431-462. DOI: 10.1080/23744731.2016.1163239.

B. E. Davies, "Heavy metal contaminated soils in an old industrial area of Wales, Great Britain: source identification through statistical data interpretation," Water, Air, & Soil Pollution, vol. 94, no. 1-2, pp. 85–98, 1997.

Bair, E.C. 2022. A Narrative Review of Toxic Heavy Metal Content of Infant and Toddler Foods and Evaluation of United States Policy. *Front. Nutr., 9,* 919913. DOI: 10.3389/fnut.2022.919913

Ballabio, C., Jiskra, M., Osterwalder, S., Borrelli, P., Montanarella, L., Panagos, P. 2021. "A Spatial Assessment of Mercury Content in the European Union Topsoil." Science of The Total Environment 769: 144755. https://doi.org/10.1016/j.scitotenv.2020.144755.

Baloch, R. M., Maesano, C. N., Christoffersen, J., Banerjee, S., Gabriel, M., Csobod, É., Annesi-Maesano, I., SINPHONIE Study group. 2020. "Indoor air pollution, physical and comfort parameters related to schoolchildren's health: Data from the European SINPHONIE study." Science of The Total Environment 739: 139870. https://doi.org/10.1016/j.scitotenv.2020.139870.

Baltrenas, P., & Kliaugiene, E. 2003. Environmental Impacts on Soils from Transport Systems in Various Cities in Lithuania. *Transactions on the Built Environment.* WIT Press (accessed on 30 December 2023).
Barbieri, M. 2016. "The Importance of Enrichment Factor (EF) and Geoaccumulation Index (Igeo) to Evaluate the Soil Contamination." *Journal of Geology & Geophysics, 5.* DOI: 10.4172/2381-8719.1000237.
Bates, D. V. 1995. "The effects of air pollution on children." Environmental Health Perspectives 103 (suppl 6): 49-53. https://doi.org/10.1289/ehp.95103s6.
Baynes, R. E. (2011). Quantitative Risk Assessment Methods for Cancer and Noncancer Effects. *Progress in Molecular Biology and Translational Science*, *112*, 259-283. https://doi.org/10.1016/B978-0-12-415813-9.00009-X
Beamer, P. I., Elish, C. A., Roe, D. J., Loh, M. M., Layton, D. W. 2012. Differences in Metal Concentration by Particle Size in House Dust and Soil. *J. Environ. Monit., 14*(3), 839. DOI: 10.1039/c2em10740f
Behrooz, R.D., Tashakor, M., Asvad, R., Esmaili-Sari, A., & Kaskaoutis, D.G. 2022. "Characteristics and Health Risk Assessment of Mercury Exposure via Indoor and Outdoor Household Dust in Three Iranian Cities." *Atmosphere, 13,* 583. DOI: 10.3390/atmos13040583.
Belyadi, H., & Haghighat, A. 2021. "Unsupervised Machine Learning: Clustering Algorithms." In *Machine Learning Guide for Oil and Gas Using Python,* edited by Gulf Professional Publishing, 125–168. DOI: 10.1016/b978-0-12-821929-4.00002-0.
Bermúdez, M.d.C.R., Chakraborty, R., Cameron, R.W., Inkson, B.J., Martin, M.V. 2023. A Practical Green Infrastructure Intervention to Mitigate Air Pollution in a UK School Playground. Sustainability, 15, 1075. https://doi.org/10.3390/su15021075.
Bern, C.R., Walton-Day, K., & Naftz, D.L. 2019. "Improved enrichment factor calculations through principal component analysis: Examples from soils near breccia pipe uranium mines, Arizona, USA." *Environmental Pollution, 248,* 90–100. DOI: 10.1016/j.envpol.2019.01.122.
Bharti, R., & Sharma, R. 2022. "Effect of heavy metals: An overview." Materials Today: Proceedings 51(Part 1): 880-885. https://doi.org/10.1016/j.matpr.2021.06.278
Bi, Y. 2015. "General situation of toxicity study on zirconium and its compounds." Chinese Journal of Industrial Medicine.
Binner, H., Sullivan, T., Jansen, M.A.K., & McNamara, M.E. 2023. "Metals in Urban Soils of Europe: A Systematic Review." The Science of the Total Environment 854: 158734. https://doi.org/10.1016/j.scitotenv.2022.158734.

Blondet, I., Schreck, E., Viers, J., Casas, S., Jubany, I., Bahí, N., ... Darrozes, J. 2019. "Atmospheric dust characterisation in the mining district of Cartagena-La Unión, Spain: Air quality and health risks assessment." Science of The Total Environment 693: 133496. https://doi.org/10.1016/j.scitotenv.2019.07.302.

Boguszewska, A., & Pasternak, K. 2004. "Mercury-influence on biochemical processes of the human organism." Annales Universitatis Mariae Curie-Sklodowska. Sectio D: Medicina 59 (2): 524-7.

Boruah H, Tyagi N, Gupta SK, Chabukdhara M and Malik T. 2023. Understanding the adsorption of iron oxide nanomaterials in magnetite and bimetallic form for the removal of arsenic from water. Front. Environ. Sci. 11:1104320. doi: 10.3389/fenvs.2023.1104320

Bradford, G. R., Chang, A. C., Page, A. L., Bakhtar, D., Frampton, J. A., Wright, H. 1996. *Background concentrations of trace and major elements in California soils* (Kearey Foundation Special Report). Kearey Foundation of Soil Science, Division of Agriculture and Natural Resources, University of California. Retrieved from http://www.envisci.ucr.edu/downloads/chang/kearney/kearneytext.html.

Briffa, J., Sinagra, E., & Blundell, R. 2020. "Heavy metal pollution in the environment and their toxicological effects on humans." Heliyon 6 (10): e04691. https://doi.org/10.1016/j.heliyon.2020.e04691.

Bucur, E., & Danet, A. 2019. "Indoor/outdoor correlations regarding indoor air pollution with particulate matter." Environmental Engineering and Management Journal 18 (2): 425-432.

Burlakovs, J., & Vircavs, M. 2012. "Heavy Metal Remediation Technologies in Latvia: Possible Applications and Preliminary Case Study Results." DOI: 10.2478/v10216-011-0038-3.

Businelli, D., Massaccesi, L., Said-Pullicino, D., & Gigliotti, G. 2009. "Long-term Distribution, Mobility and Plant Availability of Compost-derived Heavy Metals in a Landfill Covering Soil." Science of The Total Environment 407(4): 1426-1435. https://doi.org/10.1016/j.scitotenv.2008.10.052.

Buivydaite, Vanda. 2005. Soil Survey and available Soil Data in Lithuania. ESB-RR9. https://esdac.jrc.ec.europa.eu/ESDB_Archive/eusoils_docs/esb_rr/n09_soilresources_of_europe/Lithuania.pdf

Candeias, C., Vicente, E., Tomé, M., Rocha, F., Ávila, P., & Célia, A. 2020. "Geochemical, Mineralogical and Morphological Characterisation of Road Dust and Associated Health Risks." Int. J. Environ. Res. Public Health 17: 1563. https://doi.org/10.3390/ijerph17051563.

Cao, Z. G., Yu, G., Chen, Y. S., Cao, Q. M., Fiedler, H., Deng, S. B., Huang, J., Wang, B. 2012. Particle Size: A Missing Factor in Risk Assessment of Human Exposure to Toxic Chemicals in Settled Indoor Dust. *Environ. Int., 49,* 24–30. DOI: 10.1016/j.envint.2012.08.010

Capitão, C., Martins, R., Santos, O., Bicho, M., Szigeti, T., Katsonouri, A., Bocca, B., Ruggieri, F., Wasowicz, W., Tolonen, H. 2022. Exposure to heavy metals and red blood cell parameters in children: A systematic review of observational studies. *Front. Pediatr., 10,* 921239. DOI: 10.3389/fped.2022.921239

Carlsen, H. K. 2014. "Health Effects of Air Pollution in Iceland." Department of Public Health and Clinical Medicine, Occupational and Environmental Medicine, Umeå University. Retrieved from http://umu.diva-portal.org/ (Electronic version). ISBN: 978-91-7601-082-2. ISSN: 0346-6612. (New Series No 1659).

Chang, S. H., Wang, K. S., Chang, H. F., Ni, W. W., Wu, B. J., Wong, R. H., Lee, H. S. 2009. Comparison of source identification of metals in road-dust and soil. *Soil and Sediment Contamination: An International Journal, 18*(5), 669–683. https://doi.org/10.1080/15320380903085691.

Charlesworth, S., De Miguel, E. & Ordóñez, A. 2011. A review of the distribution of particulate trace elements in urban terrestrial environments and its application to considerations of risk. Environ Geochem Health 33, 103–123. https://doi.org/10.1007/s10653-010-9325-7

Chasapis, C.T., Ntoupa, PS.A., Spiliopoulou, C.A. et al. 2020. "Recent aspects of the effects of zinc on human health." Arch Toxicol 94: 1443–1460. https://doi.org/10.1007/s00204-020-02702-9

Chavent, M., Guégan, H., Kuentz, V., Patouille, B., & Saracco, J. 2008. "PCA- and PMF-based methodology for air pollution sources identification and apportionment." *Environmetrics, 20,* 928–942. DOI: 10.1002/env.963.

Chen, H., Lu, X., Chang, Y., Xue, W. Heavy metal contamination in dust from kindergartens and elementary schools in Xi'an, China. *Environ. Earth Sci.* 2013, 71, 2701–2709. DOI: 10.1007/s12665-013-2648-9

Chen, H., Lu, X., Gao, T., & Chang, Y. 2016. "Identifying Hot-Spots of Metal Contamination in Campus Dust of Xi'an, China." *International Journal of Environmental Research and Public Health, 13,* 555. DOI: 10.3390/ijerph13060555.

Choo, C., & Jalaludin, J. 2015. "An overview of indoor air quality and its impact on respiratory health among Malaysian school-aged children." Reviews on Environmental Health 30 (1): 9-18. https://doi.org/10.1515/reveh-2014-0065.

Chu, H., Liu, Y., Xu, N. 2023. Concentration, sources, influencing factors and hazards of heavy metals in indoor and outdoor dust: A review. Environ Chem Lett 21, 1203–1230. https://doi.org/10.1007/s10311-022-01546-2

Codling, E.E., Chaney, R.R., & Mulchi, C.L. 2008. "Effects of Broiler Litter Management Practices on Phosphorus, Copper, Zinc, Manganese, and Arsenic Concentrations in Maryland Coastal Plain Soils." Commun. Soil Sci. Plant Anal. 39: 1193–1205.

Collett, J.L. 1993. Atmospheric Deposition Processes. In: Schulin, R., Desaules, A., Webster, R., von Steiger, B. (eds) Soil Monitoring. Monte Verità. Birkhäuser, Basel. https://doi.org/10.1007/978-3-0348-7542-4_8

Comero, S., Gawlik, B., & Capitani, L. 2009. *Positive Matrix Factorisation (PMF) : an introduction to the chemometric evaluation of environmental monitoring data using PMF.* Publications Office. https://data.europa.eu/doi/10.2788/2497.

Custodio, M., Espinoza, C., Orellana, E., Chanamé, F., Fow, A., & Peñaloza, R. 2022. "Assessment of toxic metal contamination, distribution and risk in the sediments from lagoons used for fish farming in the central region of Peru." *Toxicology Reports, 9,* 1603–1613. DOI: 10.1016/j.toxrep.2022.07.016.

D. Voutsa, A. Grimanis, and C. Samara, "Trace elements in vegetables grown in an industrial area in relation to soil and air particulate matter," Environmental Pollution, vol. 94, no. 3, pp. 325–335, 1996.

Dagys, J. 1968 "Vilniaus miesto pramonė 1938–1940 metais", *Ekonomika*, 8(3), pp. 5–16. doi:10.15388/Ekon.1968.14437.

Daultrey, S. 1976. *Principal Components Analysis.* Geo Abstracts Ltd., Norwich.

David, W.L., & Paloma, I.B. 2009. Migration of Contaminated Soil and Airborne Particulates to Indoor Dust. Environ. Sci. Technol, 43, 8199–8205.

Davis, J. J., & Gulson, B. L. 2005. "Ceiling (attic) dust: A 'museum' of contamination and potential hazard." Environmental Research 99 (2): 177-194. https://doi.org/10.1016/j.envres.2004.10.011.

DGE Baltic Soil. 2022. "Vilniaus Miesto Viešųjų, Socialiai ir Taršai Jautrių, Potencialiai Užterštų Bei Praeities Taršos Šaltinių Teritorijų Dirvožemio (Ar Grunto) Monitoringo Rezultatai 2017–2021 Metų Laikotarpiu." Retrieved from https://aplinka.vilnius.lt/wp-content/uploads/2022/08/2017-2021-Dirvozemio-ataskaita-VVA.pdf (accessed on December 6, 2023).

DGE Baltic Soil. 2023. "Vilniaus Miesto Dirvožemio Ir Grunto Monitoringo Rezultatai 2023 Metais." Retrieved from https://aplinka.vilnius.lt/wp-

content/uploads/2023/10/vilniaus-dirvozemio-monitroringas-2023.pdf (accessed on December 6, 2023).
Ding, Q., Cheng, G., Wang, Y., & Zhuang, D. 2017. "Effects of natural factors on the spatial distribution of heavy metals in soils surrounding mining regions." Science of The Total Environment 578: 577-585. https://doi.org/10.1016/j.scitotenv.2016.11.001.
Dingle, J. H., Kohl, L., Khan, N., Meng, M., Shi, Y. A., Pedroza-Brambila, M., ... Chan, A. W. H. 2021. "Sources and composition of metals in indoor house dust in a mid-size Canadian city." Environmental Pollution 289: 117867. https://doi.org/10.1016/j.envpol.2021.117867.
Dotaniya, M.L., Dotaniya, C.K., Solanki, P., Meena, V.D., & Doutaniya, R.K. 2020. "Lead Contamination and Its Dynamics in Soil–Plant System." In D.K. Gupta, S. Chatterjee, & C. Walther (Eds.), Lead in Plants and the Environment, pp. 83–98. Springer, Cham.
Doyi, I.N.Y., Isley, C.F., Soltani, N.S., & Taylor, M.P. 2019. "Human exposure and risk associated with trace element concentrations in indoor dust from Australian homes." *Environmental International, 133,* 105125. DOI: 10.1016/j.envint.2019.105125.
Dundulis, K. 2006. "Sand Soils of Lithuanian Coastal Area and Their Geotechnical Properties." Geology, Environmental Science, Engineering.
Duruibe, J.O., Ogwuegbu, M.O.C., & Egwurugwu, J.N. 2007. "Heavy metal pollution and human biotoxic effects." International Journal of Physical Sciences 2 (5): 112-118. Available online at http://www.academicjournals.org/IJPS.
E. I. B. Chopin and B. J. Alloway, “Trace element partitioning and soil particle characterisation around mining and smelting areas at Tharsis, R´ıotinto and Huelva, SW Spain,” Science of the Total Environment, vol. 373, no. 2-3, pp. 488–500, 2007.
E. Martley, B. L. Gulson, and H. R. Pfeifer, “Metal concentrations in soils around the copper smelter and surrounding industrial complex of Port Kembla, NSW, Australia,” Science of the Total Environment, vol. 325, no. 1–3, pp. 113–127, 2004.
Eeftens, M., Tsai, M.-Y., Ampe, C. Anwander, B., Beelen, R., Bellander, T., Cesaroni, G., Cirach, M., Cyrys, J., De Hoogh, K. 2012. Spatial variation of PM2.5, PM10, PM2.5 absorbance and PMcoarse concentrations between and within 20 European study areas and the relationship with NO2 – Results of the ESCAPE project. Atmos. Environ, 62, 303–317. https://doi.org/10.1016/j.atmosenv.2012.08.038.

El-Amier, Y.A., Alghanem, S.M., & El-Alfy, M.A. 2018. "Ecological Risk Assessment of Heavy Metal Pollution in Topsoil of Mediterranean Coast: A Case Study of Mareotis Coast, Egypt."
European Commission. 2023a. "A New Tool Maps the State of Soil Health Across Europe." Available on: EU Science Hub.
European Commission. 2023b. "Soil Health." Available on: EU Environment.
European Commission. 2023c. "Soil and Land." Available on: EU Environment.
European Commission. 2023d. "Soil Monitoring Law." Available on: EU Environment.
Fageria, N.K. 2009. The Use of Nutrients in Crop Plants. Boca Raton, FL, USA: Experimental Agriculture - EXP AGR. https://doi.org/10.1017/S0014479709007789.
Faisal, M., Wu, Z., Wang, H., Hussain, Z., & Shen, C. 2021. "Geochemical Mapping, Risk Assessment, and Source Identification of Heavy Metals in Road Dust Using Positive Matrix Factorization (PMF)." *Atmosphere* 12: 614. https://doi.org/10.3390/atmos12050614.
Fakhri, Y., Akhlaghi, M., Daraei, H., 2023. The concentration of potentially toxic elements (zinc, iron, manganese) bound to $PM_{2.5}$ in the indoor air of urban schools: A global systematic review and meta-analysis. *Air Quality, Atmosphere and Health, 16*, 77–84. https://doi.org/10.1007/s11869-022-01257-1.
Fan, H., Zhao, C., Yang, Y., Yang, X. 2021. Spatio-Temporal Variations of the PM2.5/PM10 Ratios and Its Application to Air Pollution Type Classification in China. Front. Environ. Sci, 9, 692440. https://doi.org/10.3389/fenvs.2021.692440.
Fiala, M., Hwang, H. M. 2021. Influence of highway pavement on metals in road dust: A case study in Houston, Texas. *Water, Air, & Soil Pollution, 232*, 185. https://doi.org/10.1007/s11270-021-05139-7.
Field, R.A., Phillips, J.L., Goldstone, M.E., Lester, J.N., & Perry, R. 1992. "Indoor/outdoor interactions during an air pollution event in Central London." Environmental Technology 13 (4): 391-408. DOI: 10.1080/09593339209385167.
Fsadni, P., Bezzina, F., Fsadni, C., & Montefort, S. 2018. "Impact of School Air Quality on Children's Respiratory Health." Indian journal of occupational and environmental medicine 22 (3): 156–162. https://doi.org/10.4103/ijoem.IJOEM_95_18.
G. E. Voglar and D. Lestan, "Solidification/stabilisation of metals contaminated industrial soil from former Zn smelter in Celje, Slovenia, using

cement as a hydraulic binder," Journal of Hazardous Materials, vol. 178, no. 1–3, pp. 926–933, 2010.
Garas, S. K., Triantafyllou, A. G., Tolis, E. I., Diamantopoulos, Ch. N., & Bartzis, J. G. 2020. "Positive matrix factorization on elemental concentrations of PM10 samples collected in areas within proximal and far from mining and power station operations in Greece." *Global NEST Journal* 22 (1): 132-142. https://doi.org/10.30955/gnj.003128.
Gareev, K.G. 2023. Diversity of Iron Oxides: Mechanisms of Formation, Physical Properties and Applications. Magnetochemistry, 9, 119. https://doi.org/10.3390/magnetochemistry9050119
Ghosh, A., Dhiman, S., Gupta, A., Jain, R. 2022. Process Evaluation of Scandium Production and Its Environmental Impact. Environments, 10, 8. https://doi.org/10.3390/environments10010008.
Ghosh, S., Sharma, A., Talukder, G. 1992. Zirconium. Biol. Trace Element Res, 35, 247–271. https://doi.org/10.1007/bf02783770.
Gianoncelli, A., & Kourousias, G. 2007. "Limitations of portable XRF implementations in evaluating depth information: An archaeometric perspective." *Applied Physics A, 89,* 857–863. DOI: 10.1007/s00339-007-4221-4.
Glaspole, S.E. 2019. "The rise and fall of strontium as a therapeutic agent and the importance of of pharmacovigilance." The School Science Review 101: 40-42.
Global Wind Atlas. Global Wind Atlas. Available online: https://globalwindatlas.info (accessed on 12 April 2023).
Gope, M., Masto, R.E., George, J., Hoque, R.R., & Balachandran, S. 2017. "Bioavailability and health risk of some potentially toxic elements (Cd, Cu, Pb and Zn) in street dust of Asansol, India." *Ecotoxicology and Environmental Safety, 138,* 231–241. DOI: 10.1016/j.ecoenv.2017.01.008.
Górny, R.L., Jędrzejczak, A., & Pastuszka, J. 1995. "[Dust particles and metals in outdoor and indoor air of Upper Silesia]." Roczniki Panstwowego Zakladu Higieny 46 (2): 151-61.
Goudarzi, G., Alavi, N., Geravandi, S., et al. 2019. "RETRACTED ARTICLE: Ambient particulate matter concentration levels of Ahvaz, Iran, in 2017." Environ Geochem Health 41: 841–849. https://doi.org/10.1007/s10653-018-0182-0.
Goudie, A. S. 2014. "Desert dust and human health disorders." Environment International 63: 101-113. https://doi.org/10.1016/j.envint.2013.10.011.

Goudie, A.S. 2020. "Dust Storms and Human Health." In Extreme Weather Events and Human Health. Springer, Cham. https://doi.org/10.1007/978-3-030-23773-8_2.

Gregorauskienė V. & Kadūnas, V. 2006. Vertical distribution patterns of trace and major elements within soil profile in Lithuania. Geological Quarterly, 2006, 50 (2): 229–237.

Gregorauskienė, V. 2012. *Regularities of vertical element distribution within the soil profile in Lithuania* (Doctoral dissertation, Vilnius University). Vilnius University, Nature Research Centre, Institute of Geology and Geography.

Gregorauskienė, V., & Kadūnas, V. 2010. "Vertical Distribution Patterns of Trace and Major Elements within Soil Profile in Lithuania." Geological Quarterly 50: 229-237.

Grigalavičienė, I., Rutkovienė, V., & Marozas, V. (n.d.). "The Accumulation of Heavy Metals Pb, Cu and Cd at Roadside Forest Soil." Polish J. Environ. Stud. 14.

Gugamsetty, B., Wei, H., Liu, C.-N., Awasthi, A., Hsu, S.-C., Tsai, C.-J., Roam, G.-D., Wu, Y.-C., & Chen, C.-F. 2012. "Source characterization and apportionment of PM10, PM2.5, and PM0.1 by using positive matrix factorization." *Aerosol and Air Quality Research* 12: 476-491. https://doi.org/10.4209/aaqr.2012.04.0084.

Gunawardana, C., Egodawatta, P., Goonetilleke, A. 2014. Role of Particle Size and Composition in Metal Adsorption by Solids Deposited on Urban Road Surfaces. *Environ. Pollut., 184,* 44–53. DOI: 10.1016/j.envpol.2013.08.010

Gunawardana, C., Goonetilleke, A,, Egodawatta, P., Dawes, L., Kokot, S. 2012

Gupta, S. 2018. Roles of metals in human health. https://doi.org/10.15406/MOJBOC.2018.02.00085

Hartwig, A., & Commission, M. 2021. "Zirconium and its compounds (except zirconium dioxide) MAK Value Documentation , supplement – Translation of the German version from 2019."

Haug, L.H. 2019. "An Indoor/Outdoor Air Quality Relationship Analysis Using Internet of Things." Environmental Science, Engineering, Computer Science.

He, Y., Peng, C., Zhang, Y., Guo, Z., Xiao, X., Kong, L. 2022. Comparison of Heavy Metals in Urban Soil and Dust in Cities of China: Characteristics and Health Risks. *Int. J. Environ. Sci. Technol.* DOI: 10.1007/s13762-022-04051-9

DENNIS R. HELSEL (USGS). John Wiley and Sons, New York. 2004. Hardcover, 268 pp. ISBN 0–471–67173–8. https://doi.org/10.2136/vzj2005.0106br

Helmisaari, H., Derome, J.R., Fritze, H., Nieminen, T.M., Palmgren, K., Salemaa, M., & Vanha-Majamaa, I. 1995. "Copper in Scots Pine Forests around a Heavy-Metal Smelter in South-Western Finland." Water, Air, and Soil Pollution 85: 1727-1732. DOI: 10.1007/BF00477229.

Hichem, N., Hadjer, Z., Fateh, S., Feriel, L., & Wang, Z. . 2022. "The potential exposure and hazards of zirconia nanoparticles: A review." Ecotoxicology and Environmental Contamination 17(1): 1–21. https://doi.org/10.5132/eec.2022.01.01

Hjortenkrans, D.S., Bergbäck, B.G., & Häggerud, A.V. 2006. "New Metal Emission Patterns in Road Traffic Environments." Environmental Monitoring and Assessment 117: 85-98. DOI: 10.1007/S10661-006-7706-2.

Höllriegl, V., and H. Z. München. 2011. "Strontium in the Environment and Possible Human Health Effects." In Encyclopedia of Environmental Health, 268-275. https://doi.org/10.1016/B978-0-444-52272-6.00638-3.

Hu, W.Y., Wang, H.F., Dong, L.R., Huang, B., Ole, K.B., Hans, C.B.H., He, Y., & Peter, E.H. 2018. "Source Identification of Heavy Metals in Periurban Agricultural Soils of Southeast China: An Integrated Approach." Environ. Pollut. 237: 650–661.

Huang, S. L., Yin, C. Y., Yap, S. Y. 2010. Particle Size and Metals Concentrations of Dust from a Paint Manufacturing Plant. *J. Hazard. Mater., 174*(1–3), 839–842. DOI: 10.1016/j.jhazmat.2009.09.129

Ignatavičius, G., Unsal, M.H., Busher, P.E., Wołkowicz, S., Satkūnas, J., Šulijienė, G., & Valskys, V. 2022. Geochemistry of Mercury in Soils and Water Sediments. AIMS Environ. Sci., 9, 261–281.

Ignatavicius, G., Valskys, V., Bulskaya, I.V., Paliulis, D., Zigmontienė, A., & Satkūnas, J. 2017. "Heavy Metal Contamination in Surface Runoff Sediments of the Urban Area of Vilnius, Lithuania." Estonian Journal of Earth Sciences 66: 13. DOI: 10.3176/EARTH.2017.04.

J. Bech, C. Poschenrieder, M. Llugany et al., "Arsenic and heavy metal contamination of soil and vegetation around a copper mine in Northern Peru," Science of the Total Environment, vol. 203, no. 1, pp. 83–91, 1997.

Jakimavičius, M. 2011. *Traffic conditions analysis based on an advanced traveller information system of Vilnius City*. Vilnius Gediminas Technical University. 7-5 (3 Volumes). ISSN 2029-7106 print / ISSN 2029-7092 online. http://enviro.vgtu.lt

Jang, J.H., Kun, L., & Jo, Y.M. 2009. "Fine dust control by HVAC in Seoul metro subway." *2009 ICCAS-SICE*, 1703-1706.
Jankaitė, A., Baltrėnas, P., & Kazlauskienė, A. 2008. "Heavy Metal Concentrations in Roadside Soils of Lithuania's Highways." Geologija 50(4): 237–245. https://doi.org/10.2478/v10056-008-0049-7.
Jankauskaitė, M., Taraškevičius, R., Zinkutė, R., & Veteikis, D. 2008. "Relationship between Landscape Self-regulation Potential and Topsoil Additive Contamination by Trace Elements in Vilnius City." Journal of Environmental Engineering and Landscape Management 16(1): 5-15. https://doi.org/10.3846/1648-6897.2008.16.5-14.
Jankauskaitė, M., Taraškevičius, R., Zinkutė, R., Veteikis, D. 2008. Relationship between landscape self-regulation potential and topsoil additive contamination by trace elements in Vilnius city. *J. Environ. Eng. Landsc. Manag., 16*, 5–15. DOI: 10.3846/1648-6897.2008.16.5-14
Järup L. 2003. "Hazards of heavy metal contamination." British medical bulletin 68: 167–182. https://doi.org/10.1093/bmb/ldg032
Jennings, V., Reid, C.E., Fuller, C.H. 2021. Green infrastructure can limit but not solve air pollution injustice. Nat. Commun, 12, 4681. https://doi.org/10.1038/s41467-021-24892-1.
Jha, K., Nandan, A., Siddiqui, N.A., Mondal, P. 2020. Sources of Heavy Metal in Indoor Air Quality. In Advances in Air Pollution Profiling and Control: Select Proceedings of HSFEA 2018; Springer Transactions in Civil and Environmental Engineering; Springer: Singapore; pp. 203–210. DOI: 10.1007/978-981-15-0954-4_13
Jung, J., Park, D., Kwon, S.-B., Choi, S., Kim, S., Lee, W., & Jeon, H. 2014. "A study on the characterization of deposited dust on HVAC ducts in subway stations." International Journal of Environmental Monitoring and Analysis 2 (6): 320-327. doi: 10.11648/j.ijema.20140206.14.
K. K. Deepali and K. Gangwar, "Metals concentration in textile and tannery effluents, associated soils and ground water," New York Science Journal, vol. 3, no. 4, pp. 82–89, 2010.
Kabata-Pendias, A. (2011). Trace elements in soils and plants (4th ed.). CRC Press. https://doi.org/10.1201/b10158
Kabir, E., Ray, S., Kim, K.-H., Yoon, H.-O., Jeon, E.-C., Kim, Y.S., Cho, Y.-S., Yun, S.-T., & Brown, R.J.C. 2012. "Current Status of Trace Metal Pollution in Soils Affected by Industrial Activities." The Scientific World Journal 2012: 916705. https://doi.org/10.1100/2012/916705.
Kabir, E., Ray, S., Kim, K.-H., Yoon, H.-O., Jeon, E.-C., Kim, Y.S., Cho, Y.-S., Yun, S.-T., & Brown, R.J.C. 2012. "Current Status of Trace Metal Pollution

in Soils Affected by Industrial Activities." The Scientific World Journal 2012: Article ID 916705. https://doi.org/10.1100/2012/916705.
Kadachi, A.N., & Al-Eshaikh, M.A. 2012. "Limits of detection in XRF spectroscopy." *X-Ray Spectrometry, 41,* 350–354. DOI: 10.1002/xrs.2412.
Kadūnas, V., Budavičius, R., Gregorauskienė, V., Katinas, V., Kliaugienė, E., Radzevičius, A., & Satkūnas, J. (Eds.). 1999. Lietuvos Geocheminis Atlasas = Geochemical Atlas of Lithuania. Vilnius: Lietuvos geologijos tarnyba, Geologijos institutas. ISBN 9986-623-26-X.
Kadūnas, V., Budavičius, R., Gregorauskienė, V., Katinas, V., Kliaugienė, E., Radzevičius, A., & Taraškevičius, R. 1999. *Lietuvos Geocheminis Atlasas - Geochemical Atlas of Lithuania.* Publisher: Geologijos institutas, Vilnius, Lithuania.
Kadūnas, V., Katinas, V., Radzevičius, A., & Zinkutė, R. 2005. "Geochemical Atlas of Lithuania – A Basis for Detailed Geochemical Investigations of Long-Term Ecological Research Sites." Acta Zoologica Lituanica 15(2): 131-135. https://doi.org/10.1080/13921657.2005.10512389.
Kalua, A., Jo, S. J., Fateminasab, S., Al-Rqaibat, S., & Opitz, C. 2020. "Impact of ventilation method on residential indoor PM dispersion during dust storm events in Saudi Arabia." Architectural Engineering and Design Management 16 (3): 191-208. DOI: 10.1080/17452007.2019.1652797.
Kamunda, C., Mathuthu, M., & Madhuku, M. 2016. Health Risk Assessment of Heavy Metals in Soils from Witwatersrand Gold Mining Basin, South Africa. International Journal of Environmental Research and Public Health. 13. 663. 10.3390/ijerph13070663.
Kaonga, C. C., Kosamu, I. B. M., Utembe, W. R. 2021. A review of metal levels in urban dust, their methods of determination, and risk assessment. *Atmosphere, 12*(7), 891. https://doi.org/10.3390/atmos12070891.
Karanasiou, A., Moreno, N., Moreno, T., Viana, M., de Leeuw, F., & Querol, X. 2012. "Health effects from Sahara dust episodes in Europe: Literature review and research gaps." Environment International 47: 107-114. https://doi.org/10.1016/j.envint.2012.06.012.
Karanasiou, A., Moreno, T., Amato, F., Lumbreras, J., Narros, A., Borge, R., Tobías, A., Boldo, E., Linares, C., Pey, J., Reche, C., Alastuey, A., & Querol, X. 2011. "Road dust contribution to PM levels – Evaluation of the effectiveness of street washing activities by means of Positive Matrix Factorization." *Atmospheric Environment* 45: 2193-2201. https://doi.org/10.1016/j.atmosenv.2011.01.067.

Kashulina, G.M. 2017. "Extreme Pollution of Soils by Emissions of the Copper–Nickel Industrial Complex in the Kola Peninsula." Eurasian Soil Science 50: 837-849. DOI: 10.1134/S1064229317070031.
Kashulina, G.M. 2018. "Monitoring of Soil Contamination by Heavy Metals in the Impact Zone of Copper-Nickel Smelter on the Kola Peninsula." Eurasian Soil Science 51: 467-478. DOI: 10.1134/S1064229318040063.
Khalifa, M., Elsaadani, Z., Qi, W.M. 2022. Spatial Distribution and Extent of Environmental Risks of Strontium Metal Concentration in Water and Bottom Sediments of Nile River, Egypt. Available online: https://www.researchsquare.com/article/rs-1337370/v1 (accessed on 20 September 2023).
Khatoon-Abadi, A., Sheikh Hoseini, A. R., & Khalili, B. 2008. "Effect of mercury on human health and the environment: an overview." International Journal of Food Safety, Nutrition and Public Health (IJFSNPH) 1 (1). Retrieved from https://www.inderscience.com/offers.php?id=18854.
Kim, H. S., Kim, Y. J., & Seo, Y. R. 2015. "An Overview of Carcinogenic Heavy Metal: Molecular Toxicity Mechanism and Prevention." Institute of Environmental Medicine for Green Chemistry, Department of Life Science, Dongguk University Biomedical Campus, Goyang, Korea. Retrieved from: http://dx.doi.org/10.15430/JCP.2015.20.4.232
Koller, M., & Saleh, H.M. 2018. "Introductory Chapter: Introducing Heavy Metals." In InTech. https://doi.org/10.5772/intechopen.74783.
Kot, F.S. 2018. "On the rubidium and lithium content and availability in the sub-arid south-eastern Mediterranean: potential health implications." Environ Geochem Health 40: 1841–1851. https://doi.org/10.1007/s10653-018-0134-8.
Krasnov, H., Katra, I., Novack, V., Vodonos, A., & Friger, M. D. 2015. "Increased indoor PM concentrations controlled by atmospheric dust events and urban factors." Building and Environment 87: 169-176. https://doi.org/10.1016/j.buildenv.2015.01.035.
Kristensen, L. J., & Taylor, M. P. 2012. Fields and forests in flames: lead and mercury emissions from wildfire pyrogenic activity. Environmental health perspectives, 120(2), a56–a57. https://doi.org/10.1289/ehp.1104672
Kummer, U., Pacyna, J., Pacyna, E., Friedrich, R. 2009. Assessment of heavy metal releases from the use phase of road transport in Europe. Atmos. Environ, 43, 640–647. DOI: 10.1016/j.atmosenv.2008.10.007
Kumpienė, J., Brännvall, E., Taraškevičius, R., Aksamitauskas, E., & Zinkutė, R. 2011. "Spatial Variability of Topsoil Contamination with Trace Elements in Preschools in Vilnius, Lithuania." J. Geochem. Explor. 108(1): 15–20. https://doi.org/10.1016/j.gexplo.2010.08.003.

Kunt, F., & Türkyılmaz, E.S. 2023. Detection of Heavy Metals in Educational Institutions' Indoor Dust and Their Risks to Health. Atmosphere, 14, 780.
Kuo, H. W., & Shen, H. Y. 2010. "Indoor and outdoor PM2.5 and PM10 concentrations in the air during a dust storm." Building and Environment 45 (3): 610-614. https://doi.org/10.1016/j.buildenv.2009.07.017.
Kurt-Karakus, P.B. 2012. "Determination of heavy metals in indoor dust from Istanbul, Turkey: Estimation of the health risk." *Environmental International, 50,* 47–55. DOI: 10.1016/j.envint.2012.09.011.
Kvietkus, K., Šakalys, J., & Valiulis, D. 2011. "Trends of Atmospheric Heavy Metal Deposition in Lithuania." Lithuanian Journal of Physics 51: 359-369. https://dx.doi.org/10.3952/lithjphys.51413.
Kvietkus, K., Šakalys, J., Didzbalis, J., Garbarienė, I., Špirkauskaitė, N., & Remeikis, V. 2013. Atmospheric aerosol episodes over Lithuania after the May 2011 volcano eruption at Grimsvötn, Iceland. Atmos. Res., 122, 93–101.
L. Borgna, L. A. Di Lella, F. Nannoni et al., "The high contents of lead in soils of Northern Kosovo," Journal of Geochemical Exploration, vol. 101, no. 2, pp. 137–146, 2009.
L. Medici, J. Bellanova, C. Belviso et al., "Trace metals speciation in sediments of the Basento River (Italy)," Applied Clay Science, vol. 53, no. 3, pp. 414–442, 2011.
L. Slavkovic, B. Skrbic, and N. Miljevic, "Antonije Onjia, Principal component analysis of trace elements in industrial soils," Environmental Chemistry Letters, vol. 2, pp. 105–108, 2004.
Lanzerstorfer, C. 2017. Variations in the Composition of House Dust by Particle Size. *J. Environ. Sci. Health, Part A, 52*(8), 770–777. DOI: 10.1080/10934529.2017.1303316
Laperche, V., & Lemière, B. 2020. "Possible Pitfalls in the Analysis of Minerals and Loose Materials by Portable XRF, and How to Overcome Them." *Minerals, 11,* 33. DOI: 10.3390/min11010033.
Latif, M.T., Yong, S.M., Saad, A., Mohamad, N., Baharudin, N.H., Bin Mokhtar, M., & Tahir, N.M. 2013. "Composition of heavy metals in indoor dust and their possible exposure: A case study of preschool children in Malaysia." *Air Quality, Atmosphere & Health, 7,* 181–193. DOI: 10.1007/s11869-013-0224-9.
Lee, D. B. N., Roberts, M., Bluchel, C. G., & Odell, R. A. 2010. "Zirconium: Biomedical and Nephrological Applications." ASAIO Journal 56(6): 550-556. https://doi.org/10.1097/MAT.0b013e3181e73f20
Lee, E.S., Fung, C.-C.D., Zhu, Y. 2015. Evaluation of a High Efficiency Cabin Air (HECA) Filtration System for Reducing Particulate Pollutants Inside

School Buses. Environ. Sci. Technol, 49, 3358–3365. https://doi.org/10.1021/es505419m.
Lei Chai, Yuhong Wang, Xin Wang, Liang Ma, Zhenxiang Cheng, Limin Su. 2021. "Pollution characteristics, spatial distributions, and source apportionment of heavy metals in cultivated soil in Lanzhou, China." *Ecological Indicators, 125,* 107507. DOI: 10.1016/j.ecolind.2021.107507.
Li, F., Huang, J., Zeng, G., Huang, X., Liu, W., Wu, H., Yuan, Y., & He, X. 2014. "Spatial distribution and health risk assessment of toxic metals associated with receptor population density in street dust: A case study of Xiandao District, Changsha, Middle China." *Environmental Science and Pollution Research, 22,* 6732–6742. DOI: 10.1007/s11356-014-3753-3.
Li, Y. M., Ma, J. H., Liu, D. X., Sun, Y. L., Chen, Y. F. 2015. Assessment of Heavy Metal Pollution and Potential Ecological Risks of Urban Soils in Kaifeng City. *Environ. Sci., 36*(3), 1037–1044. PMID: 25929074
Li, Y., Ye, F., Wang, A., Wang, D., Yang, B., Zheng, Q., Sun, G., & Gao, X. 2016. "Chronic Arsenic Poisoning Probably Caused by Arsenic-Based Pesticides: Findings from an Investigation Study of a Household." International journal of environmental research and public health 13 (1): 133. https://doi.org/10.3390/ijerph13010133.
Li, Y., Yu, Y., Yang, Z., Shen, Z., Wang, X., Cai, Y. 2016. A Comparison of Metal Distribution in Surface Dust and Soil among Super City, Town, and Rural Area. *Environ. Sci. Pollut. Res., 23*(8), 7849–7860. DOI: 10.1007/s11356-015-5911-7
Lim, S.-R., Kang, D., Ogunseitan, O.A., Schoenung, J.M. 2013. Potential Environmental Impacts from the Metals in Incandescent, Compact Fluorescent Lamp (CFL), and Light-Emitting Diode (LED) Bulbs. Environ. Sci. Technol, 47, 1040–1047. DOI: 10.1021/es302886m
Linde, M. 2005. Trace Metals in Urban Soils: Stockholm as a Case Study. Acta Universitatis Agriculturae Sueciae, 2005:111. Department of Soil Sciences, Swedish University of Agricultural Sciences. ISBN: 91-576-6910-4. Available at: https://res.slu.se/id/publ/13086.
Linde, M., Bengtsson, H., & Öborn, I. 2001. "Concentrations and Pools of Heavy Metals in Urban Soils in Stockholm, Sweden." Water, Air, & Soil Pollution: Focus 1: 83–101. https://doi.org/10.1023/A:1017599920280.
Liu, M.-M., Wang, D., Zhao, Y., Liu, Y.-Q., Huang, M.-M., Liu, Y., Sun, J., Ren, W.-H., Zhao, Y.-D., He, Q.-C., Dong, G.-H. 2013. "Effects of Outdoor and Indoor Air Pollution on Respiratory Health of Chinese Children from 50 Kindergartens." Journal of Epidemiology 23 (4): 280-287. https://doi.org/10.2188/jea.JE20120175.

Ljung, K., Selinus, O., & Otabbong, E. 2006. "Metals in Soils of Children's Urban Environments in the Small Northern European City of Uppsala." The Science of the Total Environment 366(2–3): 749-59. DOI: 10.1016/J.SCITOTENV.2005.09.073.

Loska, K., Wiechuła, D., & Korus, I. 2004. "Metal contamination of farming soils affected by industry." Environment International 30 (2): 159-165. https://doi.org/10.1016/S0160-4120(03)00157-0.

Ludia Bityukova. 1993. "Heavy Metals in the Soils of Tallinn (Estonia) and Its Suburbs." Geomicrobiology Journal 11(3–4): 285-298. DOI: 10.1080/01490459309377958.

Jordan M. M., M. A. Montero, S. Pina, and E. Garc ´ ´ıaSanchez, "Mineralogy and distribution of Cd, Ni, Cr, and Pb in ´ biosolids-amended soils from castellon province (NE, Spain)," ´ Soil Science, vol. 174, no. 1, pp. 14–20, 2009.

Nadal M., M. Mari, M. Schuhmacher, and J. L. Domingo, "Multi-compartmental environmental surveillance of a petrochemical area: levels of micropollutants," Environment International, vol. 35, no. 2, pp. 227–235, 2009.

Nadal M., M. Schuhmacher, and J. L. Domingo, "Levels of metals, PCBs, PCNs and PAHs in soils of a highly industrialized chemical/petrochemical area: temporal trend," Chemosphere, vol. 66, no. 2, pp. 267–276, 2007.

Mahanta, M. J., Bhattacharyya, K. G. 2010. Total Concentrations, Fractionation and Mobility of Heavy Metals in Soils of Urban Area of Guwahati, India. *Environ. Monit. Assess., 173*(1–4), 221–240. DOI: 10.1007/s10661-010-1383-x

Mainka, A., & Zajusz-Zubek, E. 2015. "Indoor Air Quality in Urban and Rural Preschools in Upper Silesia, Poland: Particulate Matter and Carbon Dioxide." Int. J. Environ. Res. Public Health 12: 7697-7711. https://doi.org/10.3390/ijerph120707697.

Mainka, A., Zajusz-Zubek, E., Kozielska, B., & Brągoszewska, E. 2018. "E3S Web of Conferences." 28: 01022. https://doi.org/10.1051/e3sconf/20182801022.

Man-tin, X. 2012. "Effects of atmospheric mercury pollution on human health in China: a review of recent studies." Journal of Environmental Health.

Mantler, M., & Schreiner, M. 2001. "X-ray analysis of objects of art and archaeology." *Journal of Radioanalytical and Nuclear Chemistry, 247,* 635–644. DOI: 10.1023/a:1010671619353.

Mao-jiang, W. 2012. "The Relationship Between Strontium and Human Health." Studies of Trace Elements and Health.

Martenies, S.E. Batterman, S.A. 2018. Effectiveness of Using Enhanced Filters in Schools and Homes to Reduce Indoor Exposures to PM2.5 from Outdoor Sources and Subsequent Health Benefits for Children with Asthma. Environ. Sci. Technol, 52, 10767–10776. https://doi.org/10.1021/acs.est.8b02053.
Masindi, V., Mkhonza, P., Tekere, M. 2021. Sources of Heavy Metals Pollution. In: Inamuddin, Ahamed, M.I., Lichtfouse, E., Altalhi, T. (eds) Remediation of Heavy Metals. Environmental Chemistry for a Sustainable World, vol 70. Springer, Cham. https://doi.org/10.1007/978-3-030-80334-6_17
Mather, T.A. 2015. "Volcanoes and the environment: Lessons for understanding Earth's past and future from studies of present-day volcanic emissions." *Journal of Volcanology and Geothermal Research* 304: 160-179. https://doi.org/10.1016/j.jvolgeores.2015.08.016.
Matyžiūtė-Jodkonienė, D. 2007. Contamination of Soil and Grass by Heavy Metals along the Main Roads in Lithuania.
Taşan M., Demir Y., Taşan s. 2022. Groundwater quality assessment using principal component analysis and hierarchical cluster analysis in Alaçam, Turkey. Water Supply 1 March 2022; 22 (3): 3431–3447. https://doi.org/10.2166/ws.2021.390
Jiang Y, Guo H., Jia Y., Yongsheg C., Chao H. 2015. Principal component analysis and hierarchical cluster analyses of arsenic groundwater geochemistry in the Hetao basin, Inner Mongolia, Geochemistry, Volume 75, Issue 2, Pages 197-205, ISSN 0009-2819, https://doi.org/10.1016/j.chemer.2014.12.002
Szczepanik M., Szyszlak-Bargłowicz J., Zając G., Koniuszy A., Hawrot-Paw M., Wolak A. 2021. The Use of Multivariate Data Analysis (HCA and PCA) to Characterize Ashes from Biomass Combustion. Energies. 14. 6887. 10.3390/en14216887
McNeill, J., Snider, G., Weagle, C. L. 2020. Large global variations in measured airborne metal concentrations driven by anthropogenic sources. *Scientific Reports, 10*, 21817. https://doi.org/10.1038/s41598-020-78789-y.
Milukaitė, A., & Klanova, J., Holoubek, I., Rimšelytė, I., & Kvietkus, K. 2008. "Persistent Organic Pollutants in Lithuania: Assessment of Air and Soil Contamination." Lithuanian Journal of Physics 48. https://doi.org/10.3952/physics.v48i4.1947.
Milukaite, A., Juozefaite, V., Mikelinskiene, A. et al. 1995. "Evaluation of Extreme Pollution Episodes over Lithuania in 1980–1994." Water Air Soil Pollut 85: 2003–2008. https://doi.org/10.1007/BF01186128.

Misiune, I., Julian J. P., Veteikis, D. 2021. Pull and push factors for use of urban green spaces and priorities for their ecosystem services: Case study of Vilnius, Lithuania,Urban Forestry & Urban Greening,Volume 58,126899,ISSN 1618-8667,https://doi.org/10.1016/j.ufug.2020.126899.

Mitsakou, C., Kallos, G., Papantoniou, N., Spyrou, C., Solomos, S., Astitha, M., and Housiadas, C. 2008. "Saharan dust levels in Greece and received inhalation doses." Atmos. Chem. Phys. 8: 7181–7192. https://doi.org/10.5194/acp-8-7181-2008.

Moghtaderi, M., Teshnizi, S.H., Moghtader, T., Ashraf, M.A., & Faraji, H. 2020. The Safety of Schools Based on Heavy Metal Concentrations in Classrooms' Dust: A Systematic Review and Meta-Analysis. Iran. J. Public Health, 49, 2287–2294.

Mohammadi Nafchi, A., Blouin, V., Kaye, N., Metcalf, A., Van Valkinburgh, K., & Mousavi, E. 2021. "Room HVAC Influences on the Removal of Airborne Particulate Matter: Implications for School Reopening during the COVID-19 Pandemic." *Energies* 14: 7463. https://doi.org/10.3390/en14227463.

Mohammed, M.A., Bulama, K., Bukar, A.M., Modu, M.A., Usman, A.A., Lawan, A.K., & Habib, G.A. 2023. "The impacts of building opening characteristics on dust particle deposition indoors." International Journal of Building Pathology and Adaptation. https://doi.org/10.1108/IJBPA-10-2022-0162.

Mohan M., S. 2016. An overview of particulate dry deposition: measuring methods, deposition velocity and controlling factors. Int. J. Environ. Sci. Technol. 13, 387–402. https://doi.org/10.1007/s13762-015-0898-7

Muezzinoglu, A., & Cukurluoglu Cizmecioglu, S. 2006. "Deposition of Heavy Metals in a Mediterranean Climate Area." Atmospheric Research 81 (1): 1-16. https://doi.org/10.1016/j.atmosres.2005.10.004.

Muhamad-Darus, F., Nasir, R.A., Sumari, S.M., Ismail, Z.S., & Omar, N.A. 2017. "Nursery Schools: Characterization of heavy metal content in indoor dust." *Asian Journal of Environmental Studies, 2,* 63–70. DOI: 10.21834/aje-bs.v2i5.223.

Müller, G. 1969. Index of geoaccumulation in sediments of the Rhine River. GeoJournal, 2, 108–118

Murakami, M., Nakajima, F., Furumai, H., Tomiyasu, B., Owari, M. 2007. Identification of particles containing chromium and lead in road dust and soakaway sediment by electron probe microanalyser. Chemosphere, 67, 2000–2010. https://doi.org/10.1016/j.chemosphere.2006.11.044.

Nagajyoti, P.C., Lee, K.D., & Sreekanth, T.V.M. 2010. "Heavy metals, occurrence and toxicity for plants: a review." Environmental Chemistry Letters 8: 199–216. https://doi.org/10.1007/s10311-010-0297-8.

Naher, S., Haseeb, A. 2006. A technical note on the production of zirconia and zircon brick from locally available zircon in Bangladesh. J. Am. Acad. Dermatol, 172, 388–393. https://doi.org/10.1016/j.jmatprotec.2005.10.013.

Naimabadi, A., Gholami, A., & Ramezani, A.M. 2020. "Determination of heavy metals and health risk assessment in indoor dust from different functional areas in Neyshabur, Iran." *Indoor Built Environment, 30,* 1781–1795. DOI: 10.1177/1420326x20963378.

Napa, Ü., Ostonen, I., Kabral, N., Kriiska, K., & Frey, J. 2017. "Biogenic and Contaminant Heavy Metal Pollution in Estonian Coniferous Forests." Regional Environmental Change 17: 2111-2120. DOI: 10.1007/s10113-017-1206-5.

Nasim Karim. 2018. "Copper and Human Health- A Review." Journal of Bahria University Medical and Dental College 8(2): 117–122. https://doi.org/10.51985/JBUMDC2018046

Nieder, R., Benbi, D.K., Reichl, F.X. 2018. "Soil-Borne Particles and Their Impact on Environment and Human Health." In Soil Components and Human Health. Springer, Dordrecht. https://doi.org/10.1007/978-94-024-1222-2_3.

Nikodēmus, O., Brūmelis, G. 1994. The spatial dynamics of atmospheric pollution in Latvia and the Baltic Republics, as measured in mosses, topsoil and precipitation. *GeoJournal, 33*, 71–80. PubMed

Nkansah, M.A., Fianko, J.R., Mensah, S., Debrah, M., Francis, G.W. 2015. Determination of heavy metals in dust from selected nursery and kindergarten classrooms within the Kumasi metropolis of Ghana. *Cogent Chem*, 1, 1119005. DOI: 10.1080/23312009.2015.1119005

Noris, F., Siegel, J. A., & Kinney, K. A. 2009. "Biological and metal contaminants in HVAC filter dust." *ASHRAE Transactions* 115 (2). Retrieved from http://www.ashrae.org.

Noris, F., Siegel, J. A., & Kinney, K. A. 2011. "Evaluation of HVAC filters as a sampling mechanism for indoor microbial communities." *Atmospheric Environment* 45 (2): 338-346. https://doi.org/10.1016/j.atmosenv.2010.10.017.

Norris, G., Duvall, R., Brown, S., & Bai, S. 2014. *EPA Positive Matrix Factorization (PMF) 5.0 Fundamentals and User Guide.* U.S. Environmental Protection Agency, EPA/600/R-14/108, April 2014.

Nõu, T., & Viljasoo, V. 2011. "The effect of heating systems on dust, an indoor climate factor." Agronomy research 9: 165-174. https://agronomy.emu.ee/vol09Spec1/p09s121.pdf.
O. A. Al-Khashman, "Heavy metal distribution in dust, street dust and soils from the work place in Karak Industrial Estate, Jordan," Atmospheric Environment, vol. 38, no. 39, pp. 6803–6812, 2004.
O'Connor, D., Hou, D., Ye, J., Zhang, Y., Ok, Y.S., Song, Y., Coulon, F., Peng, T., Tian, L. 2018. Lead-based paint remains a major public health concern: A critical review of global production, trade, use, exposure, health risk, and implications. Environ. Int, 121, 85–101. https://doi.org/10.1016/j.envint.2018.08.052.
Olawoyin, R., Oyewole, S. A., & Grayson, R. L. 2012. "Potential risk effect from elevated levels of soil heavy metals on human health in the Niger delta." Ecotoxicology and Environmental Safety 85: 120–130. https://doi.org/10.1016/j.ecoenv.2012.08.002
Oliva, S.R., & Fernández Espinosa, A.J. 2007. "Monitoring of heavy metals in topsoils, atmospheric particles, and plant leaves to identify possible contamination sources." Microchemical Journal 86 (1): 131-139. https://doi.org/10.1016/j.microc.2007.01.003.
Olowoyo, J.O., Mugivhisa, L.L., Magoloi, Z.G. 2016. Composition of Trace Metals in Dust Samples Collected from Selected High Schools in Pretoria, South Africa. *Appl. Environ. Soil Sci.* 2016, 1–9. DOI: 10.1155/2016/5829657
Olujimi, O., Steiner, O., & Goessler, W. 2015. "Pollution indexing and health risk assessments of trace elements in indoor dusts from classrooms, living rooms and offices in Ogun State, Nigeria." *Journal of African Earth Sciences, 101,* 396–404.
Orru, H., Laukaitienė, A., & Zurlytė, I. 2012. "Particulate air pollution and its impact on health in Vilnius and Kaunas." Medicina (Kaunas, Lithuania) 48 (9): 472–477.
Othman, M., Latif, M. T., & Matsumi, Y. 2019. "The exposure of children to PM2.5 and dust in indoor and outdoor school classrooms in Kuala Lumpur City Centre." Ecotoxicology and Environmental Safety 170: 739-749. https://doi.org/10.1016/j.ecoenv.2018.12.042.
P. K. Govil, J. E. Sorlie, N. N. Murthy et al., "Soil contamination of heavy metals in the Katedan Industrial Development Area, Hyderabad, India," Environmental Monitoring and Assessment, vol. 140, no. 1–3, pp. 313–323, 2008.

Pacyna, J.M. 1986. "Atmospheric trace elements from natural and anthropogenic sources." In Toxic Metals in the Atmosphere, edited by J.O. Nriagu and C.I. Davidson, Chap. 2. New York: Wiley.
Panagos, P., Van Liedekerke, M., Yigini, Y., Montanarella, L. 2013. "Contaminated Sites in Europe: Review of the Current Situation Based on Data Collected through a European Network." Journal of Environmental and Public Health 2013: 158764. https://doi.org/10.1155/2013/158764.
Pegas, P. N., Evtyugina, M. G., Alves, C. A., Nunes, T., Cerqueira, M., Franchi, M., ... Freitas, M. do C. 2010. "Outdoor/indoor air quality in primary schools in Lisbon: a preliminary study." Química Nova 33 (5). https://doi.org/10.1590/S0100-40422010000500027.
Peng, C., Wang, M., Chen, W. 2016. Modelling Cadmium Contamination in Paddy Soils Under Long-Term Remediation Measures: Model Development and Stochastic Simulations. *Environ. Pollut., 216,* 146–155. DOI: 10.1016/j.envpol.2016.05.038
Pichhode, M., & Kumar, N. 2016. "Effect of Heavy Metals on Plants: An Overview." https://doi.org/10.13140/RG.2.2.27583.87204.
Polyakova, E. V. 2012. "STRONTIUM IN WATER-SUPPLY SOURCES OF ARKHANGELSK REGION AND ITS IMPACT ON HUMAN HEALTH." Ekologiya Cheloveka (Human Ecology) 19 (2): 9-14. doi: 10.17816/humeco17503.
Punia, A. 2021. Role of temperature, wind, and precipitation in heavy metal contamination at copper mines: a review. Environ Sci Pollut Res 28, 4056–4072. https://doi.org/10.1007/s11356-020-11580-8
Qin, G., Niu, Z., Yu, J., Li, Z., Ma, J., & Xiang, P. 2021. "Soil Heavy Metal Pollution and Food Safety in China: Effects, Sources and Removing Technology." Chemosphere 267: 129205. https://doi.org/10.1016/j.chemosphere.2020.129205.
Qin, G., Niu, Z., Yu, J., Li, Z., Ma, J., & Xiang, P. 2021. "Soil heavy metal pollution and food safety in China: Effects sources and removing technology." Chemosphere 267: 129205. https://doi.org/10.1016/j.chemosphere.2020.129205
Qing, X., Yutong, Z., & Shenggao, L. 2015. "Assessment of heavy metal pollution and human health risk in urban soils of steel industrial city (Anshan), Liaoning, Northeast China." *Ecotoxicology and Environmental Safety, 120,* 377–385. DOI: 10.1016/j.ecoenv.2015.06.019.
Radhi, A.B., Shartooh, S.M., & Al-Heety, E.A. 2021. "Heavy Metal Pollution and Sources in Dust from Primary Schools and Kindergartens in Ramadi City, Iraq." *Iraqi Journal of Science,* 1816–1828. DOI: 10.24996/ijs.2021.62.6.7.

Rama Jyothi, N. 2021. "Heavy Metal Sources and Their Effects on Human Health." IntechOpen. https://doi.org/10.5772/intechopen.95370.
Rashed, M.N. 2008. Total and Extractable Heavy Metals in Indoor, Outdoor and Street Dust from Aswan City, Egypt. Clean-soil Air Water, 36, 850-857.
Recknagel, S., Radant, H., Kohlmeyer, R. 2014. Survey of mercury, cadmium and lead content of household batteries. Waste Manag, 34, 156–161. DOI: 10.1016/j.wasman.2013.09.024
Reimann, C., Siewers, U., Tarvainen, T., Bityukova, L., Eriksson, J., Gilucis, A., Gregorauskiene, V., Lukashev, V., Matinian, N.N., & Pasieczna, A. 2000. "Baltic Soil Survey: Total Concentrations of Major and Selected Trace Elements in Arable Soils from 10 Countries Around the Baltic Sea." Science of The Total Environment 257(2–3): 155-170. https://doi.org/10.1016/S0048-9697(00)00515-5.
Rim, K. T., Koo, K. H., & Park, J. S. 2013. "Toxicological Evaluations of Rare Earths and Their Health Impacts to Workers: A Literature Review." Safety and Health at Work 4 (1): 12-26. https://doi.org/10.5491/SHAW.2013.4.1.12.
Ritz, B., Hoffmann, B., & Peters, A. 2019. "The effects of fine dust, ozone, and nitrogen dioxide on health." Dtsch Arztebl Int 116: 881–6. DOI: 10.3238/arztebl.2019.0881.
Rodríguez-Chávez, T.B., Rine, K.P., Almusawi, R.M. 2021. "Outdoor/Indoor Contaminant Transport by Atmospheric Dust and Aerosol at an Active Smelter Site." Water Air Soil Pollut 232: 226. https://doi.org/10.1007/s11270-021-05168-2.
Ross, S.M. 1994. Toxic Metals in Soil-Plant Systems. Chichester: Wiley.
Ruhling, A. et al. 1992. "Atmospheric Heavy Metal Deposition in Northern Europe 1990." Nord 12.
S. Maas, R. Scheifler, M. Benslama et al., “Spatial distribution of heavy metal concentrations in urban, suburban and agricultural soils in a Mediterranean city of Algeria,” Environmental Pollution, vol. 158, no. 6, pp. 2294–2301, 2010.
S. O. Fakayode and P. C. Onianwa, “Heavy metal contamination of soil, and bioaccumulation in Guinea grass (Panicum maximum) around Ikeja Industrial Estate, Lagos, Nigeria,” Environmental Geology, vol. 43, no. 1-2, pp. 145–150, 2002.
S. R. Tariq, M. H. Shah, N. Shaheen, A. Khalique, S. Manzoor, and M. Jaffar, “Multivariate analysis of trace metal levels in tannery effluents in relation to soil and water: a case study from Peshawar, Pakistan,” Journal of Environmental Management, vol. 79, no. 1, pp. 20–29, 2006.

S. Rodr´ıguez, X. Querol, A. Alastuey et al., “Comparative PM10-PM2.5 source contribution study at rural, urban and industrial sites during PM episodes in Eastern Spain,” Science of the Total Environment, vol. 328, no. 1–3, pp. 95–113, 2004.

S. Shallari, C. Schwartz, A. Hasko, and J. L. Morel, “Heavy metals in soils and plants of serpentine and industrial sites of Albania,” Science of the Total Environment, vol. 209, no. 2-3, pp. 133–142, 1998.

S. Srinivasa Gowd, M. Ramakrishna Reddy, and P. K. Govil, “Assessment of heavy metal contamination in soils at Jajmau (Kanpur) and Unnao industrial areas of the Ganga Plain, Uttar Pradesh, India,” Journal of Hazardous Materials, vol. 174, no. 1–3, pp. 113–121, 2010.

S. Tokahoglu and S. Kartal, “Multivariate analysis of the data and speciation of heavy metals in street dust samples from the Organized Industrial District in Kayseri (Turkey),” Atmospheric Environment, vol. 40, no. 16, pp. 2797–2805, 2006.

Saarimaa, V., Kaleva, A., Ismailov, A., Laihinen, T., Virtanen, M., Levänen, E., Väisänen, P. 2022. Corrosion product formation on zinc-coated steel in wet supercritical carbon dioxide. Arab. J. Chem, 15, 103636. DOI: 10.1016/j.arabjc.2021.103636

Saha, A., Gupta, B.S., Patidar, S., & Martínez-Villegas, N. 2022. "Spatial distribution and source identification of metal contaminants in the surface soil of Matehuala, Mexico based on positive matrix factorization model and GIS techniques." *Frontiers in Soil Science, 2.* DOI: 10.3389/fsoil.2022.1041377.

Saraga, D., Maggos, T., Sadoun, E., Fthenou, E., Hassan, H., Tsiouri, V., Karavoltsos, S., Sakellari, A., Vasilakos, C., & Kakosimos, K. 2017. "Chemical Characterization of Indoor and Outdoor Particulate Matter (PM2.5, PM10) in Doha, Qatar." Aerosol Air Qual. Res. 17: 1156-1168. https://doi.org/10.4209/aaqr.2016.05.0198.

Schneider, T. 2008. "Dust and fibers as a cause of indoor environment problems." *SJWEH Suppl* 2008;(4):10–1. ISSN 0355-3140.

Schwartz, J. 2004. "Air Pollution and Children's Health." Pediatrics 113 (Supplement_3): 1037–1043. https://doi.org/10.1542/peds.113.S3.1037.

Schweitzer, M. D., Calzadilla, A. S., Salamo, O., Sharifi, A., Kumar, N., Holt, G., & Campos, M. 2018. "Lung health in the era of climate change and dust storms." Environmental Research 163: 36-42. https://doi.org/10.1016/j.envres.2018.02.001.

Sesli, Y., Ozomay, Z., Kandirmaz, E.A.; Ozcan, A. 2018. The Investigation of Using Zirconium Oxide Microspheres in Paper Coating; Marmara University,

School of Applied Sciences, Printing Technologies: Istanbul, Turkey. https://doi.org/10.24867/GRID-2018-p15
Sezgin, N., Ozcan, H., Demir, G., Nemlioglu, S., & Bayat, C. 2004. Determination of heavy metal concentrations in street dusts in Istanbul E-5 highway. Environ. Int, 29, 979–985.
Shi, T., & Wang, Y. 2021. "Heavy metals in indoor dust: Spatial distribution, influencing factors, and potential health risks." *Science of the Total Environment, 755,* 142367. DOI: 10.1016/j.scitotenv.2020.142367.
Shilton, V., Giess, P., Mitchell, D., & Williams, C. 2002. "The Relationships between Indoor and Outdoor Respirable Particulate Matter: Meteorology, Chemistry and Personal Exposure." Indoor and Built Environment 11 (5): 266-274. doi:10.1177/1420326X0201100503.
Shrivastav, R. 2001. "Atmospheric Heavy Metal Pollution." Reson 6: 62–68. https://doi.org/10.1007/BF02994594.
Smith, J. L., & Lee, K. 2003. "Soil as a Source of Dust and Implications for Human Health." Advances in Agronomy 80: 1-32. https://doi.org/10.1016/S0065-2113(03)80001-9.
Soldi, T., Riolo, C., Alberti, G., Gallorini, M., Peloso, G. 1996. Environmental vanadium distribution from an industrial settlement. Sci. Total Environ, 181, 45–50. https://doi.org/10.1016/0048-9697(95)04958-4.
Song, X., Wang, P., Van Zwieten, L. 2022. Towards a better understanding of the role of Fe cycling in soil for carbon stabilization and degradation. carbon res 1, 5. https://doi.org/10.1007/s44246-022-00008-2
Soriano, A., Pallarés, S., Pardo, F., Vicente, A.B., Sanfeliu, T., & Bech, J. 2012. "Deposition of Heavy Metals from Particulate Settleable Matter in Soils of an Industrialized Area." Journal of Geochemical Exploration 113: 36-44. https://doi.org/10.1016/j.gexplo.2011.03.006.
Sorkheh, M., Asgari, H. M., Zamani, I., & Ghanbari, F. 2022. "The relationship between dust sources and airborne bacteria in the southwest of Iran." Environmental Science and Pollution Research 29: 82045–82063. https://doi.org/10.1007/s11356-022-21563-6.
Source characterisation of road dust based on chemical and mineralogical composition, Chemosphere, Volume 87, Issue 2, Pages 163-170, ISSN 0045-6535, https://doi.org/10.1016/j.chemosphere.2011.12.012.
Souto-Oliveira, C.E., Kamigauti, L.Y., Andrade, M.d.F., & Babinski, M. 2021. "Improving Source Apportionment of Urban Aerosol Using Multi-Isotopic Fingerprints (MIF) and Positive Matrix Factorization (PMF): Cross-Validation and New Insights." *Frontiers in Environmental Science, 9.* DOI: 10.3389/fenvs.2021.623915.

Srivastava, D., Xu, J., Vu, T. V., Liu, D., Li, L., Fu, P., Hou, S., Shi, Z., & Harrison, R. M. 2021. "Insight into PM2.5 sources by applying positive matrix factorization (PMF) at an urban and rural site of Beijing." *Atmospheric Chemistry and Physics* 21: 14703-14724. https://doi.org/10.5194/acp-21-14703-2021.

Staykova, J., Dimitrova, J., & Titopoulou, M. 2022. "Air Quality and Health Status of the Population in a District of Bulgaria." JBRES 3 (12): 1541-1544. doi: 10.37871/jbres1632.

Steinnes, E., Allen, R.O., Petersen, H.M., Rambæk, J.P., & Varskog, P. 1997. "Evidence of Large Scale Heavy-Metal Contamination of Natural Surface Soils in Norway from Long-Range Atmospheric Transport." Science of The Total Environment 205(2–3): 255-266. https://doi.org/10.1016/S0048-9697(97)00209-X.

Steinnes, E., Solberg, W., Petersen, H.M. et al. 1989. "Heavy Metal Pollution by Long Range Atmospheric Transport in Natural Soils of Southern Norway." Water Air Soil Pollut 45: 207–218. https://doi.org/10.1007/BF00283452.

Stern, B. R., Solioz, M., Krewski, D., Aggett, P., Aw, T. C., Baker, S., Crump, K., Dourson, M., Haber, L., Hertzberg, R., Keen, C., Meek, B., Rudenko, L., Schoeny, R., Slob, W., & Starr, T. 2007. "Copper and human health: biochemistry, genetics, and strategies for modeling dose-response relationships." Journal of toxicology and environmental health. Part B, Critical reviews 10(3): 157–222. https://doi.org/10.1080/10937400600755911

Sternbeck, J., Sjödin, Å., Andréasson, K. 2002. Metal emissions from road traffic and the influence of resuspension—results from two tunnel studies. Atmos. Environ, 36, 4735–4744. DOI: 10.1016/s1352-2310(02)00561-7

Sternberg, T., & Edwards, M. 2017. "Desert Dust and Health: A Central Asian Review and Steppe Case Study." Int. J. Environ. Res. Public Health 14: 1342. https://doi.org/10.3390/ijerph14111342.

Straffelini, G., Gialanella, S. 2016. Airborne particulate matter from brake systems: An assessment of the relevant tribological formation mechanisms. Wear 2021, 478-479, 203883. DOI: 10.1016/j.wear.2021.203883

Stranger, M., Potgieter-Vermaak, S. S., & Van Grieken, R. 2008. "Characterization of indoor air quality in primary schools in Antwerp, Belgium." Indoor Air 18 (6): 454-463. https://doi.org/10.1111/j.1600-0668.2008.00545.x.

Sulaiman, F.R., Bakri, N.I.F., Nazmi, N., Latif, M.T. 2016. Assessment of heavy metals in indoor dust of a university in a tropical environment. *Environ. Forensics*, 18, 74–82. DOI: 10.1080/15275922.2016.1263903

Suryawanshi, P.V., Rajaram, B.S., Bhanarkar, A.D., & Rao, C.V.C. 2016. Determining heavy metal contamination of road dust in Delhi, India. Atmósfera, 29, 221–234.
Susinski, M., Dodocioiu, A.M., & Mocanu, R. 2009. "Environmental issues induced by the sterile dump from Husnicioara District Mehedinți."
Swain, C.K. 2024. Environmental pollution indices: a review on concentration of heavy metals in air, water, and soil near industrialization and urbanisation. Discov Environ 2, 5. https://doi.org/10.1007/s44274-024-00030-8
Ta, A., KhKh, K., Eb, T., & Ai, I. 1989. "Experimental study of the effect of rubidium on the cardiovascular system." Gigiena truda i professional'nye zabolevaniia: 16-20.
Taraskevicius, R., Zinkutė, R., Čyžius, G.J., Kaminskas, M., & Jankauskaitė, M. 2013. "Soil Contamination in One of Preschools Influenced by Metal Working Industry." Vide. Tehnologija. Resursi - Environ. Technol. Resour. 1: 83-86.
Taraškevičius, R., Zinkutė, R., Jankauskaitė, M. 2008. "Differences of Vilnius Topsoil Contamination in the Neris River Valley Due to Anthropogenic Factors." Geologija 50(3): 135–142. https://doi.org/10.2478/v10056-008-0039-9.
Tarvainen, T., Reichel, S., Müller, I., Jordan, I., Hube, D., Eurola, M., Loukola-Ruskeeniemi, K. 2020. Arsenic in agro-ecosystems under anthropogenic pressure in Germany and France compared to a geogenic as region in Finland. J. Geochem. Explor, 217, 106606. https://doi.org/10.1016/j.gexplo.2020.106606.
Tchounwou, P. B., Ayensu, W. K., Ninashvili, N., & Sutton, D. 2003. "Environmental exposure to mercury and its toxicopathologic implications for public health." Environmental toxicology 18 (3): 149–175. https://doi.org/10.1002/tox.10116.
Timothy, N., & Williams, E.T. 2019. "Environmental Pollution by Heavy Metal: An Overview." International Journal of Environmental Chemistry 3 (2): 72-82. https://doi.org/10.11648/j.ijec.20190302.14.
TomTom. 2025. Traffic congestion map of Vilnius. Retrieved February 2, 2025, from https://www.tomtom.com/traffic-index/vilnius-traffic/
Tóth, G., Hermann, T., Szatmári, G., Pásztor, L. 2016. "Maps of Heavy Metals in the Soils of the European Union and Proposed Priority Areas for Detailed Assessment." Science of The Total Environment 565: 1054-1062. https://doi.org/10.1016/j.scitotenv.2016.05.115.
Tóth, T., Kovács, Z.A., Rékási, M. 2019. XRF-measured rubidium concentration is the best predictor variable for estimating the soil clay content

and salinity of semi-humid soils in two catenas. Geoderma, 342, 106–108. https://doi.org/10.1016/j.geoderma.2019.02.011.
Tumas, R. 2000. "Evaluation and Prediction of Nonpoint Pollution in Lithuania." Ecological Engineering 14(4): 443-451. https://doi.org/10.1016/S0925-8574(99)00068-3.
Tumas, R. 2000. Evaluation and prediction of nonpoint pollution in Lithuania. *Ecological Engineering, 14*(4), 443-451. DOI: 10.1016/S0925-8574(99)00068-3
Umeobi, E. C., Azuka, C. V., Ofem, K. I., John, K., Nemeček, K., Jidere, C. M., Ezeaku, P. I. 2024. Evaluation of potentially toxic elements in soils developed on limestone and lead-zinc mine sites in parts of southeastern Nigeria. *Heliyon, 10*(7), e27503. https://doi.org/10.1016/j.heliyon.2024.e27503.
USEPA. 2016. *Definition and procedure for the determination of the method detection limit, revision 2.* (2016). Available at: https://www.epa.gov/sites/default/files/2016-12/documents/mdl-procedure_rev2_12-13-2016.pdf
Usuda, K., Kono, R., Ueno, T., et al. 2014. "Risk Assessment Visualization of Rubidium Compounds: Comparison of Renal and Hepatic Toxicities, In vivo." Biol Trace Elem Res 159: 263–268. https://doi.org/10.1007/s12011-014-9937-3.
V. Cappuyns, S. Van Herreweghe, R. Swennen, R. Ottenburgs, and J. Deckers, “Arsenic pollution at the industrial site of Reppel-Bocholt (North Belgium),” Science of the Total Environment, vol. 295, no. 1–3, pp. 217–240, 2002.
Vasarevičius, S., Greičiūtė, K. 2004. Investigation of soil pollution with heavy metals in Lithuanian military grounds. *J. Environ. Eng. Landsc. Manag., 12*, 132–137. DOI: 10.3846/16486897.2004.9636834
Vega, F.A., Andrade, M.L., & Covelo, E.F. 2010. "Influence of Soil Properties on the Sorption and Retention of Cadmium, Copper and Lead, Separately and Together, by 20 Soil Horizons: Comparison of Linear Regression and Tree Regression Analyses." J. Hazard. Mater. 174: 522–533.
Vilnius Aplinka. 2023. "[Soil Pollution in Vilnius]." https://aplinka.vilnius.lt/aplinkos-kokybe/uzterstos-teritorijos/dirvozemio-tarsa/#17.
Vilnius Municipality. Particulate Matter. Available online: https://maps.vilnius.lt/ (accessed on 12 April 2023).
Vilnius Municipality. Soil Pollution. Available online: https://aplinka.vilnius.lt/aplinkos-kokybe/uzterstos-teritorijos/dirvozemio-tarsa/ (accessed on 20 December 2023).

Vlasov, D. Ramírez, O. Luhar, A. 2022. Road Dust in Urban and Industrial Environments: Sources, Pollutants, Impacts, and Management. Atmosphere 2022, 13, 607. https://doi.org/10.3390/atmos13040607
Vosoughi, M., Zavieh, F.S., Mokhtari, S.A., & Sadeghi, H. 2021. "Health risk assessment of heavy metals in dust particles precipitated on the screen of computer monitors." *Environmental Science and Pollution Research, 28,* 40771–40781. DOI: 10.1007/s11356-021-13407-6.
Wallner, P., Kundi, M., Moshammer, H., Piegler, K., Hohenblum, P., Scharf, S., ... Hutter, H.-P. 2012. "Indoor air in schools and lung function of Austrian school children." Journal of Environmental Monitoring 14: 1976-1982. https://doi.org/10.1039/C2EM30059A.
Wang, Q., Gu, J., & Wang, X. 2020. "The impact of Sahara dust on air quality and public health in European countries." Atmospheric Environment 241: 117771. https://doi.org/10.1016/j.atmosenv.2020.117771.
Wang, Q., Lu, X., Pan, H. 2016. Analysis of heavy metals in the re-suspended road dusts from different functional areas in Xi'an, China. Environ. Sci. Pollut. Res, 23, 19838–19846. DOI: 10.1007/s11356-016-7200-5
Werkenthin, M., Kluge, B., & Wessolek, G. 2014. "Metals in European Roadside Soils and Soil Solution – A Review." Environmental Pollution 189: 98-110. https://doi.org/10.1016/j.envpol.2014.02.025.
Witkowska, D., Słowik, J., Chilicka, K. 2021. Heavy Metals and Human Health: Possible Exposure Pathways and the Competition for Protein Binding Sites. *Molecules, 26,* 6060. DOI: 10.3390/molecules26196060
Wu, Y., Li, X., Yu, L., Wang, T., Wang, J., Liu, T. 2022. "Review of Soil Heavy Metal Pollution in China: Spatial Distribution, Primary Sources, and Remediation Alternatives." Resources, Conservation and Recycling 181: 106261. https://doi.org/10.1016/j.resconrec.2022.106261.
X. Li and C. Huang, "Environment impact of heavy metals on urban soil in the vicinity of industrial area of Baoji city, P.R. China," Environmental Geology, vol. 52, no. 8, pp. 1631–1637, 2007.
Yang, W., Sohn, J., Kim, J., Son, B., & Park, J. 2009. "Indoor air quality investigation according to age of the school buildings in Korea." Journal of Environmental Management 90 (1): 348-354. https://doi.org/10.1016/j.jenvman.2007.10.003.
Yesilkanat, C.M., & Kobya, Y. 2021. Spatial Characteristics of Ecological and Health Risks of Toxic Heavy Metal Pollution from Road Dust in the Black Sea Coast of Turkey. Geoderma Reg, 25, e00388.
Yuan, Y., Alahmad, B., Kang, C.-M., Al-Marri, F., Kommula, V., Bouhamra, W., ... Koutrakis, P. 2020. "Dust Events and Indoor Air Quality in Residential

Homes in Kuwait." Int. J. Environ. Res. Public Health 17: 2433. https://doi.org/10.3390/ijerph17072433.

Yuan, Y., Alahmad, B., Kang, C.-M., Al-Marri, F., Kommula, V., Bouhamra, W., & Koutrakis, P. 2020. "Dust Events and Indoor Air Quality in Residential Homes in Kuwait." *Int. J. Environ. Res. Public Health* 17: 2433. https://doi.org/10.3390/ijerph17072433.

Zacco, A., Resola, S., Lucchini, R., Albini, E., Zimmerman, N., Guazzetti, S., & Bontempi, E. 2009. "Analysis of settled dust with X-ray Fluorescence for exposure assessment of metals in the province of Brescia, Italy." *Journal of Environmental Monitoring, 11,* 1579–1585. DOI: 10.1039/b906430c.

Zamora-Ledezma, C., Negrete-Bolagay, D., Figueroa, F., Zamora-Ledezma, E., Ni, M., Alexis, F., & Guerrero, V.H. 2021. "Heavy metal water pollution: A fresh look about hazards novel and conventional remediation methods." Environmental Technology & Innovation 22: 101504. https://doi.org/10.1016/j.eti.2021.101504.

Zamora-Ledezma, C., Negrete-Bolagay, D., Figueroa, F., Zamora-Ledezma, E., Ni, M., Alexis, F., & Guerrero, V.H. 2021. "Heavy Metal Water Pollution: A Fresh Look about Hazards, Novel and Conventional Remediation Methods." Environmental Technology & Innovation 22: 101504. https://doi.org/10.1016/j.eti.2021.101504.

Zeider, K., Manjón, I., Betterton, E.A., Sáez, A.E., Sorooshian, A., & Ramírez-Andreotta, M.D. 2023. Backyard aerosol pollution monitors: Foliar surfaces, dust enrichment, and factors influencing foliar retention. Environ. Monit. Assess, 195, 1200.

Zeneli, Lulzim, & Nexhat Daci. 2014. "Strontium and its relationship with trace elements Mg, Cu, Co, and Mo in human blood and serum." Toxicological & Environmental Chemistry 96 (5): 808-813. DOI: 10.1080/02772248.2014.959016.

Zhang, L., Morisaki, H., Wei, Y., Li, Z., Yang, L., Zhou, Q., ... Tang, N. 2019. "Characteristics of air pollutants inside and outside a primary school classroom in Beijing and respiratory health impact on children." Environmental Pollution 255 (Part 1): 113147. https://doi.org/10.1016/j.envpol.2019.113147.

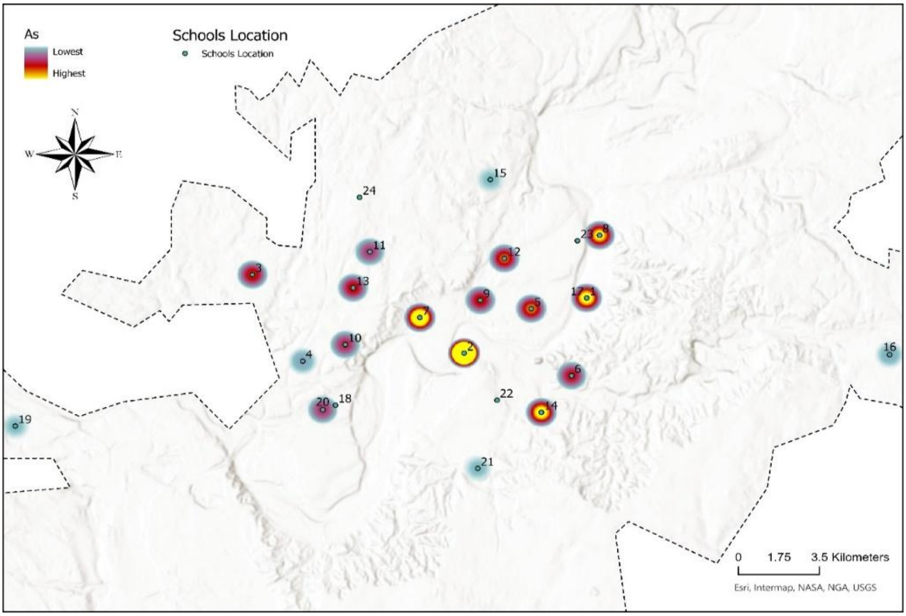
Zhao, D., Chen, H., Yu, E., Luo, T. 2019. PM2.5/PM10 Ratios in Eight Economic Regions and Their Relationship with Meteorology in China. Adv. Meteorol, 2019, 1–15. https://doi.org/10.1155/2019/5295726.

Zheng, Y., Gao, Q., Wen, X., Yang, M., Chen, H., Wu, Z., & Lin, X. 2012. "Multivariate statistical analysis of heavy metals in foliage dust near pedestrian bridges in Guangzhou, South China in 2009." *Environmental Earth Sciences, 70,* 107–113. DOI: 10.1007/s12665-012-2107-z.

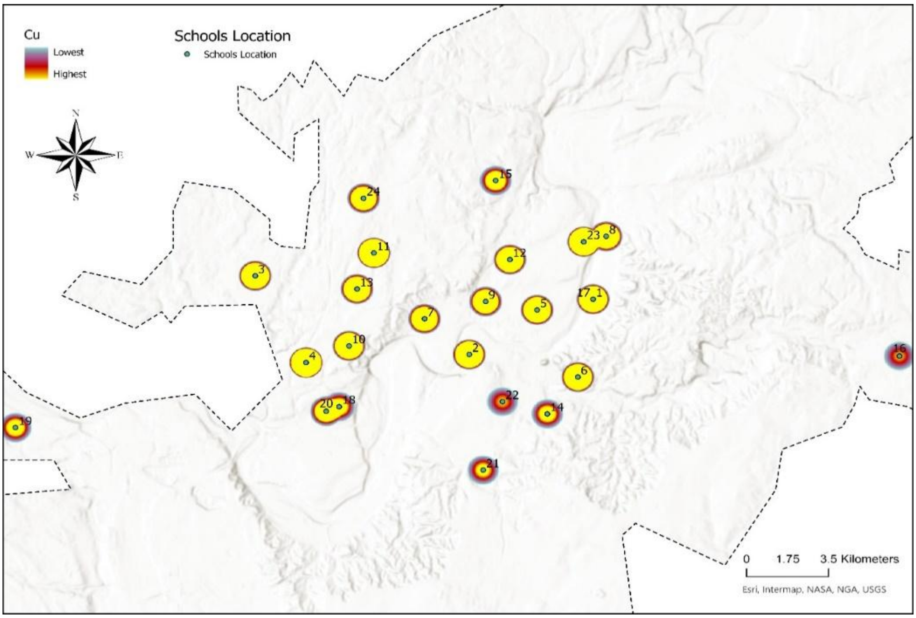
Zhou, L., Liu, G., Shen, M., Liu, Y., Lam, P. K. S., et al. 2021. "Characteristics of indoor dust in an industrial city: Comparison with outdoor dust and atmospheric particulates." Chemosphere 272: 129952. https://doi.org/10.1016/j.chemosphere.2021.129952.

Žiliūtė, L., Laurinavičius, A., & Vaitkus, A. 2010. Investigation into traffic flows on high-intensity streets of Vilnius city. *Transport, 25*(3), 244-251. https://doi.org/10.3846/transport.2010.30

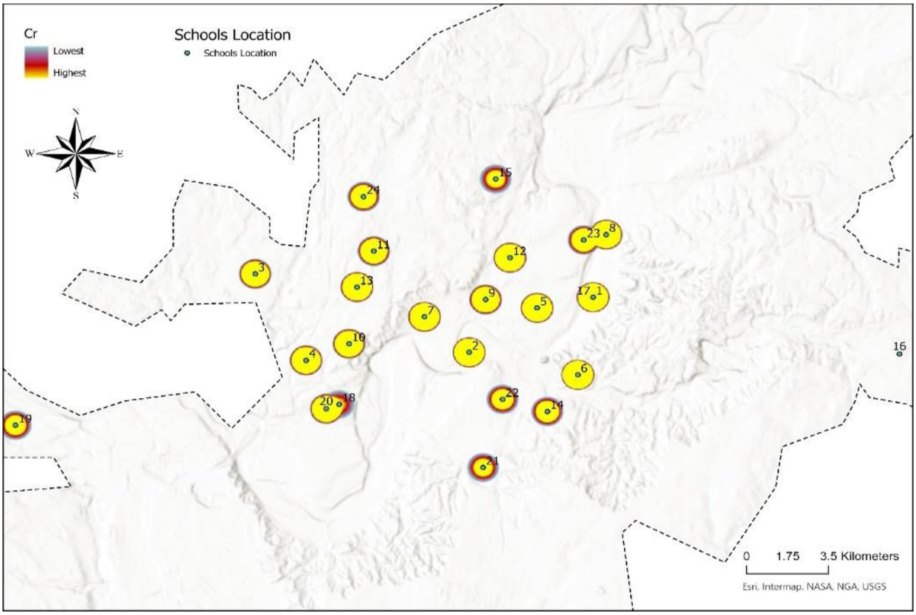
Zulaikhah, S. T., Wahyuwibowo, J., & Pratama, A. A. n.d. "Mercury and its effect on human health: a review of the literature." International Journal of Public Health Science (IJPHS). doi: 10.11591/ijphs.v9i2.20416.



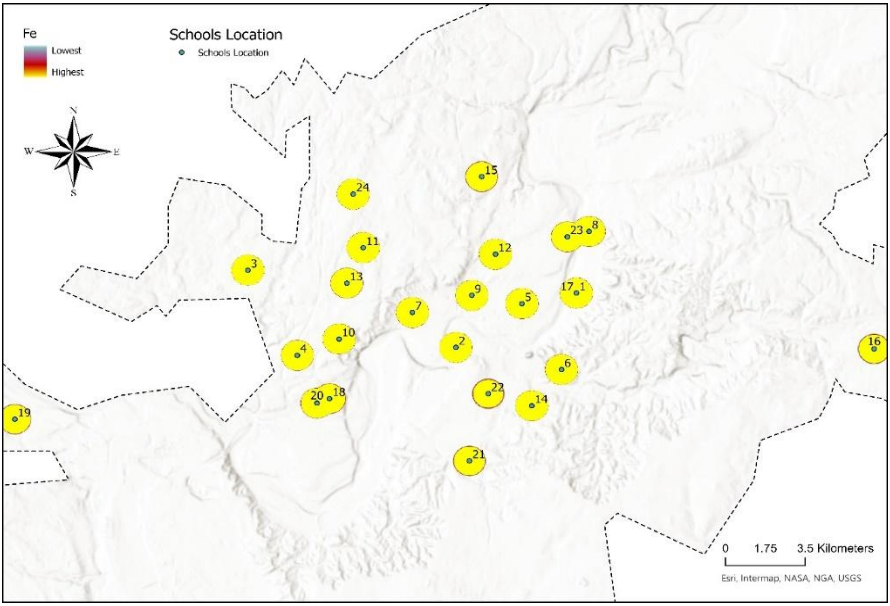
Supplementary Figure 1. Heat map of As.



Supplementary Figure *2*. Cu



Supplementary Figure 3. Cr



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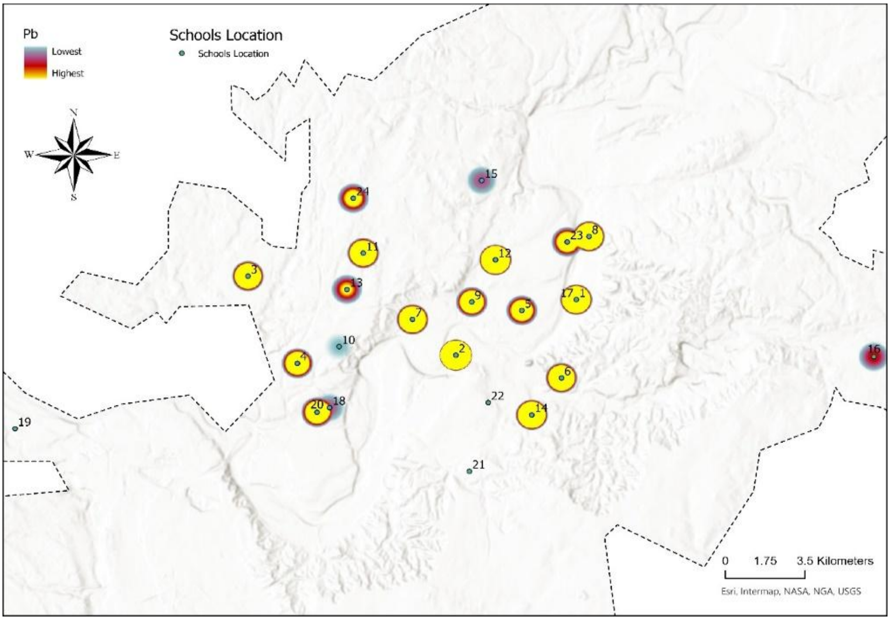
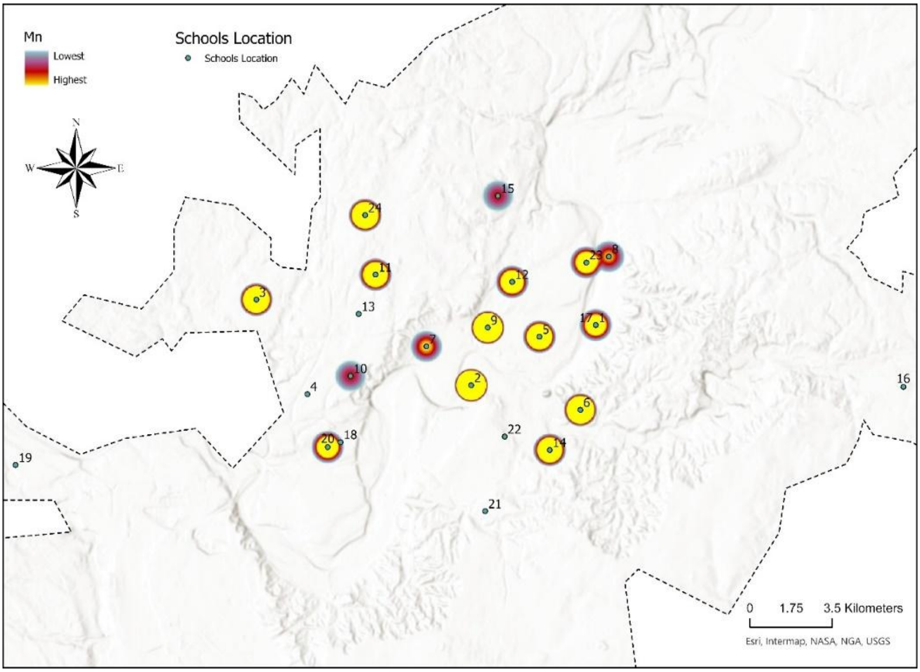
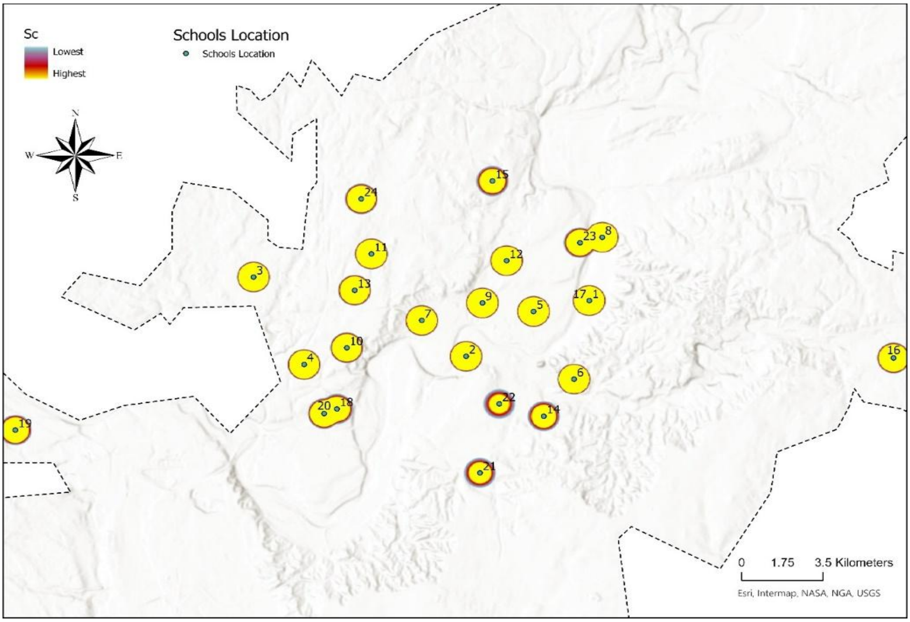


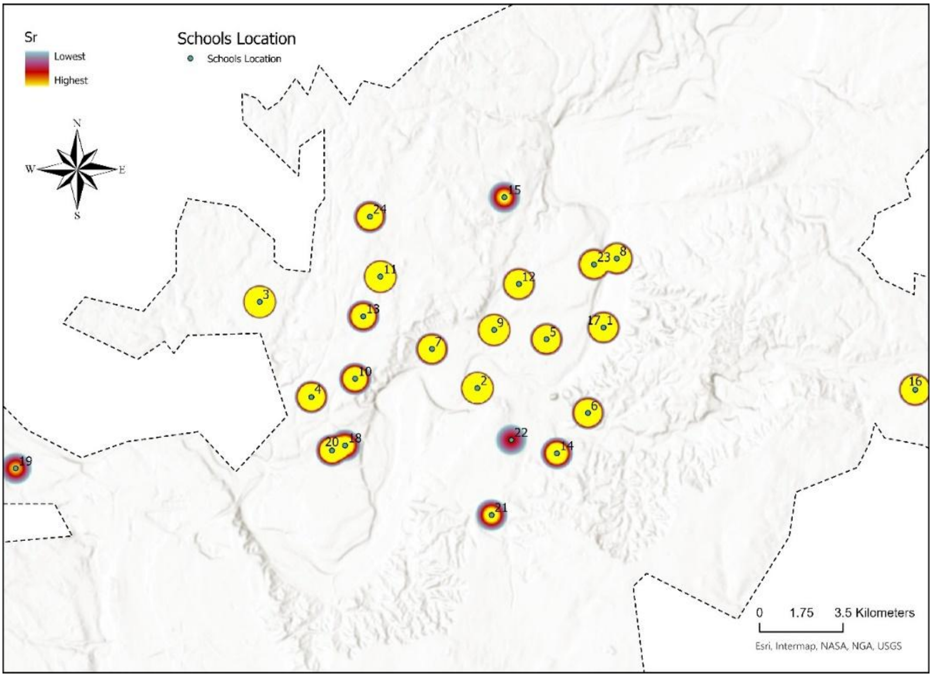
Supplementary Figure 5. Pb



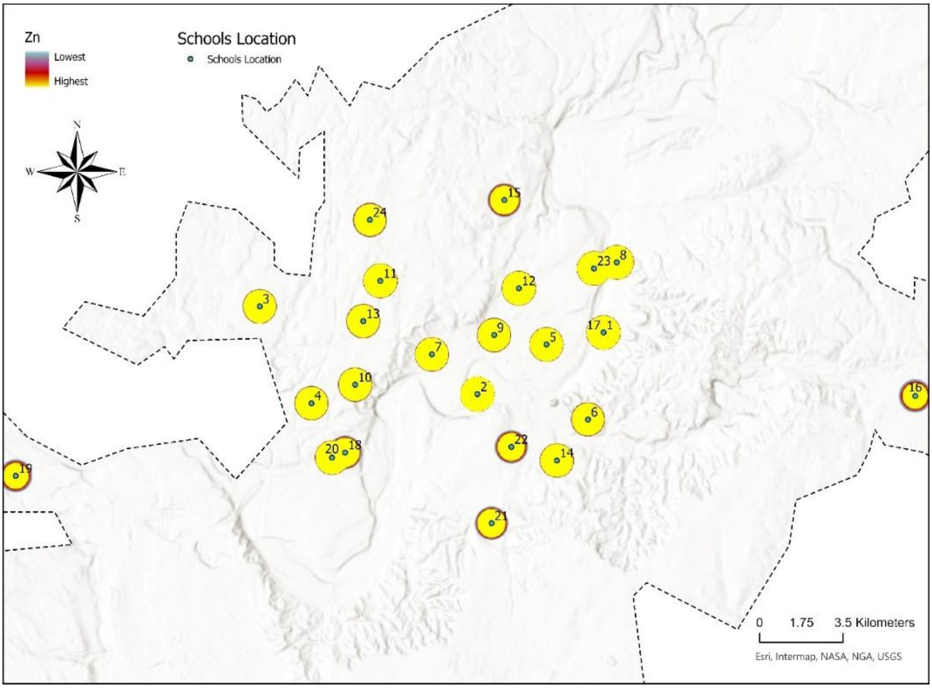
Supplementary Figure 6. Mn



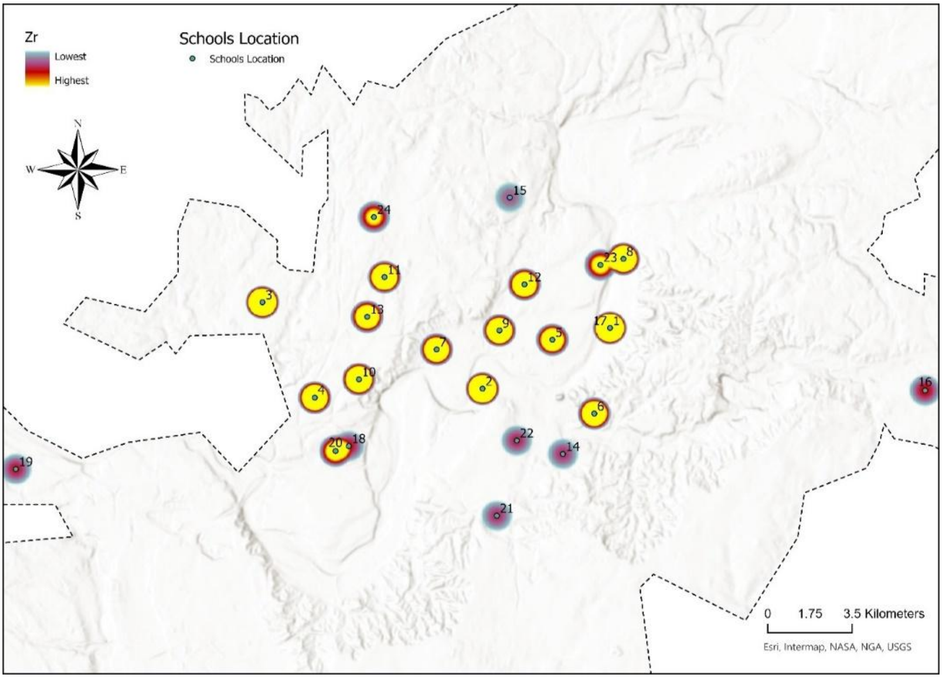
Supplementary Figure 7. Sc



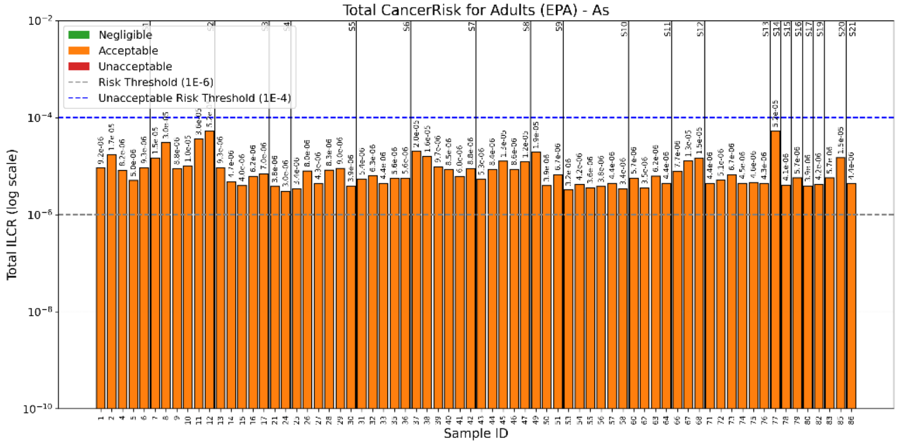
Supplementary Figure 8. Sr



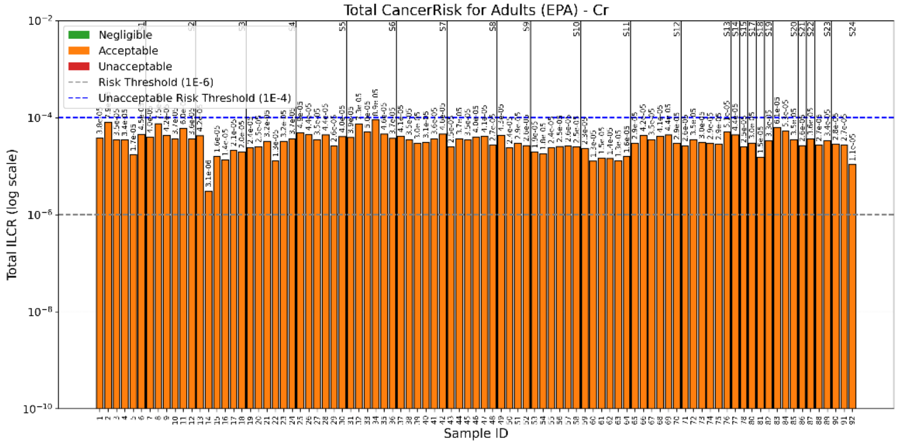
Supplementary Figure 9. Zn



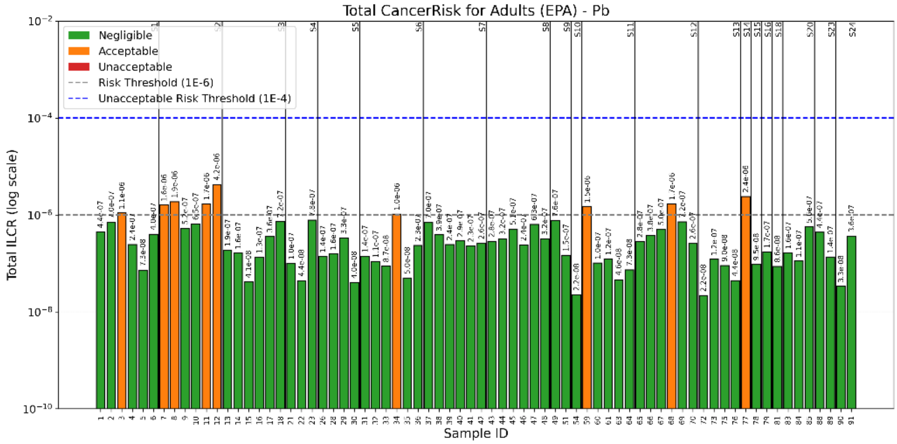
Supplementary Figure 10. Zr



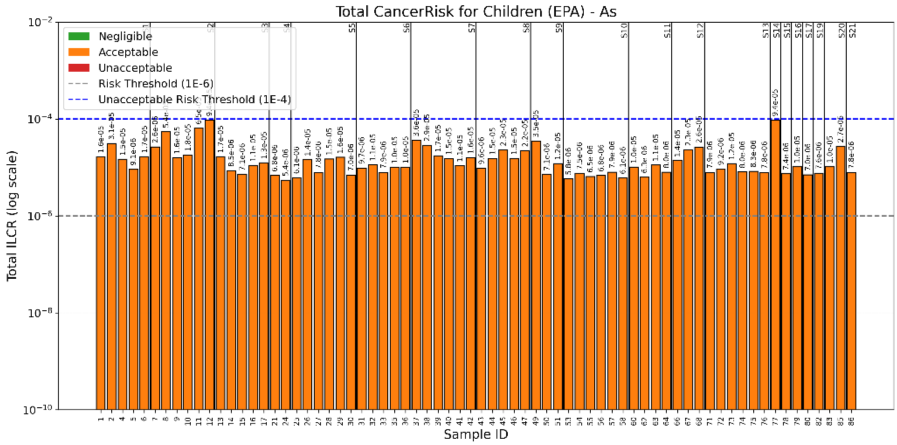
Supplementary Figure 11. Total Cancer Risk Adult EPA As



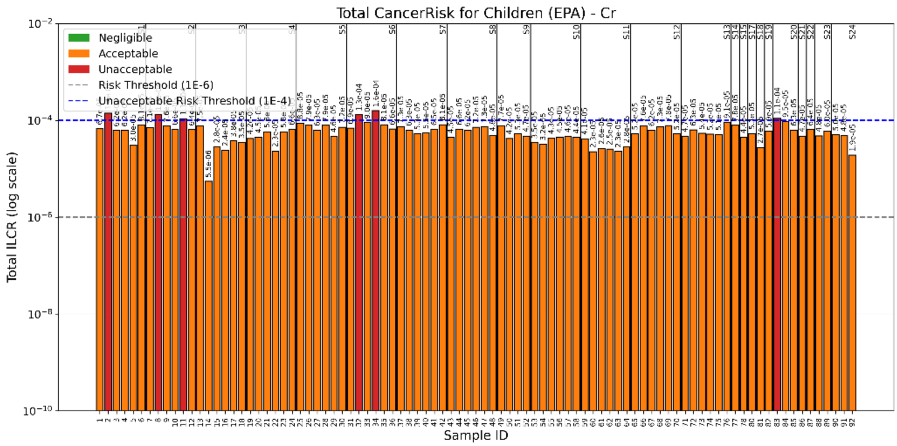
Supplementary Figure 12. Total Cancer Risk Adult EPA Cr



Supplementary Figure 13. Total Cancer Risk Adult EPA Cr



Supplementary Figure 14. Total Cancer Risk Children EPA As



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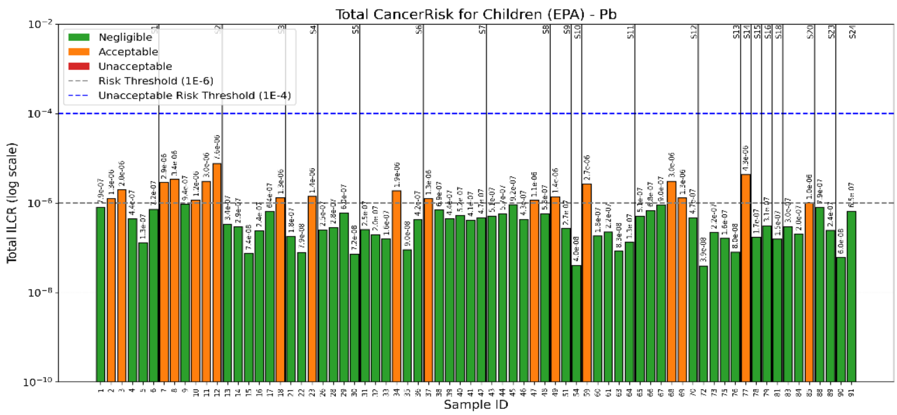
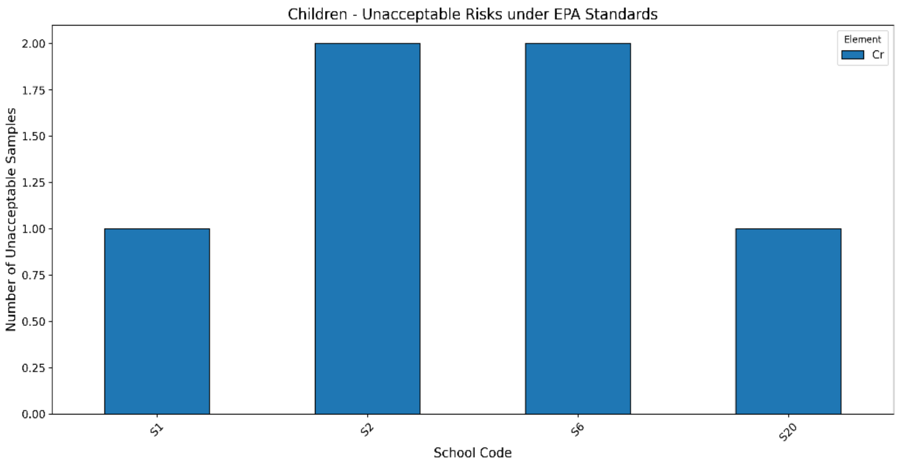
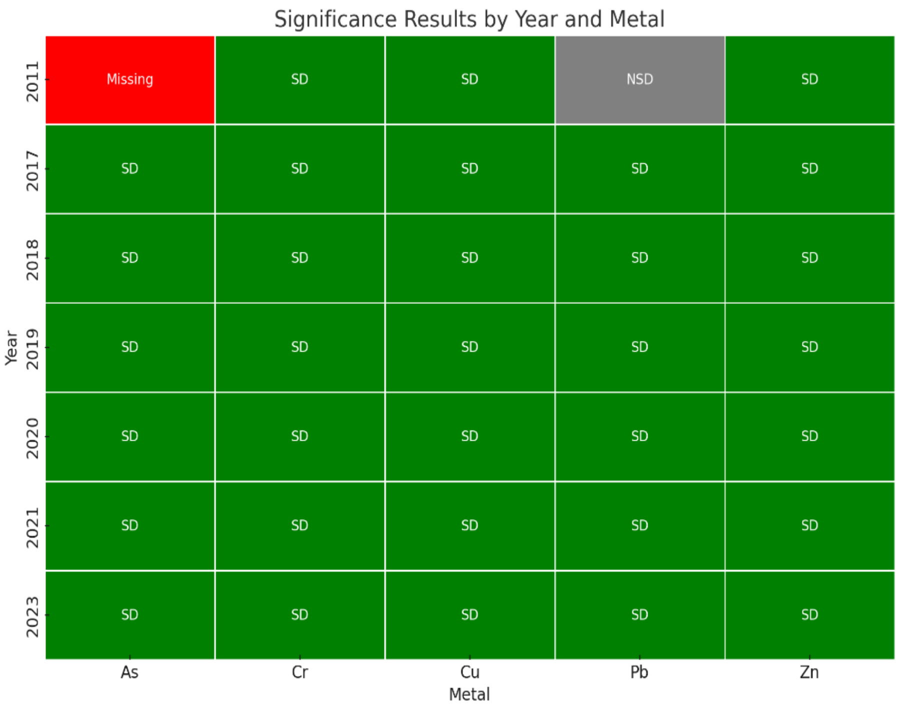


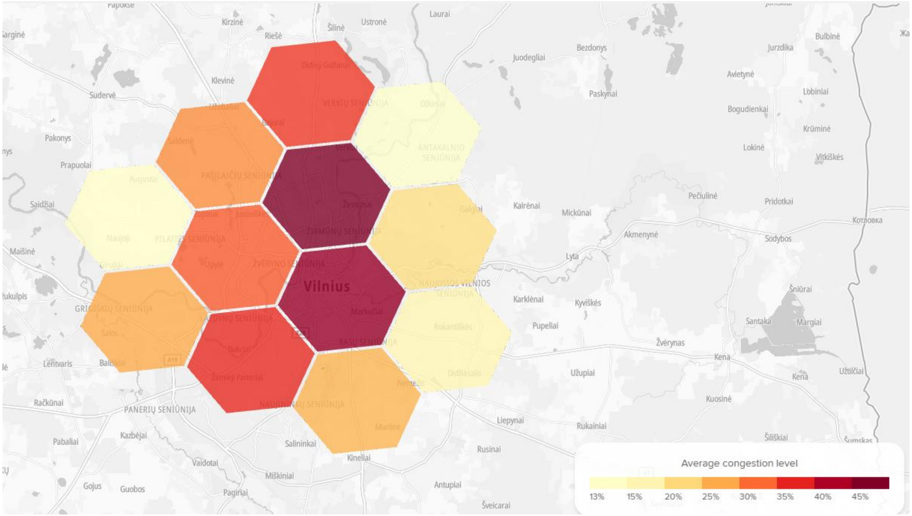
Supplementary Figure 16. Total Cancer Risk Children EPA Pb



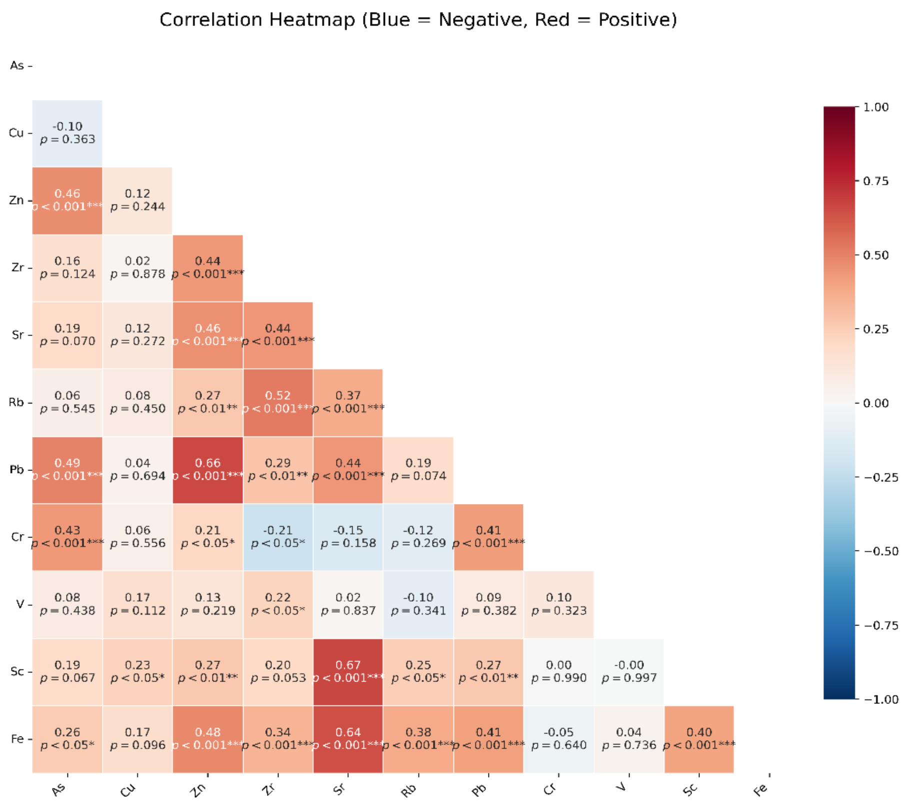
Supplementary Figure 17. EPA Children unacceptable risks in Cr



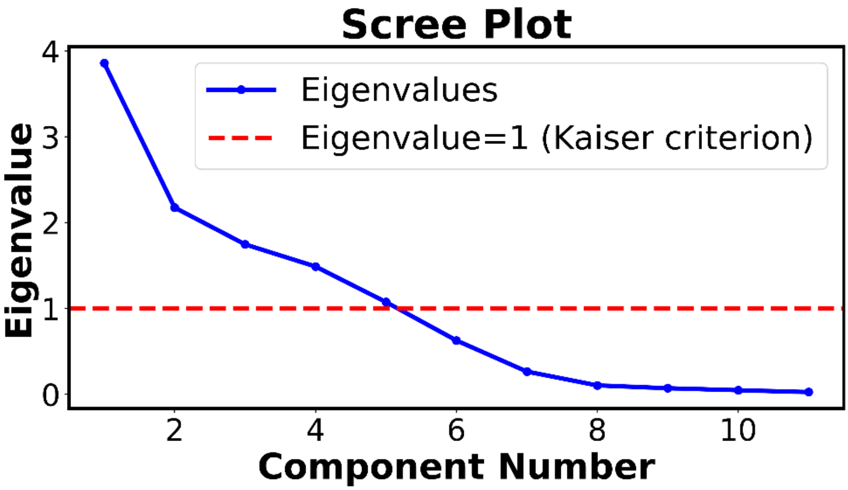
Supplementary Figure 18. Year-by-Year Statistical Comparison of Soil Metal Concentrations vs. a Long-Term Dust Baseline (Dust: 2022–2023, Soil: 2011–2023)



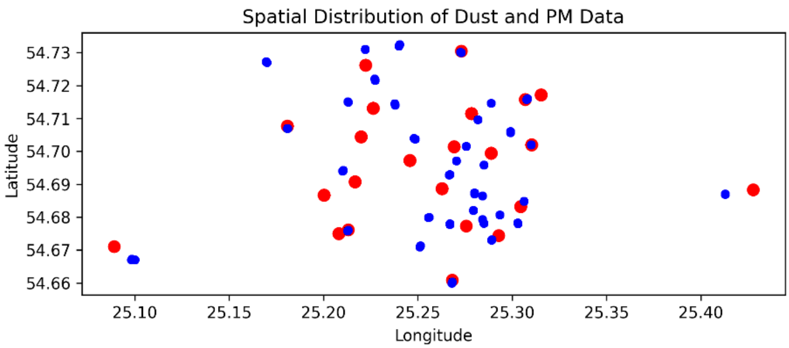
Supplementary Figure 19. Average Congestion Level in Vilnius (TomTom, 2025)



Supplementary Figure 20. Heatmap of the Dust data for Spearman Correlation.



Supplementary Figure 21. Scree Plot for the PCA Cumulative Explained Variance Component



Supplementary Figure 22. Location of PM and Dust samples were taking place.

Supplementary Table 1. PCA and K-Means Clustering Results for Soil and Dust Contaminants (2017-2023)

Year	Contaminant Type	Contaminant Name	PC1	PC2	Cluster
2017	Soil	Pb_soil	7.225286	-2.43743	2
2017	Soil	As_soil	0.980146	-5.76522	2
2017	Soil	Cu_soil	5.008326	2.264711	2
2017	Soil	Cr_soil	1.105171	8.113729	1
2017	Soil	Zn_soil	5.35139	-2.55793	2

2017	Dust	Pb_dust	-6.17037	-2.21164	3
2017	Dust	As_dust	-6.11916	-1.17189	3
2017	Dust	Cu_dust	0.185267	2.037717	1
2017	Dust	Cr_dust	-3.30023	2.917226	1
2017	Dust	Zn_dust	-4.26583	-1.18928	3
2018	Soil	Pb_soil	-4.265	-3.56915	3
2018	Soil	As_soil	-3.25672	-0.07661	3
2018	Soil	Cu_soil	-5.39572	-2.888	3
2018	Soil	Cr_soil	-3.21347	4.806688	2
2018	Soil	Zn_soil	-2.79696	-1.06121	3
2018	Dust	Pb_dust	6.546319	-0.69075	1
2018	Dust	As_dust	6.178116	-1.27159	1
2018	Dust	Cu_dust	-0.33074	8.015435	2
2018	Dust	Cr_dust	1.534113	-3.68626	1
2018	Dust	Zn_dust	5.000047	0.421443	1
2019	Soil	Pb_soil	-4.355	-4.17678	3
2019	Soil	As_soil	-3.09134	-3.96798	3
2019	Soil	Cu_soil	-4.78124	0.956511	3
2019	Soil	Cr_soil	-4.69627	-1.76102	3
2019	Soil	Zn_soil	-2.88645	3.979079	2
2019	Dust	Pb_dust	7.153886	-0.29779	1
2019	Dust	As_dust	7.222278	-0.41392	1
2019	Dust	Cu_dust	-1.34934	7.461009	2
2019	Dust	Cr_dust	3.725804	-3.05405	1
2019	Dust	Zn_dust	3.057662	1.274932	1
2020	Soil	Pb_soil	6.913077	-0.96805	3
2020	Soil	As_soil	1.838454	-0.11027	3
2020	Soil	Cu_soil	5.241012	-1.21768	3
2020	Soil	Cr_soil	2.835056	0.277552	3
2020	Soil	Zn_soil	4.597758	-2.29789	3
2020	Dust	Pb_dust	-5.82771	-2.15789	1
2020	Dust	As_dust	-6.00698	-2.36735	1
2020	Dust	Cu_dust	-0.72546	9.864009	2
2020	Dust	Cr_dust	-3.63517	-1.2118	1
2020	Dust	Zn_dust	-5.23004	0.18936	1
2021	Soil	Pb_soil	-3.39543	8.007009	3
2021	Soil	As_soil	-2.74387	7.426599	3

2021	Soil	Cu_soil	-7.15433	-4.69234	2
2021	Soil	Cr_soil	-7.1928	-4.63819	2
2021	Soil	Zn_soil	-6.26712	-2.17337	2
2021	Dust	Pb_dust	7.17925	-1.05957	1
2021	Dust	As_dust	7.423053	-1.79531	1
2021	Dust	Cu_dust	0.714594	0.864594	1
2021	Dust	Cr_dust	4.729661	-0.16966	1
2021	Dust	Zn_dust	6.706985	-1.76976	1
2023	Soil	Pb_soil	-5.80942	-0.41485	3
2023	Soil	As_soil	-6.04743	-0.9385	3
2023	Soil	Cu_soil	-5.71903	-0.3824	3
2023	Soil	Cr_soil	-3.97906	-1.86233	3
2023	Soil	Zn_soil	-4.28088	-0.74002	3
2023	Dust	Pb_dust	6.058309	-2.77708	1
2023	Dust	As_dust	6.928048	-2.62907	1
2023	Dust	Cu_dust	1.26338	9.589813	2
2023	Dust	Cr_dust	5.079272	-0.74787	1
2023	Dust	Zn_dust	6.506803	0.902299	1

Supplementary Table 2. Nearest Distances (meters)

Code	**nearest_pm_sensor_distance**	**Nearest_road_distance**	**nearest_traffic_light_distance**	**nearest_railway_distance**	**nearest_bus_stop_distance**
S1	34.30783	24.36398	71.52597	3512.877	144.2316
S2	909.0804	18.67521	143.6547	3142.978	73.86588
S3	117.2136	41.86276	128.9232	7414.76	52.94475
S4	1765.67	60.72938	198.6256	5333.676	678.9278
S5	812.6591	45.14866	237.4718	2288.303	359.4715
S6	368.4688	23.67594	44.76388	1655.558	358.6533

S7	1281.219	39.39599	107.0203	5375.329	217.9886
S8	849.841	31.09047	66.89787	6302.556	149.5832
S9	730.1153	45.23178	163.0483	3609.855	290.7374
S10	970.2132	41.6311	229.9543	5345.216	215.8479
S11	1300.016	42.46039	118.7362	8722.864	258.6856
S12	520.1384	28.79222	159.9752	4816.864	299.1319
S13	2169.922	35.601	28.31186	7754.695	169.5692
S14	460.0417	41.02038	84.44447	339.5543	66.07887
S15	93.49866	49.87214	286.1547	8514.529	305.5684
S16	1671.201	0.065116	126.5809	1049.447	421.0629
S17	34.30783	24.36398	71.52597	3512.877	144.2316
S18	23.53767	15.54228	277.1855	3167.146	282.1489
S19	1283.343	38.50405	100.125	1079.819	173.0789
S20	531.7513	25.78871	222.5536	3409.568	300.5707

S21	111.6792	0.11135	338.2863	637.5745	546.2199
S22	962.9919	11.22974	65.36473	1694.513	460.2053
S23	100.4606	12.09553	145.9016	5714.946	196.5407
S24	912.8719	36.60629	210.3259	10672.38	354.5297

Supplementary Table 3. Igeo description for Mean explanation

Element	Mean Igeo	Interpretation	Range
As	1.38±0.99	**Moderately contaminated** (1–2)	0.28–4.22 (up to heavily–extremely)
Cu	2.82±0.68	**Moderately to heavily contaminated** (2–3)	2.08–5.02 (some up to heavily–extremely)
Zn	4.57±1.53	**Heavily to extremely contaminated** (4–5)	2.49–8.69 (max = extremely)
Pb	1.50±1.51	**Moderately contaminated** (1–2)	-2.09–4.65 (some extremely)
Cr	1.45±0.5	**Moderately contaminated** (1–2)	0.43–2.30

Supplementary Table 4.EF and PLI description for Mean explanation

Element	Mean EF	Interpretation	Range
As	45.7±20.24	Severe to Extreme	16.22-77.73
Cu	147.82±120.47	Severe to Extreme	20.05-516.17
Zn	617.37±806.20	Extreme Enrichment	108.05-3425.32
Pb	52.94±36.13	Moderate to Extreme	4.96-152.91
Cr	53.3±34.76	Moderate to Extreme	10.73-143.67

Supplementary Table 5. CF and PLI description for Mean explanation

Element	Mean EF	Interpretation	Range
As	5.29± 5.97	**Considerable to Very high** (1 - 6)	1.82–27.98 (some very high)
Cu	12.20± 8.91	**Very high** (≥ 6)	6.33–48.81
Zn	72.41± 128.85	**Very high** (≥ 6)	8.44–620.44
Pb	7.14± 9.19	**Considerable to Very high** (1 - 6)	0.35–37.62
Cr	4.34±1.41	**Considerable to Very high** (1 - 6)	2.02- 7.37
PLI	**Mean PLI across 24 schools:** 1.82	**Std:** ±0.82	**Min PLI:** 0.865 **Max PLI:** 4.144

Supplementary Table 6. Mean and SD for the PM data.

X	**Y**	**pm1**	**pm2_5**	**pm10**	**pm1SD**	**pm2_5SD**	**pm10SD**
25.0984	**54.66729**	8.87199	14.88644	18.00503	7.089961	11.06497	15.25232
25.16963	**54.72728**	12.35406	19.74492	23.34971	7.721609	12.87017	16.8226
25.18068	**54.70702**	7.964293	12.65051	15.10343	6.324729	8.657615	13.3289
25.21046	**54.69423**	8.573814	13.08918	15.29448	6.133181	7.715084	11.75971
25.21262	**54.67579**	8.164769	12.8841	15.19805	5.432334	6.852293	10.46987
25.21283	**54.71504**	8.388376	12.92234	14.68563	5.472987	6.957972	10.29097
25.22205	**54.73103**	8.360127	14.04869	16.08825	5.995366	10.20575	12.90991
25.22719	**54.72162**	7.669742	11.45	13.15959	5.318189	6.577322	9.888752
25.23774	**54.71449**	9.385224	13.15622	14.70478	5.960086	7.202178	10.63004
25.24033	**54.73241**	9.060533	14.64851	17.77596	6.133746	10.04142	13.96606
25.24845	**54.70372**	7.877286	11.65578	13.18065	5.85814	7.310568	10.69597

25.2514	**54.67135**	10.24792	16.57727	19.76847	6.767832	11.01453	14.64843
25.25557	**54.67989**	9.734284	15.85135	18.41998	6.799727	11.12457	14.50291
25.26661	**54.69287**	8.920821	13.9041	16.31021	6.546422	8.667559	12.70193
25.267	**54.67775**	8.854592	13.27653	15.42408	6.14368	8.054192	11.98572
25.26819	**54.66026**	8.635736	13.34467	15.41447	5.612698	7.495974	11.02951
25.27265	**54.7301**	8.981746	15.08422	18.10521	6.611494	10.53144	14.02552
25.27936	**54.68205**	4.483704	6.614444	7.446667	2.683475	3.810538	4.439101
25.27995	**54.68735**	10.32796	18.3396	22.73142	7.126291	11.4237	16.21441
25.28931	**54.67311**	10.59189	18.38993	22.37079	7.476296	12.91569	17.44006
25.29908	**54.70567**	9.242794	15.15352	18.38695	6.542526	10.28793	13.86608
25.30291	**54.67825**	10.54236	18.12687	22.27712	7.674992	13.10551	18.1315
25.30771	**54.71579**	8.638031	13.4513	15.66684	5.740572	7.735666	11.5173
25.31005	**54.70199**	9.629475	15.69935	18.88911	6.793089	10.9459	14.67361

25.41293	**54.6869**	9.567107	15.11574	17.89711	6.733136	9.895284	14.29718

Supplementary Table 7. PCA loadings and Vector Length of each elements

	PC1	**PC2**	**PC3**	**Vector Length**
As	0.462794	-0.17645	-0.21644	0.540517
Cu	0.079088	0.246156	0.525796	0.585926
Zn	0.421624	0.156523	0.247633	0.513408
Zr	0.178046	0.207359	0.205965	0.342228
Sr	0.08795	0.575532	-0.3411	0.674777
Rb	0.175776	-0.08215	0.203168	0.280931
Pb	0.477242	-0.1258	-0.16868	0.521574
Cr	0.23135	-0.13483	-0.14178	0.30299
V	0.314967	0.37941	0.309062	0.581958
Sc	-0.04299	0.528437	-0.46293	0.703845
Fe	0.386177	-0.21352	-0.23862	0.50166

SANTRAUKA-SUMMARY

TYRIMO TIKSLAS IR UŽDAVINIAI

Tyrimo tikslas:

Ištirti, kaip lauko aplinkos tarša sunkiaisiais metalais (SM), esančiais dirvožemyje ir ore esančiose kietosiose dalelėse, prisideda prie mokyklų vidaus patalpų užterštumo dulkėmis, bei įvertinti šios taršos poveikį vaikų sveikatai.

Tyrimo uždaviniai:

Šiam tikslui pasiekti buvo suformuluoti šie uždaviniai:

Įvertinti SM kiekį patalpų dulkėse, surinktose iš bendrojo lavinimo mokyklų patalpų, siekiant nustatyti užterštumo lygį ir suprasti teršalų kaupimosi dėsningumus.

Išanalizuoti ryšį tarp lauko ore esančių kietųjų dalelių koncentracijos ir SM kiekio patalpų dulkėse.

Ištirti ryšį tarp SM koncentracijos dirvožemyje už mokyklų ribų ir jų koncentracijos vidaus dulkėse, įvertinant dirvožemio užterštumo įtaką patalpų aplinkos kokybei.

Įvertinti galimą riziką moksleivių sveikatai, kurią kelia patalpų dulkėse esantys SM, integruojant poveikio modeliavimą su epidemiologiniais duomenimis ir nustatant nekarcinogeninę bei karcinogeninę riziką.

TYRIMO NAUJUMAS

Pateikiama išsami analizė apie SM koncentracijas mokyklų vidaus dulkėse, suteikiant naujų įžvalgų apie užterštumo lygį ir pasiskirstymo modelius švietimo įstaigose – tai tema, kuri ankstesniuose Lietuvos tyrimuose buvo mažai nagrinėta.

Atskleidžiamas ryšys tarp SM kiekių lauko dirvožemyje ir jų koncentracijos patalpų dulkėse, padedantis geriau suprasti teršalų pernešimo iš lauko į vidaus aplinką mechanizmus.

Parodoma sąsaja tarp lauko oro kietųjų dalelių (KD) koncentracijos ir SM kiekio patalpų dulkėse, kas leidžia suprasti, kaip oro kokybė veikia vidaus aplinką.

Atliktas išsamus sveikatos rizikos vertinimas, kuriame poveikio modeliavimas sujungtas su epidemiologiniais duomenimis. Tyrimas pateikia naujų įžvalgų apie nekarcinogeninę ir karcinogeninę riziką, kurią vaikams kelia SM patalpų dulkėse.

PROBLEMOS

Nepakanka duomenų apie SM koncentracijas ir pasiskirstymo modelius mokyklų patalpų dulkėse, todėl sunku įvertinti taršos lygį ir parengti strategijas vaikų sveikatos apsaugai.

Neaiškūs teršalų pernešimo iš lauko dirvožemio ir oro KD į vidaus patalpas mechanizmai, todėl sudėtinga nustatyti intervencijos taškus taršai sumažinti.

Trūksta kompleksinių sveikatos rizikos vertinimų, kurie apjungtų poveikio modeliavimą ir epidemiologinius duomenis, siekiant įvertinti SM keliamos rizikos mastą.

GINAMI PAREIŠKIMAI

SM koncentracijos mokyklų patalpų dulkėse rodo reikšmingą užterštumo lygį.

Egzistuoja ryšys tarp SM (pvz., As, Cu, Zn, Pb) koncentracijos lauko dirvožemyje ir jų kiekio vidaus dulkėse.

Nustatytas ryšys tarp lauko oro kietųjų dalelių koncentracijos ir SM koncentracijos patalpų dulkėse.

Vaikai, besimokantys mokyklose, yra veikiami SM esančių patalpų dulkėse, kas gali sukelti tiek nekarcinogeninį, tiek karcinogeninį poveikį jų sveikatai.

ĮVADAS

Ši disertacija nagrinėja sunkiųjų metalų (SM) taršą mokyklų vidaus dulkėse ir jos galimą poveikį vaikų sveikatai. Pasaulyje vis daugiau dėmesio skiriama miesto dulkėtumo, kaip svarbaus taršos šaltinio, tyrimams, tačiau Lietuvoje, ypač švietimo įstaigose, šis klausimas lieka menkai ištirtas. Vidaus dulkės – tai kietųjų dalelių mišinys, kuriame aptinkami sunkieji metalai iš įvairių šaltinių: išorinių (oro tarša, dirvožemis) ir vidinių (statybinės medžiagos, buitinė veikla). Vaikai, ypač dėl savo elgsenos ir fiziologinių ypatybių, yra itin jautrūs šių teršalų poveikiui.

Šio tyrimo tikslas – nustatyti sunkiųjų metalų koncentracijas Vilniaus mokyklų vidaus dulkėse, įvertinti jų ryšį su išoriniais aplinkos šaltiniais (pvz., dirvožemiu, ore esančiomis kietosiomis dalelėmis) bei atlikti sveikatos rizikos analizę, orientuotą į vaikų grupes. Tyrimas išsiskiria tuo, kad pirmą kartą Lietuvoje taikoma kompleksinė metodika, apjungianti aplinkosauginius, statistinius ir rizikos vertinimo aspektus. Šis darbas prisideda prie aplinkos sveikatos tyrimų plėtros Lietuvoje, atkreipdamas dėmesį į mokyklų vidaus aplinkos kokybę ir jos reikšmę vaikų gerovei.

MEDŽIAGOS IR METODAI

Tyrimas buvo atliktas Vilniaus mieste, kuris dėl savo dydžio, klimato sąlygų ir urbanistinės struktūros yra tinkamas analizuoti miesto dulkių ir dirvožemio užterštumą sunkiaisiais metalais (SM). Duomenys buvo renkami iš 24 bendrojo lavinimo mokyklų, atrinktų pagal jų geografinę padėtį, pastatymo metus, artumą taršos šaltiniams bei renovacijos istoriją. Mėginiai imti 2022–2023 m., iš vietų, kurios dažniausiai nėra reguliariai valomos (pvz., už radiatorių, aukštų lentynų, sporto salių kampų), siekiant surinkti ilgai susikaupusias dulkes.

Dulkių mėginiai buvo laikomi sterilizuotuose konteineriuose ir analizuoti naudojant rentgeno fluorescencinės spektrometrijos (XRF) metodą su „Niton XL2" ir „SPECTRO XEPOS" įranga. Tyrime taip pat naudoti anksčiau surinkti viršutinio dirvožemio duomenys iš Lietuvos geologijos tarnybos (Kadūnas ir kt., 1999; Kumpienė ir kt., 2011), leidžiantys įvertinti fonines teršalų vertes bei istorinį užterštumą.

Buvo taikomi šie taršos vertinimo indeksai:

- Geoakumuliacijos indeksas (Igeo) – leidžiantis įvertinti metalo kaupimosi laipsnį palyginti su fono verte.
- Užterštumo faktorius (CF) – nustatantis kiek kartų SM koncentracija viršija foninę normą.
- PLI (Taršos apkrovos indeksas), mCd (modifikuotas užterštumo laipsnis) ir EF (praturtėjimo koeficientas) – bendram ir antropogeniniam taršos lygiui įvertinti.

Sveikatos rizika buvo vertinama remiantis JAV EPA gairėmis, atskirai modeliuojant suaugusiųjų ir vaikų poveikį per keturis kelius: nurijimą, įkvėpimą, dermalinį (odos) poveikį ir garų įkvėpimą (pvz., Hg). Buvo apskaičiuoti pavojaus koeficientai (HQ), pavojaus indeksas (HI) ir kancerogeninės rizikos rodikliai, remiantis referencinėmis dozėmis ir poveikio faktoriais.

Statistinė analizė atlikta naudojant Python (versija 3.12.1), taikant tokias bibliotekas kaip pandas, numpy, scipy, statsmodels, scikit-posthocs. Prieš lyginamuosius testus (t-testas, Mann-Whitney U) atliktas normalumo įvertinimas (Shapiro-Wilk testas). Rezultatų stiprumas vertintas naudojant Coheno d efektų dydžius.

Erdvinė analizė atlikta naudojant ArcGIS Pro 3.0. Sukurti užterštumo žemėlapiai, vizualizuojant SM pasiskirstymą, naudojant IDW interpoliaciją dirvožemio ir oro duomenims, o šilumos žemėlapius dulkių duomenims.

Naudojant GeoPandas ir KDTree, buvo įvertinti atstumai nuo mokyklų iki potencialių taršos šaltinių (pvz., keliai, PM jutikliai, geležinkeliai, autobusų stotelės).

Taršos šaltiniams identifikuoti naudotas Positive Matrix Factorization (PMF) modelis (EPA PMF 5.0), kuris suskaidė stebėtas koncentracijas į šaltinių profilius bei jų įtaką. Modelio patikimumas įvertintas naudojant „bootstrap“ metodą. Taip pat naudoti PCA (pagrindinių komponentų analizė) ir hierarchinė klasterizacija (HCA), leidžiantys atskleisti duomenų struktūras bei identifikuoti dominančius veiksnius ir taršos klasterius.

Šis metodų derinys užtikrino išsamią ir daugiapakopę dulkių užterštumo vertinimo analizę bei sudarė pagrindą mokyklų vidaus aplinkos rizikų modeliavimui.

REZULTATAI IR DISKUSIJA

Sunkiųjų metalų taršos vertinimas dulkėse Vilniaus mokyklose

Analizuojant dulkių mėginius iš 24 Vilniaus mokyklų, nustatyta, kad sunkiųjų metalų (As, Cu, Zn, Pb) koncentracijos labai skiriasi. Šios medžiagos gali kilti iš eismo, statybų ir atmosferinių teršalų. Ypač variavo Cu koncentracijos (nuo 51,28 mg/kg iki 395,37 mg/kg), o As buvo gana vienoda, kas rodo, jog Cu priklauso nuo vietinių šaltinių, o As – nuo foninės taršos.

Vizualizacija (4 pav.) parodė skirtingus elementų (As, Cu, Zn, Pb, Cr) pasiskirstymo modelius, rodančius įvairių taršos šaltinių įtaką. Lentelė 8 pateikia lyginamąją analizę su kitomis šalimis: Vilniuje As siekė iki 69,96 mg/kg, kai tuo tarpu, pavyzdžiui, Pietų Afrikoje vos 0,78 mg/kg.

Taršos įvertinimas

5 pav. matomas Taršos koeficientas (CF) rodo labai aukštą Zn taršą (CF iki 27,98). 6 pav. modifikuotas CF išryškino didelę taršą S2, S14 ir S23 mokyklose (maksimali mCF reikšmė – 63,73). 7 pav. Taršos krūvio indeksas (PLI) patvirtino šiuos rezultatus.

Lentelė 7 pateikia vidutines sunkiųjų metalų koncentracijas kiekvienoje mokykloje. Lentelė 8 pateikia tarptautinius duomenis, parodant, kad kai kurių elementų lygiai Vilniuje viršija pasaulinius vidurkius.

Praturtinimo ir geoakumuliacijos analizės

Praturtinimo faktoriai (EF, 8 pav.) parodė, kad Cu, Zn ir Sc yra ypač stipriai praturtinti, o Zr ir Rb – mažai. Geoakumuliacijos indeksas (Igeo, 9 pav.) parodė labai aukštą Zn taršą (Igeo > 5) keliose mokyklose. S23 ir S16 rodė ekstremalų Cu ir Sc užterštumą.

Pagrindinių komponentų analizė (PCA)

Pirmi du komponentai paaiškino 52,57 % visos variacijos. PC1 buvo susijęs su antropogeniniais šaltiniais (As, Pb, Zn), PC2 – su geogeniniais (Sr, Sc). Ši analizė padėjo identifikuoti galimus taršos šaltinius.

HCA – Hierarchinė klasterių analizė

Duomenys išskirti į tris klasterius:

- Klasteris 1: Sr, Sc – geogeniniai šaltiniai
- Klasteris 2: As, Pb, Cr, Fe – antropogeniniai šaltiniai
- Klasteris 3: Cu, Zn, Zr, Rb, V – urbanistiniai šaltiniai

Geografinis mokyklų išsidėstymas taip pat rodo taršos profilių įvairovę – dalis mokyklų patiria didesnę riziką dėl tam tikrų metalų.

Spearmano koreliacinė analizė

Koreliacijos tarp As, Zn, Pb, Cr ir Fe atskleidė bendrus taršos šaltinius, ypač transportą ir natūralų dulkių pakėlimą. Cu turėjo silpnas koreliacijas, kas rodo vietinių šaltinių įtaką (pvz., pastatų korozija). Zr ir Rb turėjo stiprią tarpusavio koreliaciją, rodydami natūralų kilmės šaltinį.

Teigiamai riboto veiksnio analizė (PMF)

Šeši identifikuoti taršos šaltiniai (veiksniai):

- F1: Natūralios kilmės dulkės (aukštas Sr, Sc)
- F2: Eismas ir natūralūs šaltiniai (aukštas Fe, vidutinis Cu, Cr)
- F3: Padangų dilimas / statybinės veiklos (aukštas Zn)
- F4: Istorinė tarša (Pb, As – iš švino dažų, buvusio kuro)
- F5: Natūrali dirvožemio dulkė (Zr, Rb)
- F6: Eismo tarša be Pb (aukštas Cu, Cr – stabdžių nusidėvėjimas)

Ribotumai

Tyrimas aprėpė 24 mokyklas – tai ne visos Vilniaus ugdymo įstaigos. Naudotas vienkartinis mėginių ėmimas negalėjo įvertinti sezoninių ar vidinių pokyčių. Be to, buvo naudojami apytiksliai sveikatos rizikos modeliai, kurie ne visada atspindi realią situaciją. Būtina papildoma analizė, įskaitant sezoninius pokyčius ir didesnį imties dydį.

Lauko kietųjų dalelių poveikio vidaus dulkių užterštumui sunkiaisiais metalais vertinimas

PM2.5/PM10 santykis

15–17 pav. pateikiami Vilniaus miesto PM2.5/PM10 santykio pokyčiai parodė laiko tendencijas: mažas santykis rodo stambiųjų dalelių (pvz., dulkės, žiedadulkės) dominavimą, o didelis – smulkiųjų, susijusių su žmogaus veikla (pvz., transportas, deginimas). Didesnis nei 0,5 santykis laikomas pavojingu sveikatai, nes rodo prastą oro kokybę.

Dėl to, kad PM2.5 lengviau patenka į vidų, didėjantis PM2.5/PM10 santykis rodo, kad smulkios pavojingos dalelės vis labiau prisideda prie dulkių taršos mokyklose, ypač netoli kelių ar pramoninių zonų.

Aplinkos artumas prie taršos šaltinių (19 pav.)

- Artumas prie PM jutiklių: Atstumas nuo mokyklų iki artimiausio PM2.5 jutiklio svyravo nuo 23,5 m (S18) iki 2170 m (S13), su mediana ~730 m. Mokyklos <500 m nuo jutiklio (pvz., S1, S3, S6, S13, S14) turėjo tikslesnį lauko oro kokybės atspindį.
- Artumas prie kelių: Mokyklos buvo nuo 0,07 m (S16) iki 60,7 m (S4) nuo artimiausio kelio. Vidutinė vertė – 31 m. Mokyklos <10 m atstumu (S16, S21) gali patirti stiprią transporto taršą, įskaitant Zn, Cu ir Pb.
- Artumas prie šviesoforų: Atstumai nuo 28,3 m (S13) iki 338,3 m (S21), medianinis – 145 m. Mokyklos arčiau nei 50 m (pvz., S6, S8, S16, S22) jautrios padidėjusiai taršai dėl dažnų stabdymo/greitėjimo ciklų.
- Artumas prie geležinkelių: Atstumai nuo 339,6 m (S14) iki 10 672 m (S24), su mediana 3610 m. Mokyklos <1000 m (pvz., S14, S16) gali būti paveiktos traukinių emisijų (Fe, Cu).
- Artumas prie autobusų stotelių: Atstumai nuo 52,9 m (S3) iki 678,9 m (S4), mediana – 215 m. Mokyklos <100 m (S3, S6, S7, S8 ir kt.) labiau veikiamos vietinių autobusų emisijų.

Lietuvoje transporto emisijos mažėja, tačiau padangų/stabdžių dilimas išaugo 83 % nuo 2005 m. Automobilių skaičius beveik padvigubėjo 1999–2010 m., o dideli sunkvežimiai stiprina dangos dėvėjimą ir dulkių kėlimą, ypač tokiose gatvėse kaip Geležinio Vilko ir Savanorių prospektas.

Koreliacijos analizė tarp PM ir dulkių metalų (20 pav.)

Nustatyti statistiškai reikšmingi, nors ir silpni–vidutiniai, ryšiai ($p < 0,05$) tarp PM1, PM2.5, PM10 ir dulkių As, Cu, Zn, Pb, Cr koncentracijų (koreliacijos koeficientai ~0,14–0,22). Šie rezultatai rodo nuoseklią nelinearinę priklausomybę tarp PM ir dulkių sudėties, ypač Cu, Zn, Pb ir As. Cr reikšmingo ryšio nerodė, kas leidžia manyti, kad egzistuoja papildomi šaltiniai (pvz., pastatų medžiagos ar vidinė veikla).

Dulkių ir paviršinio dirvožemio užterštumo koreliacija Vilniaus mokyklose

Dirvožemio užterštumas

Remiantis Kumpienės ir kt. (2011) bei DGE Baltic Soil Monitoring (2021, 2023) duomenimis, sunkiųjų metalų koncentracijos Vilniaus mokyklų dirvožemyje labai skiriasi. Labiausiai išsiskyrė Zn ir Pb – jų koncentracijos dažnai viršijo leistinas ribas. Zn kiekiai keitėsi per metus, ypač miesto zonose, paveiktose eismo ir pramonės. Cu ir Cr buvo stabilesni, tačiau kartais fiksuoti vietiniai šuoliai, o As – nežymiai, bet taip pat svyravo.

Kadūno ir kt. (1999) geocheminis žemėlapis parodė, kad dirvožemio tipas, aukštis ir atstumas iki miesto infrastruktūros daro reikšmingą įtaką metalų pasiskirstymui.

Statistiniai palyginimai: dirvožemis vs. dulkės

Pasitelkus Mann-Whitney U ir Welch t-testus, buvo įvertinta, ar dulkių mėginiuose esančių metalų koncentracijos statistiškai skiriasi nuo dirvožemio:

- As: Dulkėse nuo 2017 m. reikšmingai didesnės As koncentracijos – rodo aktyvų kaupimąsi iš atmosferos.
- Cu: Visais metais nuo 2011 m. dulkėse reikšmingai daugiau Cu nei dirvožemyje.
- Zn: Ženkliai didesnės koncentracijos dulkėse – siejama su padangų nusidėvėjimu, pramone, cinkuotais paviršiais.
- Pb: Nuo 2017 m. pradėjo reikšmingai dominuoti dulkėse.
- Cr: Visais metais Cr reikšmingai daugiau dulkėse, tikėtina dėl paviršių korozijos.

Tokie skirtumai aiškinami tuo, kad dulkės – smulkesnės, turinčios didesnį paviršiaus plotą, organinių medžiagų ir katijonų mainų gebą, dėl ko geriau adsorbuoja metalus (Gunawardana et al., 2014; Cao et al., 2012).

Koreliacijos su dulkėmis (Spearmano analizė, 22 pav.)

- Zn: Reikšminga teigiama koreliacija 2017 m. ($\rho = 0.50$), 2018 m. ($\rho = 0.26$), 2020 m. ($\rho = 0.24$) – rodo nuoseklų kaupimąsi iš tų pačių šaltinių.
- Pb: 2011 m. buvo vidutinė koreliacija ($\rho = 0.48$), rodanti bendrą kilmę tais metais.
- As ir Cr: 2019 m. reikšminga koreliacija (As: $\rho = 0.30$, Cr: $\rho = 0.31$), silpnesni ryšiai 2018–2020 m.

Tačiau daugeliu atvejų koreliacijos tarp dulkių ir dirvožemio metalų nebuvo reikšmingos. Galimos priežastys:

Skirtingi vietiniai veiksniai (pvz., pramonė, žemės ūkis), Metalų savybės (mobilumas, tirpumas), Aplinkos sąlygos (krituliai, vėjas), Laiko kaita (pvz., istorinė dirvožemio tarša prieš atmosferinį kaupimąsi dulkėse).

Užterštumo kontekstas

- Pb pasižymi gebėjimu keliauti dideliais atstumais oru – tai svarbus vietinės ir globalios taršos šaltinis (Li et al., 2016).
- Istorinė tarša (švino dažai, buvusios veiklos) vis dar turi poveikį dulkių ir dirvožemio taršai.
- Meteorologija, ypač Baltijos jūros poveikis, reikšmingai lemia teršalų sklaidą.
- Karinių veiklų poveikis taip pat turi būti vertinamas (Vasarevičius ir kt., 2004).
- Cu koncentracijos prie autobusų stotelių ir geležinkelio siejamos su ilgalaikiu Cu naudojimu stabdžių detalėse.

Dirvožemio duomenų apžvalga

Lentelė 9 rodo, kad As, Cr, Cu, Zn ir Pb koncentracijos dirvožemyje labai svyravo tarp 1999 ir 2023 m. – dėl transporto, statybų, netinkamos atliekų tvarkos ir buvusios žemės ūkio praktikos.

Lentelė 10 apibendrina statistiką: visais metais (išskyrus Pb 2011 m.) dulkėse koncentracijos buvo reikšmingai didesnės nei dirvožemyje ($p < 0.05$).

Sunkiųjų metalų poveikio sveikatai vertinimas dulkių mėginiuose iš Vilniaus mokyklų

Pavojingumo indeksas (Hazard Index)

23 pav. rodo bendrą pavojingumo indeksą (HI) suaugusiems ir vaikams. Vertė virš 1 reiškia nekarcinogeninį pavojų sveikatai.

Suaugusiems (grafikas A): visos HI reikšmės < 1 – nėra nekarcinogeninės rizikos. Vaikams (grafikas B): 12 mokyklų viršija As HI > 1, 9 mokyklos – Pb HI > 1, viena – Zn ir V HI > 1. Tai reiškia didelį pavojų vaikų sveikatai.

Karcinogeninės rizikos vertinimas

Pagal EPA rekomendacijas:

Suaugusiems: As ir Cr viršija saugias ribas, Pb – arti ribinės reikšmės. Vaikams: As ir Pb – nesaugūs, Cr – daugumoje ribų, bet kai kur viršija slenkstį. Reikalinga papildoma sveikatos institucijų analizė.

Dideli metalų kiekiai (ypač As ir Pb) kelia didelį pavojų vaikų sveikatai. Jie gali būti siejami su:

Artumu keliams, pramonei, Seno statybos laikotarpio mokyklomis (dėl statybinių medžiagų, kurios gali turėti Zn, Cu), Aplinkos sąlygomis – vėjais, dirvožemio pakėlimu (24–25 pav. rodo didesnius As ir Pb kiekius dirvožemyje nei mokyklose).

Sveikatos pasekmės

Vaikai yra ypač pažeidžiami sunkiųjų metalų poveikiui:

- Trumpalaikės pasekmės: elgesio pokyčiai, kvėpavimo problemos, anemija
- Ilgalaikės pasekmės: neurologiniai sutrikimai, IQ mažėjimas, vėžiniai susirgimai, inkstų pažeidimai
- Xenobiotiniai metalai (As, Pb, Cr, Cu) gali sukelti įvairių sistemų (nervų, širdies, reprodukcinės) pažeidimus.

Kai kurie metalai (ypač Pb) dažnai pernešami oru dideliais atstumais. Jų šaltiniai – transportas, dažai, alyva, bėgių nusidėvėjimas. Šie šaltiniai buvo įtariami, bet tyrime nebuvo tiesiogiai išmatuoti – todėl būtini papildomi tyrimai su oro kokybės ir dirvožemio analizėmis.

Mokyklų lokacijos netoli kelių, traukinių stotelių ar komercinių zonų kelia papildomą riziką dėl ilgalaikio kaupimosi pastatuose.

IŠVADOS

- Patalpų dulkėse Vilniaus mokyklose sunkiųjų metalų koncentracija labai svyravo: Pb lygiai siekė nuo 5,3 ± 3,99 mg/kg iki 564,25 ± 16,81 mg/kg. Nepaisant šios variacijos, visi taršos rodikliai (pvz., praturtinimo koeficientai ir geoakumuliacijos indeksas) buvo vienodai aukšti. Be to, vidutinis taršos apkrovos indeksas (Pollution Load Index) buvo 1,82 ± 0,82, rodantis blogėjančią aplinkos kokybę.
- Nesupervizinio mokymosi modeliai, tokie kaip PCA, HCA ir K-Means klasterizavimas kartu su koreliacijos analize atskleidė skirtingas elementų grupes patalpų dulkėse: PC1 apėmė As, Pb, Cr ir Fe. Šių elementų statistiškai reikšmingi pakrovimai į PC1 kartu su jų žinomais ryšiais su transporto priemonių emisijomis rodo antropogeninę įtaką. PC2 susidėjo iš elementų, tokių kaip Sr ir Sc, kas gali reikšti natūralių, litogeninių medžiagų dominavimą. PC3 pasižymėjo elementais, tokiais kaip Cu, Zn, Zr, Rb ir V, kurie siejami su mišriomis natūralių ir antropogeninių veiksnių įtakomis, skirtingomis nuo PC1.

- PM2,5/PM10 santykis vidutiniškai siekė 0,74, su pavieniais šuoliais viršijančiais 2,7, kas rodo reikšmingą smulkiosios kietosios dalelės patekimo į patalpas lygį. Atstuminės koreliacijos analizės tarp PM1, PM2,5 ir PM10 ir patalpų dulkių As, Cu, Zn, Pb bei Cr koncentracijų parodė nuolat teigiamus koeficientus nuo 0,14 iki 0,22 (visi $p < 0,05$), nurodančius silpną, tačiau statistiškai reikšmingą ryšį – tai reiškia, kad išorinės emisijos, tikėtina susijusios su eismu ir miesto veikla, prasiskverbia į patalpas.
- Ilgalaikė stebėsena rodo, kad patalpų dulkės kaupia sunkiuosius metalus intensyviau nei aplinkinis dirvožemis. Nors dirvožemio metalo lygiai kinta laikui bėgant, patalpų dulkėse nuolat nustatomi daug didesni As, Cu, Zn, Pb ir Cr kiekiai. Statistinė ir koreliacinė analizė tarp dirvožemio ir dulkių taršos rodo bendrus taršos šaltinius, o patalpų dulkės tarnauja kaip labiau koncentruotas šių teršalų rezervuaras. Sveikatos rizikos vertinimas, naudojant bendrą pavojingumo indeksą (Total Hazard Index, THI), rodo, kad nekarcinogeniniai pavojai skiriasi priklausomai nuo elemento ir populiacijos: 50 % mokyklų As lygiai viršija saugias ribas (THI > 1, iki 7,1), o 37,5 % mokyklų – Pb lygiai, siekiantys iki 7,55. Vadovaujantis EPA gairėmis, suaugusiųjų kancerogeninis pavojus visose mokyklose yra priimtinas arba nereikšmingas. Vaikams daugumoje mokyklų rizika yra saugi, tačiau 16,7 % mokyklų rizika viršija leistiną lygį, kas rodo, jog vaikai yra labiau pažeidžiami.

LIST OF SCIENTIFIC PUBLICATIONS

Unsal, M.H., Ignatavičius, G. & Valskys, V. 2024. "Unveiling Heavy Metal Links: Correlating Dust and Topsoil Contamination in Vilnius Schools" Land 13, no. 1: 79. https://doi.org/10.3390/land13010079

Unsal, M.H., Ignatavičius G., Valiulis A., Prokopciuk N., Valskienė R., & Valskys, V. 2024. "Assessment of Heavy Metal Contamination in Dust in Vilnius Schools: Source Identification, Pollution Levels, and Potential Health Risks for Children" Toxics 12, no. 3: 224. https://doi.org/10.3390/toxics12030224

Unsal, M.H., Ignatavičius, G., Valskienė, R., Valskys, V. 2023. Long-Term Heavy Metal Accumulation in Sediment Dust of Schools in Vilnius: A Case Study. In Proceedings of the 12th International Scientific Conference, Environmental Engineering, Vilnius, Lithuania, 27–28 April 2023.

Ignatavičius, G., Unsal, M.H., Busher, P.E., Wołkowicz, S., Satkūnas, J., Šulijienė, G., Valskys, V. 2022. Geochemistry of mercury in soils and water sediments[J]. AIMS Environmental Science, 9(3): 277-297. doi: 10.3934/environsci.2022019

Ignatavičius, G., Unsal, M. H., Busher, P. E., Wołkowicz, S., Satkūnas, J., Valskys, V. 2022. Mercury and methylmercury in Baltic Sea sediments, and Polish and Lithuanian soils. Geological Quarterly, 66(3), Article 22. https://gq.pgi.gov.pl/article/view/33342

Scientific events where the results of the dissertation were presented

15th Conference of Young Scientists of Lithuania BioFuture: Perspectives in the Natural and Life Sciences Conference: Mercury and Methylmercury in Baltic Sea Sediments, Polish and Lithuanian Soils.

12th International Conference “Environmental Engineering“: Heavy Metals in Long-Term Accumulated Indoor Dust of Schools and It‘s Potential Health Risks

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