**THESIS OF THE DOCTORAL (PhD) DISSERTATION**

# ASSESSMENT OF THE EFFECTS OF ASBESTOS CEMENT ROOFING ON RAINWATER QUALITY PARAMETERS AND SUITABILITY FOR USE THROUGH THE GROWTH AND DEVELOPMENT OF *TRIFOLIUM PRATENSE*, *MEDICAGO SATIVA* AND *SOLANUM LYCOPERSICUM*


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# 1. RESEARCH OBJECTIVES

In the course of my research, I defined an integrated set of research objectives extending from the material-structural examination of asbestos cement (hereinafter: AC) products, through water-quality analysis, to plant physiological and toxicological assessment.

- **C-1:** My primary objective was to investigate the chrysotile content and detectability of different types of AC products. Using Fourier-transform infrared (hereinafter: FT-IR) spectroscopy, I analysed corrugated roofing sheets, flat AC sheets and AC pipes, comparing samples exposed to environmental conditions with samples protected from such exposure. I investigated the relationship between matrix degradation and chrysotile detectability, as well as the influence of structural characteristics on asbestos detectability and environmental durability.
- **C-2:** My second objective was to characterise the quality of rainwater that had come into contact with AC roofing. I determined pH, specific electrical conductivity, total hardness, total suspended solids, TOC, $BOD_5$ and $COD_p$, together with Zn, Pb and Hg concentrations, and compared these values with those of normal rainwater samples. The aim was to assess metal mobilisation from the cement matrix, changes in water quality and the environmental safety of the water for irrigation use.
- **C-3:** My third objective was to investigate the plant physiological effects of contaminated water on the model plants *Trifolium pratense*, *Solanum lycopersicum* and *Medicago sativa*. I analysed germination percentage, germination dynamics, changes in root length and shoot height at different doses, and evaluated dose–

response relationships using linear regression. On this basis, I determined the magnitude and biological significance of the abiotic stress effect.

– **C-4:** My fourth objective was the integrated assessment of environmental and toxicological mechanisms of action. I investigated the potential synergistic effects of asbestos fibres and mobilised heavy metals, as well as the relationships between plant responses indicative of oxidative stress and morphological changes. The aim was to qualitatively assess the agricultural and food-chain safety risks associated with contaminated irrigation water.

– **C-5:** My fifth objective was to provide a scientific basis for municipal asbestos abatement and phase-out strategies. I interpreted the water-quality alterations in rainwater runoff from AC roofs and the associated plant physiological responses within the context of municipal environmental risk, with particular attention to rainwater harvesting and irrigation use. My aim was to develop a criteria framework supporting the prioritisation of municipal asbestos phase-out programmes and environmental-safety decision-making.

# 2. MATERIALS AND METHODS

This dissertation follows a publication-based format; therefore, the methodological framework was built around an integrated presentation of the analytical and experimental procedures detailed in my previously published, peer-reviewed, quartile-ranked papers. The research comprised several interrelated levels of investigation, including the material-structural and degradation characterisation of asbestos cement products, analysis of the physicochemical parameters of rainwater in contact with roof surfaces, experimental investigation of the plant physiological effects of contaminated water, and a systems-level assessment of the municipal sustainability and risk framework.

## 2.1 FT-IR analysis of AC samples

Three types of asbestos cement (AC) products were used in the analyses: corrugated roofing sheets, flat sheets and AC pipes. Two sample groups were distinguished within each category: samples exposed to environmental conditions and samples stored under protected conditions (n = 5–5). The exposed samples were typically 30–40-year-old products collected from buildings in use or during demolition at different locations in Győr, whereas the separated samples consisted of unused, stored elements from the same region. The samples were kept under hermetically sealed conditions until analysis. Environmental exposure was evaluated comparatively without the application of artificial stress, based on the assumption that the in-service samples had been jointly affected by ageing, physical damage, solar radiation and precipitation. During preparation, the surfaces were cleaned with distilled water and isopropyl alcohol; the samples were then cut to a uniform size and embedded in a multicomponent epoxy resin. Disc specimens measuring 30 × 5 mm were prepared in preformed moulds, polished to obtain an appropriate surface quality and mounted on the instrument stage. Measurements were performed using a PerkinElmer Spectrum 400 infrared spectrometer (PerkinElmer, MA, USA) with a 0.3 mm-diameter diamond crystal under adequate contact conditions. Absorbance maps were generated by scanning the measurement points, and the resulting spectra were matched against a spectral library to identify the components. Asbestos types exhibit characteristic absorption in the 1200–900 $cm^{-1}$ and 600–300 $cm^{-1}$ wavenumber ranges, while chrysotile was distinguished from amphibole asbestos by the double hydroxyl-group bands at 3693 and 3648 $cm^{-1}$. Repeated FT-IR measurements were performed on all samples; the goodness of fit to the chrysotile spectra was recorded, and statistical evaluation was conducted using the ten best spectral matches. Because the measurements are sensitive to moisture and carbon dioxide, a background spectrum was recorded before each analysis to eliminate environmental interference.

## 2.2 Rainwater sampling, simulation experiments and water-analytical measurements

The study employed a mixed-methods approach combining field sampling and laboratory analytical measurements, thereby enabling comparison between real environmental exposure scenarios and controlled laboratory conditions. The investigation focused on a corrugated asbestos cement (AC) roofing sheet widely used during the 1980s and typically containing 8–10 wt% chrysotile, together with cement and other binding materials. Rainwater samples were collected at 20 sampling points in Győr: ten samples were taken directly from water running off AC roof surfaces, and ten from uncontaminated rainfall that had not come into contact with a roof surface. This dual sampling strategy enabled the roof-surface effect to be distinguished from the atmospheric background load. Samples were collected in 10-litre glass vessels, and sampling times were selected with reference to current meteorological conditions and preceding air-quality indicators. Contact with any material that could have influenced the chemical composition of the water was excluded during sampling. The roof sheets examined differed in age and degree of erosion; no detailed mineralogical or surface characterisation was performed, and their condition was assessed on the basis of visual documentation. Rainfall intensity and duration were also recorded during field data collection; the mean intensity was approximately 10 mm per 5 minutes, and the typical event duration was 15–20 minutes. Only the first 5 litres of runoff from each rainfall event were collected because of the greater contaminant load during the initial first-flush phase. In parallel with natural sampling, distilled-water simulation experiments were conducted: 1 litre of distilled water was passed over an eroded AC sheet over 300 seconds to investigate leaching processes resulting from direct contact with the material under controlled conditions. The use of distilled water minimised the effect of background ions and enabled material-specific changes to be distinguished. The collected natural and simulated samples were subjected to laboratory water analysis. I determined pH, specific electrical conductivity, total suspended solids, total hardness, total organic carbon (TOC), chemical oxygen demand

and biochemical oxygen demand ($COD_p$ and $BOD_5$), and measured mercury, lead and zinc concentrations. The measurements were carried out in accordance with the relevant Hungarian (MSZ) and European (EN) standards, ensuring the comparability and reliability of the results.

### 2.3 Plant physiological investigations

The plant physiological investigations included three cultivars of each of three plant species: *Trifolium pratense* (Salino, Rozeta and Altaswede), *Medicago sativa* (Emiliana, Gea and Algonquin) and *Solanum lycopersicum* (Manó, Kecskeméti 549 and Mobil). The seeds were obtained from Hungarian distributors. The experiment aimed to determine the dose-dependent effects of water contaminated with asbestos cement on germination and early plant development under controlled laboratory conditions. The seeds were surface-sterilised in a 2.0% NaOCl solution for 2 minutes and then rinsed three times with sterile distilled water. Ten seeds were subsequently placed on moistened cotton pads in sterile Petri dishes; five replicates were used at each treatment level, resulting in a total of 350 experimental units per plant species. Germination was conducted at 22 ± 1 °C under digitally controlled temperature and regulated ventilation. Full-spectrum LED lighting provided controlled light conditions during the early post-emergence cultivation stage. To prepare the treatment solutions, asbestos cement products commonly used in Hungary and containing 8.0–10.0% chrysotile and 90.0–92.0% cement were mechanically ground to a fine powder. The resulting material was suspended in double-distilled water at concentrations of 1.00, 2.00, 5.00, 10.0, 25.0 and 50.0 mg/L. The suspensions were incubated at room temperature for 24 hours, filtered through a 0.45 μm pore-size filter, and stored in sterile glass containers in the dark at 4 °C until use. Double-distilled water served as the control. Germination percentage and germination time were recorded continuously during the experiment, and root length was also measured. Shoot height was determined on day 31 as a parameter of early vegetative growth. Measurements were made using a Burg-Wächter Precise PS-72150 digital calliper. The results were reported as mean ± standard error, and differences among treatments were evaluated using one-way analysis of variance (ANOVA) at a significance level of 0.05.

### 2.4 Regional asbestos burden and a framework for the asbestos-free urban transition

The transition towards asbestos-free urban and semi-urban environments is a complex, multidimensional process shaped by regulatory frameworks, technological innovation, stakeholder participation and socioeconomic conditions. To interpret this process, I applied the Multi-Level Perspective (MLP) framework, which enables the interactions among niche innovations, regime structures and macro-environmental factors to be examined. Within the model, the niche level was represented by sociotechnical innovations supporting asbestos-free and sustainable urban development, the regime level by regulatory, economic and social structures, and the macro level by global sustainability trends, climate policy and public-health priorities. The asbestos-free transition cannot be understood solely as a technological or policy issue; it also requires consideration of historical embeddedness, institutional rigidity and structural interactions. The examination of regional differences is particularly important because industrial legacies, the age of the building stock, the nature of infrastructure and institutional capacities may produce spatially differentiated asbestos burdens and adaptive capacities. To capture these differences empirically, I applied a taxonomic index-construction method to the Hungarian NUTS (Nomenclature of Territorial Units for Statistics) level 2 regions. NUTS 2 denotes the second level of the European Union's territorial classification for statistical purposes. The indicator system was based on *Mesothelioma malignum* incidence rates from the Hungarian National Cancer Registry; the study period was 2005–2020, with analyses conducted for 2005, 2010, 2015 and 2020. Data expressed in different units were standardised, and a reference value was derived from the maximum standardised values. The deviation of each region from this reference was measured using Euclidean distance, after which a dimensionless integrated index ranging from 0 to 1 was calculated, with higher values indicating a greater relative asbestos burden.

# 3. RESULTS AND THEIR EVALUATION

This chapter presents and interprets the empirical results of the research in light of the theoretical framework and methodological approach described above. The results are evaluated within a logical, sequential structure across several levels of investigation: material-structural and degradation analyses, changes in water-quality parameters, plant physiological responses, and the calculation of the regional asbestos-burden index. The purpose of the chapter is not merely to describe the data, but also to interpret them causally and systemically, with particular attention to the conditions of the asbestos-free urban transition and its sustainability implications. The evaluation emphasises dose–effect relationships, the significance of spatial differences, and the formulation of policy and environmental-safety conclusions.

## 3.1 Environmental mobility of asbestos fibres in the soil–water system

In my review research, I examined the environmental behaviour of asbestos fibres along the water–soil continuum, with particular attention to mobility mechanisms and the complexity of exposure pathways. The purpose of the analysis was to interpret the environmental transport processes of asbestos within a systems-level framework and to identify research gaps arising from approaches traditionally focused on inhalation exposure. The results demonstrated that the environmental risk of asbestos cannot be restricted to its airborne dispersal. Processes occurring in aquatic and soil media substantially affect fibre mobility, availability and potential ecotoxicological effects. The physicochemical properties of asbestos fibres—particularly fibre geometry, surface charge and mineralogical composition—play a decisive role in their movement through the soil–water system.

The analysis demonstrated that water pH, ionic composition and colloidal fraction significantly influence the dispersion and aggregation of asbestos fibres. Binding to soil colloids and interactions with dissolved organic matter may modify fibre transport, thereby affecting the potential for entry into surface water and groundwater. I established that soil is not merely a passive medium but functions as an active geochemical system capable of both mobilising and immobilising asbestos fibres. Particular emphasis was placed on the environmental degradation of asbestos cement products. Ageing and weathered asbestos cement structures were identified as potential diffuse sources from which fibres may gradually enter the soil–water system. This process is especially relevant in rainwater runoff and surface infiltration, where physical erosion and chemical dissolution mechanisms jointly contribute to environmental loading. A key finding was that the environmental transport of asbestos is not a linear process but the outcome of a complex, multifactorial system. Soil–water interactions, hydrological conditions and the degradation state of the material jointly determine exposure pathways. I also identified a substantial gap in the current literature concerning plant and microbial interactions, which limits the reliability of ecosystem-level risk assessments. The environmental behaviour of asbestos can only be interpreted through an integrated, holistic approach. Exposure models remain incomplete unless interactions among air, soil and water are considered. The results confirm that investigating environmental mobility is essential for a more accurate determination of the long-term ecotoxicological and public-health risks of asbestos and provides a scientific basis for the further development of complex environmental risk-assessment models.

## 3.2 Detection of chrysotile asbestos in AC samples

### 3.2.1 Detection of chrysotile asbestos in corrugated asbestos cement sheets

For corrugated asbestos cement sheet samples exposed to environmental conditions, the mean chrysotile identification score was 0.747193 (1). The highest spectral match, 0.810265 (1), was recorded at the fourth measurement point of sample AC-S-E-5, whereas the lowest detection value, 0.689985 (1), was recorded at the sixth measurement point of sample AC-S-E-4. Among the samples, AC-S-E-2 showed the highest mean chrysotile detection score, 0.754616 (1), while the lowest, 0.742706 (1), was observed for AC-S-E-4. For samples protected from environmental exposure (separated samples), the mean chrysotile identification score was 0.645437 (1). The highest match, 0.701221 (1), was measured at the fourth measurement point of sample AC-S-S-4, while the lowest value, 0.569845 (1), was measured at the eighth measurement point of sample AC-S-S-5. Among the separated samples, the highest mean value, 0.678396 (1), was found for AC-S-S-1 and the lowest, 0.605309 (1), for AC-S-S-5. The comparison ratio for the complete sample series was 115.8%. The mean deviation from the mean was 0.00634 (1) for the exposed samples and 0.00429 (1) for the separated samples. A clear difference was observed in the spectral matching of chrysotile in the corrugated asbestos cement samples: the environmentally exposed (E) samples showed higher mean values of approximately 0.75 (1), whereas the separated (S) samples displayed greater variability and a central value of approximately 0.65 (1). Because the samples were based on an asbestos cement matrix, a broad peak appeared at approximately 1400 $cm^{-1}$. This can be explained by the stretching vibrations of C–O bonds in calcium silicate hydrate, calcium sulphate and calcium carbonate present in the cement mixture. Two characteristic spectral regions could be distinguished: O–H stretching vibrations occurred in the 3700–3500 $cm^{-1}$ range, while lattice vibrations appeared in the 1200–500 $cm^{-1}$ range.

### 3.2.2 Detection of chrysotile asbestos in flat asbestos cement sheets

The environmentally exposed flat asbestos cement samples had a mean chrysotile identification score of 0.753552 (1). The highest match, 0.820001 (1), was measured at the tenth measurement point of sample AC-F-E-3, while the lowest value, 0.694962 (1), was measured at the second measurement point of sample AC-F-E-1. Sample AC-F-E-4 had the highest mean detection score, 0.773262 (1), whereas AC-F-E-1 had the lowest, 0.723031 (1). The separated samples had a mean chrysotile identification score of 0.632309 (1). The highest match, 0.698857 (1), was measured at the fifth measurement point of sample AC-F-S-4, while the lowest value, 0.548113 (1), was measured at the fourth measurement point of sample AC-F-S-1. The highest mean value, 0.660611 (1), was recorded for AC-F-S-4 and the lowest, 0.602867 (1), for AC-F-S-1. The comparison ratio for the complete sample series was 119.2%. The mean deviation from the mean was 0.00540 (1) for the exposed samples and 0.00801 (1) for the separated samples. In the environmentally exposed flat-sheet samples, the spectral match for chrysotile was more pronounced, and chrysotile was more readily identifiable and exhibited a higher match score. The separated samples displayed greater variability, with a central value below 0.65 (1). The characteristic chrysotile peaks occurred in the 3700–3500 $cm^{-1}$ and 1200–500 $cm^{-1}$ ranges. As with the corrugated sheets, a broad peak was also observed at approximately 1400 $cm^{-1}$ because of the presence of the asbestos cement matrix.

### 3.2.3 Detection of chrysotile asbestos in asbestos cement pipes

The environmentally exposed asbestos cement pipe samples had a mean chrysotile identification score of 0.800607 (1). The highest match, 0.888831 (1), was measured at the fifth measurement point of sample AC-P-E-4, while the lowest value, 0.742121 (1), was measured at the eighth measurement point of sample AC-P-E-2. The highest mean value, 0.811325 (1), was recorded for AC-P-E-3 and the lowest, 0.793972 (1), for AC-P-E-4. The separated samples had a mean chrysotile identification score of 0.794369 (1). The highest match, 0.865548 (1), was measured at the seventh measurement point of sample AC-P-S-2, while the lowest value, 0.724580 (1), was measured at the ninth measurement point of sample AC-P-S-3. The highest mean value, 0.808812 (1), was recorded for AC-P-S-2 and the lowest, 0.775610 (1), for AC-P-S-4. The comparison ratio for the complete sample series was 100.8%. The mean deviation from the mean was 0.00617 (1) for the exposed samples and 0.00438 (1) for the separated samples. The spectral match in the environmentally exposed pipe samples was more uniform, and chrysotile was more readily detectable and more consistent. The exposed samples showed a trend around 0.80 (1), while the separated samples were approximately 0.79 (1). The characteristic chrysotile peaks also occurred in the 3700–3500 $cm^{-1}$ and 1200–500 $cm^{-1}$ ranges. The broad peak at approximately 1400 $cm^{-1}$ indicates the presence of the asbestos cement matrix.

## 3.3 Results of the water-analytical investigations

### 3.3.1 Water-quality characteristics of normal rainwater and rainwater in contact with asbestos cement

The investigation revealed substantial differences between the water-quality parameters of normal rainwater and rainwater that had come into contact with an asbestos cement (AC) roof surface. The pH of water collected from AC surfaces was significantly higher, with a mean of 7.09 ± 0.162 compared with 5.73 ± 0.265 for normal rainwater, representing an increase of 23.8%. This indicates that the roofing material exerted an alkalising effect, probably due to the leaching of calcium-containing compounds. The altered pH may influence the solubility and mobility of other contaminants. A similar trend was observed for electrical conductivity, which reflects the total concentration of dissolved ions. Mean conductivity was 49.8 ± 6.00 µS/cm in samples that had contacted AC, compared with 31.4 ± 8.88 µS/cm in normal rainwater. This suggests that the roofing material introduced additional dissolved ions into the water, probably through the leaching of mineral components and other substances.

Total suspended solids (TSS) showed only a slight increase in samples from AC roofs: the mean value was 6.40 ± 1.51 mg/L, compared with 6.00 ± 0.94 mg/L in normal rainwater. The 6.67% increase indicates that the roofing material did not contribute substantially to the particulate load. Nevertheless, even a small increase in TSS may affect water quality because suspended solids can transport other contaminants, including heavy metals. Water hardness increased markedly by 92.6% in samples from AC surfaces compared with normal rainwater. Mean hardness was 32.4 ± 9.75 mg CaO/L in the AC samples and 16.8 ± 5.35 mg CaO/L in normal rainwater. This increase in hardness was probably attributable to calcium and magnesium compounds leaching from the cement matrix. Increased water hardness may lead to scale formation in pipework and reduce the effectiveness of detergents. Analysis of total organic carbon (TOC) and oxygen-demand parameters provided further information on the effects of AC roofing on water quality. TOC increased by 40.9% in samples that had contacted AC, reaching a mean of 5.10 ± 1.08 mg/L compared with 3.62 ± 0.57 mg/L in normal rainwater. This substantial increase indicates that water in contact with AC contained more organic matter, which may have originated from the wash-off of organic particles accumulated on the roof surface or from the leaching of organic components. Higher TOC levels are often associated with increased microbial activity, which may affect water quality, particularly during storage or distribution. Biochemical oxygen demand ($BOD_5$) increased slightly, by 4.92%, in the AC-contact samples. The mean $BOD_5$ value was 4.05 ± 0.88 mg/L in the AC samples and 3.86 ± 0.25 mg/L in normal rainwater. Although the increase was moderate, it suggests that organic matter present in water that had contacted AC may have been more biologically available, or that the microbial load may have been slightly higher, potentially affecting suitability for irrigation or drinking-water use. The most pronounced change was observed for chemical oxygen demand ($COD_p$), with values in the samples that had contacted AC roofs substantially exceeding those of normal rainwater. This increase indicates a greater quantity of oxidisable organic and inorganic substances, probably originating from the roofing material or contaminants deposited on its surface. A high $COD_p$ value indicates potential contamination and, in the absence of appropriate treatment, may adversely affect receiving aquatic ecosystems.

### 3.3.2 Heavy-metal contamination of normal rainwater and rainwater in contact with asbestos cement

Analysis of metal concentrations in rainwater revealed substantial differences between normal samples and samples that had come into contact with an asbestos cement (AC) roof surface. The concentrations of zinc (Zn), lead (Pb) and mercury (Hg) increased considerably in the AC-contact samples, indicating that the roofing material may act as a potential source of these heavy metals. Total zinc concentration in normal rainwater was very low, with a mean of 0.0024 ± 0.0005 mg/L, close to the detection limit. This indicates that, in the absence of contact with metal surfaces, the zinc content of rainwater is minimal and derives primarily from atmospheric contamination. By contrast, zinc concentration in rainwater that had contacted an AC roof surface increased markedly to 0.155 ± 0.012 mg/L. This substantial increase probably originated from zinc present in the asbestos cement material, either as a result of the manufacturing process or through atmospheric contaminants accumulated on the roof surface. The increase in zinc concentration demonstrates that AC roofing can materially alter the zinc content of harvested rainwater. However, for irrigation use, the measured concentration of 0.155 mg/L is not necessarily adverse in itself, because zinc is an essential micronutrient and may also be beneficial to plants within this concentration range.

The analysis also showed that the AC roof surface significantly increased lead and mercury concentrations in harvested rainwater. Lead concentration increased by 127.3%, from 0.00133 ± 0.0004 mg/L in normal samples to 0.003 ± 0.002 mg/L in samples that had contacted AC. Lead is a highly toxic metal, particularly for children, and its presence in water is strictly regulated. The elevated lead level indicates that asbestos cement may contain lead, either as a manufacturing impurity or as a result of environmental deposition. This particularly supports the need for regular monitoring of rainwater harvested from such surfaces, especially when intended for household use. A similar trend was observed for mercury: its concentration increased by 75.4% in AC-contact samples, from 0.0011 ± 0.0003 mg/L in normal rainwater to 0.002 ± 0.0004 mg/L. Although the absolute concentrations were low, the relative increase was substantial and raises the possibility of long-term exposure and bioaccumulation.

### 3.3.3 Water-quality characteristics of simulated water samples in contact with asbestos cement

Analysis of the simulated water samples enabled examination of how asbestos cement (AC) roofing affects water-quality parameters relative to neutral and mildly acidic initial conditions. The pH of the simulated samples was 7.21 ± 0.154, consistent with the pH-increasing trend observed in natural rainwater that had contacted AC. This indicates that alkaline asbestos cement has a substantial neutralising effect, particularly in a mildly acidic medium. Under neutral initial conditions, however, the shift in pH was minimal, suggesting that the effect of AC is more pronounced in acidic water. Electrical conductivity in the simulated samples increased to 6.80 ± 1.55 μS/cm from the near-zero value of distilled water. This increase indicates that AC roofing introduced additional dissolved ions into the water, probably through the leaching of mineral components and other soluble substances. The increase in ionic content may influence the ion balance and usability of the water. Total suspended solids (TSS) in the simulated samples amounted to 5.30 ± 0.48 mg/L, comparable with the values measured in rainwater that had contacted AC roofs. This similarity suggests that AC roofing may constitute a potential source of particulate matter, which may also act as a carrier of other contaminants and affect water transparency and quality. The measured total hardness of the simulated water samples was 17.7 ± 2.67 mg CaO/L, within the range observed for normal rainwater. This indicates that AC roofing can increase water hardness, probably through the dissolution of calcium compounds. Increased hardness may influence interactions with pipework, detergents and other processes, which is an important consideration when evaluating the effects of AC roofs on water quality. Comparative analysis of organic-matter content and oxygen demand in the simulated samples and natural rainwater samples that had contacted AC provided further information on the mechanism of action. Total organic carbon

(TOC) in the simulated samples was 3.82 ± 0.14 mg/L, lower than the mean value of 5.10 ± 1.08 mg/L in rainwater samples that had contacted AC. This difference suggests that AC roofing may contribute additional organic matter to the water, potentially through leaching or contaminants accumulated on the roof surface. The lower TOC value in simulated samples, which were not exposed to the full environmental load, supports the assumption that atmospheric contaminants deposited on the roof surface play a substantial role in the increase in TOC. The mean biochemical oxygen demand ($BOD_5$) of the simulated rainwater samples was 3.44 ± 0.42 mg/L, slightly lower than the values measured in normal rainwater and AC-contact samples. The slight increase in $BOD_5$ in AC-contact samples suggests that the organic matter present may have been more biologically available or that the microbial load was higher, probably because of organic components originating from the roofing material. The $BOD_5$ value of the simulated samples, which was close to that of normal rainwater, indicates that biochemical oxygen demand remained relatively stable in the absence of external organic loading, thereby highlighting the role of AC roofs in increasing organic contamination. Chemical oxygen demand in the simulated samples ($COD_p$) was 3.43 ± 0.41 mg/L, practically identical to that of normal rainwater. By contrast, $COD_p$ in the AC-contact samples was significantly higher, at 6.71 ± 1.78 mg/L. This substantial increase indicates the presence of greater quantities of oxidisable organic and inorganic substances, probably originating from the asbestos cement material and environmental contaminants accumulated on the roof surface. The relatively low $COD_p$ value of the simulated samples suggests that, in the absence of external contaminants, the AC material alone does not cause a substantial increase in $COD_p$, although in combination with environmental factors it may contribute to an increase in oxidisable components.

### 3.3.4 Heavy-metal contamination of simulated samples in contact with asbestos cement

Analysis of heavy-metal concentrations in the simulated water samples, in comparison with both rainwater that had contacted an asbestos cement (AC) roof surface and normal rainwater, enabled important conclusions to be drawn regarding the effect of the AC material on water quality. The mean total zinc concentration in the simulated samples was 0.0361 ± 0.002 mg/L, substantially exceeding the zinc levels characteristic of normal rainwater. Zinc concentration was even higher in natural rainwater samples that had contacted AC, with a mean of 0.155 ± 0.012 mg/L. This considerable increase indicates that AC roofing acts as a substantial source of zinc, probably through leaching from the cement matrix or the presence of zinc-containing additives used during manufacture.

The elevated zinc concentration in the simulated samples indicates that the AC material can release zinc into water even in the absence of environmental contaminants. This demonstrates that asbestos cement roofing itself may contribute to an increase in the zinc content of harvested rainwater. However, the measured concentration cannot in itself be regarded as an unequivocal environmental or human-health risk, because zinc is an essential micronutrient and may play a beneficial role when the water is used for irrigation. The mean lead concentration in the simulated samples was 0.0021 ± 0.0017 mg/L, slightly higher than that of normal rainwater but lower than the concentration measured in AC-contact samples. This suggests that the AC material may contribute to the presence of lead to some extent, although its role is less dominant than in the case of zinc. The slight increase in the simulated samples indicates that limited lead leaching may occur from the AC material, whereas the greater increase observed in natural rainwater that had contacted AC was probably attributable to environmental lead accumulated on the roof surface. This emphasises the importance of considering both material composition and environmental exposure when evaluating the lead content of rainwater harvested from AC roofs. Total mercury concentration in the simulated samples was 0.0063 ± 0.009 mg/L, substantially exceeding the values characteristic of normal rainwater. This elevated concentration suggests that mercury may be present in the AC material and may be released under certain conditions. However, the high variability of the measured values indicates that further investigation is required to understand the sources and behaviour of mercury more precisely.

## 3.4 Results of germination and early vegetative-growth experiments

### 3.4.1 Results for *Trifolium pratense*

The control germination percentage was 88.0% for Salino, 86.0% for Rozeta and 90.0% for Altaswede. The mean germination percentage of Trifolium pratense seeds at 25.0 °C was 88.0 ± 2.00%. A decrease was already observed at a dose concentration of 1 mg/L: germination declined to 82.0% for Salino, 80.0% for Rozeta and 84.0% for Altaswede. At 50 mg/L, germination decreased to 48.0% for Salino, 46.0% for Rozeta and 58.0% for Altaswede, corresponding to 54.5%, 53.5% and 64.4% of the control, respectively. The mean for the entire group was 93.2 ± 0.16% at 1.00 mg/L and 57.5 ± 6.04% at 50 mg/L, expressed as a percentage of the control. The mean difference between the two dose levels was −35.7 ± 5.90%. The coefficient of determination ($R^2$) was 0.9840 for Salino, 0.9974 for Rozeta and 0.9313 for Altaswede, indicating a strong dose–response relationship. At 1 mg/L, root length was reduced to 95.2% of the control for Salino, 95.7% for Rozeta and 95.5% for Altaswede. At 50 mg/L, root length decreased to 47.6% for Salino, 47.8% for Rozeta and 54.5% for Altaswede. The corresponding mean values were 95.4 ± 0.21% at 1.00 mg/L and 50.0 ± 3.94% at 50.0 mg/L. The $R^2$ values were 0.9791 for Salino, 0.9941 for Rozeta and 0.9611 for Altaswede. At 1 mg/L, shoot height was reduced to 91.3% for Salino, 90.5% for Rozeta and 93.2% for Altaswede relative to the control. At 50 mg/L, shoot height decreased to 41.3% for Salino, 45.2% for Rozeta and 45.5% for Altaswede. The mean was 91.7 ± 1.39% at 1.00 mg/L and 44.0 ± 2.34% at 50.0 mg/L. The respective $R^2$ values were 0.9439, 0.9662 and 0.9828.

### 3.4.2 Results for *Medicago sativa*

The control germination percentage was 92.0% for Emiliana, 94.0% for Gea and 90.0% for Algonquin. The mean germination percentage of Medicago sativa seeds at 25.0 °C was 92.0 ± 2.00%. A detectable decrease was already observed at 1 mg/L: germination declined to 88.0% for Emiliana and Gea and to 86.0% for Algonquin. At 50 mg/L, germination decreased to 54.0% for Emiliana, 50.0% for Gea and 52.0% for Algonquin. Expressed as a percentage of the control, these values were 58.7%, 53.2% and 57.8%, respectively. The mean for the entire group was 94.9 ± 1.15% at 1.00 mg/L and 56.6 ± 2.95% at 50.0 mg/L. The mean difference between the two dose levels was −38.4 ± 1.81%. The coefficient of determination ($R^2$) was 0.9827 for Emiliana, 0.9842 for Gea and 0.9754 for Algonquin, indicating a strong dose–response relationship. At 1 mg/L, root length was reduced to 97.2% for Emiliana, 97.1% for Gea and 91.4% for Algonquin relative to the control. At 50 mg/L, root length decreased to 59.7% for Emiliana, 60.3% for Gea and 57.1% for Algonquin. The corresponding mean values were 95.2 ± 3.30% at 1.00 mg/L and 59.1 ± 1.68% at 50.0 mg/L. The $R^2$ values were 0.9781 for Emiliana, 0.9788 for Gea and 0.9844 for Algonquin. At 1 mg/L, shoot height was reduced to 98.8% for Emiliana, 96.3% for Gea and 96.4% for Algonquin relative to the control. At 50 mg/L, shoot height decreased to 65.9% for Emiliana, 69.1% for Gea and 69.9% for Algonquin. The mean was 97.2 ± 1.41% at 1.00 mg/L and 68.3 ± 2.14% at 50.0 mg/L. The $R^2$ values were 0.9818 for Emiliana, 0.9604 for Gea and 0.9429 for Algonquin.

### 3.4.3 Results for *Solanum lycopersicum*

The control germination percentage was 92.0% for Manó and 90.0% for both Kecskeméti 549 and Mobil. The mean germination percentage of *Solanum lycopersicum* seeds at 25.0 °C was 90.7 ± 1.15%. A measurable decrease was already observed at 1 mg/L: germination declined to 88.0% for Manó and Kecskeméti 549 and to 86.0% for Mobil. At 50 mg/L, germination decreased to 58.0% for Manó, 60.0% for Kecskeméti 549 and 56.0% for Mobil. Expressed as a percentage of the control, these values were 63.0%, 66.7% and 62.2%, respectively. The mean for the entire group was 96.3 ± 1.26% at 1.00 mg/L and 64.0 ± 2.36% at 50.0 mg/L. The mean difference between the two dose levels was −32.4 ± 1.13%. The coefficient of determination ($R^2$) was 0.9909 for Manó, 0.9823 for Kecskeméti 549 and 0.9916 for Mobil, indicating a close dose–response relationship. At 1 mg/L, root length was reduced to 91.9% for Manó, 87.5% for Kecskeméti 549 and 80.3% for Mobil relative to the control. At 50 mg/L, root length decreased to 55.4% for Manó, 59.7% for Kecskeméti 549 and 52.6% for Mobil. The corresponding mean values were 86.6 ± 5.87% at 1.00 mg/L and 55.9 ± 3.57% at 50.0 mg/L. The $R^2$ values were 0.9961 for Manó, 0.9710 for Kecskeméti 549 and 0.9908 for Mobil. At 1 mg/L, shoot height was reduced to approximately 95.6% in all three cultivars (Manó: 95.6%; Kecskeméti 549: 95.5%; Mobil: 95.6%) relative to the control. At 50 mg/L, shoot height decreased to 56.0% for Manó, 59.6% for Kecskeméti 549 and 57.8% for Mobil. The mean was 95.6 ± 0.05% at 1.00 mg/L and 57.8 ± 1.75% at 50.0 mg/L. The $R^2$ values were 0.9916 for Manó, 0.9863 for Kecskeméti 549 and 0.9941 for Mobil.

### 3.5 Systems-level analysis of the asbestos-free urban transition within the MLP framework

The strengthening of the global sustainability discourse, growing public awareness of public-health risks and international environmental-policy commitments jointly exert substantial structural pressure on urban systems. In the long term, these factors weaken the stability of solutions based on asbestos-containing infrastructure and contribute to the social legitimacy of asbestos abatement; however, they do not in themselves produce systemic transformation. If regime institutions fail to respond adaptively, the transition remains slow and fragmented. Analysis at the regime level showed that the persistence of the asbestos-containing built environment is a consequence of deep institutional and economic embeddedness. Construction and demolition practices, regulatory structures and a market logic based on cost optimisation jointly create path dependency and structural lock-in, slowing the diffusion of sustainable alternatives. The transition may accelerate when stricter regulation, financial incentives and social expectations emerge simultaneously; consequently, asbestos abatement requires not only technological but also institutional and economic transformation. Analysis at the niche level indicated that asbestos-free construction materials, innovative decontamination technologies and circular-economy models offer viable alternatives to the existing system. Their scaling, however, depends substantially on stable financing, a supportive regulatory environment and acceptance within the sector. The analysis also showed a risk that the original sustainability objectives may be partly displaced during widespread diffusion, particularly when cost-efficiency considerations override long-term environmental-safety priorities.

## 3.6 Regional assessment of asbestos burden

Because no standardised methodology is currently available for the interdisciplinary assessment of asbestos burden, the analysis was conducted using a preselected group of indicators. Following data organisation, categorisation and standardisation, Euclidean distance values were calculated and used to derive an alternative, dimensionless asbestos-burden index. The national mean index value was 0.3098 in

2005, 0.2377 in 2010, 0.2612 in 2015 and 0.3389 in 2020, indicating an overall low but increasing level of asbestos burden. The increase was primarily attributable to the upward trend in mesothelioma incidence and, in certain regions, to growth in the quantity of asbestos-containing waste. Over the entire period from 2005 to 2020, the mean rate of change was +28.5%; this comprised a decrease of −18.0% between 2005 and 2010, an increase of +24.5% between 2010 and 2015, and an increase of +33.2% between 2015 and 2020. The results revealed substantial regional differences: over the full period, the largest increases occurred in Western Transdanubia (+182.9%) and the Northern Great Plain (+111.5%), while the largest decrease was observed in the Southern Great Plain (−56.3%). Four of the eight regions showed an increase and four a decrease. Based on the extreme values for each interval, the Northern Great Plain recorded the largest increase between 2005 and 2010 (+48.7%), while the Southern Great Plain showed the greatest decrease (−57.9%); between 2010 and 2015, Western Transdanubia recorded the largest increase (+122.2%), while the largest decline occurred in the Pest Region (−55.9%); and between 2015 and 2020, the Northern Great Plain (+78.6%) and Northern Hungary (−27.0%) represented the two extremes of the range of change. The results demonstrate that asbestos burden can be characterised multidimensionally and quantified using the available indicators, although broadening the indicator set could further refine the index. The spatial pattern of asbestos burden is differentiated; therefore, regional differences must be considered in the policy and strategic planning of the asbestos-free urban transition.

# 4. NEW SCIENTIFIC RESULTS

**Thesis 1: I demonstrated that matrix degradation in asbestos cement products exposed to environmental conditions significantly affects the FT-IR spectral detectability of chrysotile, and that an increase in the spectral match score may serve as an indicator of declining structural integrity.**

Rationale: I showed that the spectral match for chrysotile was systematically higher in corrugated and flat asbestos cement elements exposed to environmental conditions (UV radiation, precipitation and temperature fluctuations) than in separated samples. This indicates that degradation of the cement matrix increases the relative spectral visibility of the fibrous components. The results demonstrate that FT-IR can be used not only for qualitative identification but also for condition-assessment purposes.

**Thesis 2: I demonstrated that the water-quality parameters of rainwater in contact with asbestos cement (pH, conductivity, hardness, TOC, $COD_p$ and $BOD_5$) differ significantly from those of normal rainwater, and that these changes can be explained by ion mobilisation from the cement matrix.**

Rationale: The investigations confirmed the alkalising effect, the increase in ionic load and the rise in oxidisable components. The results demonstrate that an asbestos cement roof surface is not a passive collection surface but an active chemical source that alters the composition of runoff water. This finding provides a new approach to risk assessment for rainwater utilisation.

**Thesis 3: I demonstrated that the concentrations of heavy metals mobilised from the asbestos cement matrix (Zn, Pb and Hg) substantially exceed those measured in normal rainwater and that mobilisation originates partly from the matrix and partly from surface accumulation.**

Rationale: Comparison of simulated and natural rainwater samples enabled matrix-derived and environmentally derived components to be distinguished. The results demonstrate that structural ageing transforms asbestos cement into a potential source of heavy-metal emissions. This finding is relevant from both environmental-toxicological and regulatory perspectives.

**Thesis 4: I demonstrated that water contaminated with asbestos cement exerts a dose-dependent growth-inhibitory effect on the model plants *Trifolium pratense*, *Medicago sativa* and *Solanum lycopersicum*, and that this effect can be adequately modelled by linear regression.**

Rationale: Significant reductions in germination percentage, root length and shoot height were detectable even at low concentrations, with high coefficients of determination ($R^2$). This confirms that alterations in water quality are translated into biological effects. The plant physiological results empirically substantiate the causal relationship between material-structural degradation and ecological response.

**Thesis 5: I developed and empirically substantiated a multilevel exposure–effect model based on the Multi-Level Perspective (MLP), extending from the material-structural degradation of asbestos cement, through alterations in water quality and biological responses, to the systems-level interpretation of the sustainability transition.**

Rationale: At the micro level, the model integrates materials-science and water-analytical processes; at the meso level, ecological and agro-environmental effects; and at the macro level, municipal and regulatory transitions. I demonstrated that the asbestos cement problem is not an isolated environmental phenomenon but a transition challenge embedded in a sociotechnical system, in which environmental effects arising from degradation feed back into regulatory and remediation structures. This systems-level approach offers a new interpretative framework for the scientific underpinning of the asbestos-free municipal transition.

**Thesis 6: I demonstrated that water running off asbestos cement surfaces exposed to environmental conditions shows significant increases in pH, conductivity, water hardness and Zn, Pb and Hg concentrations, and that the resulting altered water exerts a dose-dependent, linearly modelled growth-inhibitory effect on agriculturally important plant species.**

Rationale: I demonstrated that the pH of rainwater in contact with asbestos cement is, on average, nearly one full unit higher than that of normal rainwater; water hardness may almost double; Zn concentration exceeds normal-rainwater values by several orders of magnitude; Pb and Hg concentrations also increase significantly; and extracts applied over the 1–50 mg/L concentration range may reduce germination percentage, root length and shoot height by 30–60%. The results demonstrate that asbestos cement degradation generates measurable ecological effects through a water-mediated exposure pathway; consequently, asbestos risk cannot be interpreted solely in terms of airborne fibrous exposure.

# 5. PUBLICATIONS RELATED TO THE DISSERTATION TOPIC

- Szandi, Á., Balog, Z., Zaka, K. S., & Macher, G. Z. (2026). Persistence of Asbestos-Containing Friction Materials in the Hungarian Waste Stream Twenty Years After the European Union Ban. INTERNATIONAL JOURNAL OF ENVIRONMENTAL RESEARCH AND PUBLIC HEALTH, 23(6), 802. http://doi.org/10.3390/ijerph23060802 (**Q1**)
- Macher, G. Z., Beke, D., & Torma, A. (2026). Index-based FT-IR assessment of chrysotile detectability and environmental durability in asbestos cement materials. SCIENTIFIC REPORTS, 1, 1–32. http://doi.org/10.1038/s41598-026-49524-w (**Q1**)
- Macher, G. Z., Bódizs, D., Sipos, D., & Schmeller, D. (2026). Asbestos Poverty as a New Paradigm for Multidimensional Urban Sustainability. JOURNAL OF URBAN HEALTH-BULLETIN OF THE NEW YORK ACADEMY OF MEDICINE, 1(1), 1–15. http://doi.org/10.1007/s11524-026-01063-5 **(D1)**
- Macher, G. Z., Torma, A., & Beke, D. (2025). The impact of asbestos cement pollution in irrigation water on physiological and germination characteristics of *Trifolium pratense, Medicago sativa*, and *Solanum lycopersicum* seeds. SCIENTIFIC REPORTS, 15(1). http://doi.org/10.1038/s41598-025-01011-4 **(Q1)**
- Macher, G. Z., Beke, D., & Torma, A. (2025). Water quality of harvested rainwater from asbestos cement roofs and its suitability for irrigation. CLEAN TECHNOLOGIES AND ENVIRONMENTAL POLICY, Recent Progress in Controlling

Pollution, Promoting Sustainable Circularity, and Aiding Resilient Farming(1), 1. http://doi.org/10.1007/s10098-025-03197-7 **(Q1)**

- Macher, G. Z. (2025). The Impact of Rainwater Quality Harvested from Asbestos Cement Roofs on Leaf Temperature in *Solanum lycopersicum* as a Plant Water Stress Indicator. WATER, 17(14). http://doi.org/10.3390/w17142070 **(Q1)**
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- Macher, G. Z., Torma, A., & Beke, D. (2025). Examining the Environmental Ramifications of Asbestos Fiber Movement Through the Water–Soil Continuum: A Review. INTERNATIONAL JOURNAL OF ENVIRONMENTAL RESEARCH AND PUBLIC HEALTH, 22(4). http://doi.org/10.3390/ijerph22040505 **(Q1)**
- Macher, G. Z., & Beke, D. (2025a). Mosses in Urban Environments as Passive Biofilters and Organisms Impacted by Asbestos-Contaminated Habitats. INTERNATIONAL JOURNAL OF ENVIRONMENTAL RESEARCH AND PUBLIC HEALTH, 22(6). http://doi.org/10.3390/ijerph22060838 **(Q1)**
- Macher, G. Z., Károly, F., Teh Boon Sung, C., Beke, D., Torma, A., & Gergely, S. (2024). Spectroscopic Analysis of Chrysotile Asbestos and its Environmental Resistance in Asbestos Cement Waste Products. PERTANIKA JOURNAL OF SCIENCE AND TECHNOLOGY, 32(6), 2441–2458. http://doi.org/10.47836/pjst.32.6.03 **(Q3)**

– Macher, G. Z., Torma, A., & Beke, D. (2024). Analysing the Environmental Durability and Chrysotile Content of Asbestos Cement Products by FT-IR Spectroscopy. CHEMICAL ENGINEERING TRANSACTIONS, 114, 709–714. http://doi.org/10.3303/CET24114119 **(Q3)**
– Gergely, Z. M., Judit, P., & Dóra, S. (2023). The Dissonances of Asbestos Exposure and the Concept of Sustainable Settlements in the Light of the European Union's 2023 Asbestos Neutrality Objective. CHEMICAL ENGINEERING TRANSACTIONS, 107, 145–150. http://doi.org/10.3303/CET23107025 **(Q3)**
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– Macher, G. Z., & Beke, D. (2024c). The impact of irrigation with harvested rainwater containing asbestos cement matrix on the germination characteristics of *Solanum lycopersicum*. ACTA PHYTOPATHOLOGICA ET ENTOMOLOGICA HUNGARICA, 59(1), 63–73. http://doi.org/10.1556/038.2024.00207
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– Macher, G. Z., Torma, A., & Beke, D. (2023a). A települési és mezőgazdasági vízgazdálkodás azbeszt okozta kockázati tényezői a

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