



© 2025. The Author(s). This is an open-access article distributed under the terms of the Creative Commons Attribution-ShareAlike 4.0 International Public License (CC BY SA 4.0, <https://creativecommons.org/licenses/by-sa/4.0/legalcode>), which permits use, distribution, and reproduction in any medium, provided that the article is properly cited.

Environmental impact of waste-derived geopolymer composites: heavy metal leaching and risk assessment

Kamila Mizerna^{1*}, Anna Król¹, Elżbieta Janowska-Renkas², Agnieszka Kaliciak-Kownacka²

¹Department of Process and Environmental Engineering, Opole University of Technology, Opole, Poland

²Department of Building Materials Engineering, Opole University of Technology, Opole, Poland

*Corresponding author's e-mail: k.mizerna@po.edu.pl

Keywords: environmental risk assessment, geopolymer composites, silica fly ash, waste glass powder, heavy metals leaching

Abstract: Geopolymers are a relatively new type of material that can be produced from waste. They may contain hazardous compounds, such as heavy metals, which pose environmental risks if released. This study presents the results of heavy metal release from molded geopolymer composites over time and evaluates the leaching mechanisms of various elements. The study also assesses the potential ecological risk of these materials, highlighting the innovative and complex nature of the research program. The geopolymer composites were produced from silica fly ash (CFA) and waste glass powder (GP), with their composition further modified using graphene and nanosilica. The study investigated materials with innovative compositions that could effectively replace traditional Portland cement-based concrete, whose production significantly contributes to carbon emissions. Leachability was assessed using the tank test method. Among the ten metals analyzed, the geopolymer composites released Ba, Cr, Mo, and Sb. The study demonstrated that the leaching process was primarily controlled by dissolution and diffusion; however, for Ba and Mo, depletion of available ions for leaching was also observed. The potential Ecological Risk Index (PERI) ranged from 21.4 to 34.5 depending on the geopolymer composition. The ecological risk analysis indicated no environmental threat from the geopolymer composites.

Introduction

One of the by-products of solid fuels combustion for energy purposes is silica fly ash. Due to its good pozzolanic properties, this material serves as a valuable raw material for cement and concrete production (Nayak et al. 2022, Król 2016). Fly ash is rich in silica and alumina, making it an excellent source material for geopolymer production (Sitarz-Palczak et al. 2019, Tho-In et al. 2018). A geopolymer is a three-dimensional, alkali-activated, cross-linked material, characterized by a stable physical structure and chemical properties. Its source materials are aluminosilicate-rich compounds. In the literature, geopolymers are often considered a potential alternative to Portland cement-based materials (Bao et al. 2021, Tan et al. 2022). Due to their unique properties, such as thermal and acid resistance, high mechanical strength, and the ability to immobilize heavy metals, geopolymers are an intriguing subject of research. From an environmental perspective, their capacity to immobilize heavy metals is particularly important, as it is linked to their ion exchange ability and large specific surface area (Król et al. 2018). Additionally, geopolymer concrete can contribute to reducing CO₂ emissions from the cement industry (Sun and Vollpracht 2020).

Geopolymers are a relatively new class of materials that can be synthesized from metakaolin, natural minerals, and various waste materials, such as slag, fly ash, red mud and volcanic tuff (Sitarz-Palczak et al. 2019). The incorporation of waste into geopolymer composites introduces the possibility of hazardous compounds, especially heavy metals. Environmental contamination by heavy metals is a major global concern due to its potential risks to human health and the soil-water environment (Zaynab et al. 2022). Mobile forms of heavy metals in waste-derived materials can leach into the environment, causing contamination. Therefore, assessing metal leachability is essential in determining the potential environmental impact of such materials (Faragó et al. 2023, Król and Mizerna 2016). The leaching process can occur through various mechanisms, including diffusion, surface wash-off, and dissolution (Hartwich and Vollpracht 2017). The tank test (EA NEN 7375) is a standardized method for evaluating leaching mechanisms in molded or monolithic materials, enabling the analysis of heavy metal release levels over an extended period. One of the key factors influencing metal leaching is the contact time between the material and the leaching. This method supports decision-making regarding the use of engineering materials in the soil-water environment and helps mitigate potential negative impacts.

A widely recognized method for assessing the potential risk of soil contamination by heavy metals is the calculation of the Potential Ecological Risk Index (PERI). Developed by Swedish scientist Håkanson, PERI is a comprehensive assessment tool that considers both the total concentration of heavy metals and their respective toxicity coefficients (Liu et al. 2021). PERI has been extensively used in evaluating the potential ecological risks associated with heavy metals in river and marine sediments, agricultural and industrial soils, and sewage sludge with potential environmental applications (Kowalik et al. 2019, Liu et al. 2021). In addition to PERI, several other indicators are commonly used to assess contamination levels, ecological effects, and the environmental impact of heavy metals. These include Potentially Toxic Elements (PTEs) (Rachwał et al. 2024), the Geoaccumulation Index (I_{geo}) (Baran and Wieczorek 2015, Yandem and Jabłońska-Czapla 2024), Predicted Environmental Concentration (PEC), and Threshold Effect Concentration (TEC) (Rozpondek and Rozpondek 2019). In this study, the authors applied PERI to assess the potential ecological risk of heavy metals in geopolymer composites, an innovative construction material that may also be introduced into soils.

The authors decided to produce geopolymer composites with an innovative composition, by partially replacing siliceous fly ash (CFA) with waste glass powder from monitor screens (GP). This decision was driven by the high demand for fly ash in the construction industry, which has led to its limited availability on the market, as well as by the opportunity to incorporate alternative waste materials into the geopolymer matrix - aligning with the principles of a circular economy. It is known that in concrete mixtures, the addition of glass powder enhances compressive strength due to its good pozzolanic properties (Tho-In et al. 2018). Therefore, waste glass powder was incorporated into the geopolymer composite. Additionally, nanosilica was introduced as a nano-additive to improve the mechanical properties. These materials contribute to increased compressive and flexural strength, as well as the modulus of elasticity, by forming gel-like reaction products such as hydrated calcium silicates (C-S-H), hydrated calcium aluminosilicates

(C-A-S-H), and hydrated sodium aluminosilicates (N-A-S-H) (Phoo-ngernkham et al. 2014). Nanoparticles integrate into geopolymer structure, positively influencing microstructure modification (Abbasi et al. 2016). Due to their extremely small size and high surface-to-volume ratio, they contribute to the development of advanced materials with improved rheological and mechanical properties, including shape memory behavior (Sobolev et al. 2009). Additionally, according to Shamsol et al. (2024), graphene further enhances the physical and mechanical properties of geopolymers.

The aim of the study was to analyze changes in the leaching levels of heavy metals (Ba, Cd, Co, Cr, Cu, Mo, Ni, Pb, Sb, and Zn) from geopolymer composites over time and to assess the potential ecological risk associated with selected heavy metals in the samples. The research evaluated the mechanisms controlling the release of heavy metals and the impact of geopolymer composites on the soil-water environment. The authors highlight the composition of geopolymers as a novel aspect of research on sustainable materials for engineering applications. The study was based on the assumption that even when incorporating two types of waste into geopolymers, it is possible to produce an environmentally safe composite.

Materials and methods

Characterization of research materials

The primary materials used to create the geopolymer composites were silica fly ash (CFA) from coal combustion and glass powder (GP) derived from waste monitor screens. These waste materials were selected based on their high SiO₂ content (60.2% in CFA and 30.5% in GP) and CaO content (23.2% in CFA and 33.5% in GP) (Janowska-Renkas et al. 2023). **Geopolymer composites were prepared with varying proportions of CFA and GP, along with the addition of nanosilica.** Two types of colloidal nanosilica were used: a hydrophilic nanosilica with a nanosilver additive in aqueous solution (CHI) and a colloidal hydrophobic nanosilica dissolved in ethanol and isopropanol (CHF). Additionally, the composition of the geopolymer mortars was modified with

Table 1. Component materials of geopolymer composites

[illegible]

graphene (Gh). The formulations of all studied composites are presented in Table 1. The geopolymer composites were synthesized using an alkaline solution containing sodium silicates and an 8-molar sodium hydroxide (NaOH) solution.

The synthesis of geopolymer materials was conducted in a water bath rack at a temperature of 85°C. After 24 hours, the samples were removed from the molds and stored at a humidity of 16±2% and a temperature of 20±2°C.

Table 2 presents the characterization of the geopolymer composites analyzed in this study in terms of heavy metal content. As the CFA content in the geopolymer composition decreased, the heavy metal concentration also declined. The composites contained of heavy metals in the following descending order: Ba>Zn>Ni>Cr/Cu>Pb>Co>Sb>Mo>Cd.

Leaching method – tank test

For monolithic samples of geopolymer composites, heavy metal leaching tests were carried out based on the tank test, as specified in the EA NEN 7375 standard “Leaching characteristics of moulded or monolithic building and waste materials. Determination of leaching of inorganic components with the diffusion test. The tank test”. This test enabled the assessment of changes in heavy metal leaching over an extended period (64 days) and provided insight into the processes governing the release of individual elements. The tank test procedure is illustrated in Figure 1. The samples were placed on supports inside glass containers and fully immersed in deionized water with a conductivity of <10 µS/cm and a neutral pH. It was crucial to ensure that each sample was positioned at least 2 cm from the container walls and completely surrounded by the leaching liquid on all sides.

Eluates were collected periodically over the 64-day period (8 fractions), pressure-filtered through a membrane filter with a pore diameter of 0.45 µm, and analyzed for pH and electrical conductivity (EC). After each eluate collection, the liquid was completely replaced, and the samples were re-immersed.

According to the procedure, the solubility of the research material matrix was evaluated based on criteria 1 (Eq. 1) and 2 (Eq. 2), which depend on the pH and electrical conductivity of the eluates. If criterion 1 is not met, the matrix is considered insoluble, and leachability can be assessed using the tank test. However, if criterion 1 is met, criterion 2 must then be evaluated. For the research materials, it was determined that matrix dissolution does not occur, allowing for the assessment of heavy metal leachability.

$$S_{7-8} > 1,5 \cdot \frac{V_p}{V} + 10^{pH_{7-8}-11.78} + 10^{2.5-pH_{7-8}} \quad (1)$$

$$S_{7-8} > 2 \cdot S_{5-6} \quad (2)$$

where: S_{7-8} – average value of the measured EC in periods 7 and 8 (µS/cm), S_{5-6} – average value of the measured EC in periods 5 and 6 (µS/cm), pH_{7-8} – average pH value in periods 7 and 8, V – volume of the leaching liquid (dm³), V_p – volume of the tested sample (dm³).

Analysis of heavy metals and quality control

To determine the total heavy metal content, sample decomposition in concentrated acids was performed in accordance with EN ISO 54321. Mineralization was carried out in a closed system using a Magnum II microwave oven (Ertec). The total heavy metal content in geopolymers and their concentrations in eluates were analyzed using ICP-OES (Inductively Coupled Plasma Optical Emission Spectrometry) with an Agilent 5800 spectrometer, following the EN ISO 11885 standard. Ten elements were evaluated: Ba, Cd, Co, Cr, Cu, Mo, Ni, Pb, Sb, and Zn. To ensure analytical precision and quality control, the analysis included triplicate measurements, blank checks, and analysis of Certified Reference Material (CPAchem, Lot: 754042) after every 20th, 40th, and 60th sample for verification. The recovery rates for the analyzed heavy metals ranged from 99–114%, and the accuracy of the

Table 2. Total content of heavy metal in geopolymer composites and their waste component

Sample	Total content of heavy metals (mg/kg)									
	Ba	Cd	Co	Cr	Cu	Mo	Ni	Pb	Sb	Zn
CFA	1059 ± 51	1.2 ± 0.1	12.3 ± 0.5	57.6 ± 2.0	53.0 ± 2.2	5.0 ± 0.3	44.8 ± 1.7	49.2 ± 1.4	6.3 ± 0.2	101 ± 5
GP	1.9 ± 0.1	<0.25	<0.30	0.29 ± 0.02	0.42 ± 0.03	0.24 ± 0.01	0.30 ± 0.02	<0.57	0.78 ± 0.06	1.6 ± 0.1
D1	186 ± 9	0.36 ± 0.01	4.6 ± 0.2	18.6 ± 0.6	19.4 ± 0.8	1.5 ± 0.2	19.5 ± 0.9	15.2 ± 0.3	2.5 ± 0.2	37.1 ± 1.9
D1CHI	198 ± 8	0.32 ± 0.02	4.8 ± 0.3	19.2 ± 0.9	17.7 ± 0.6	1.5 ± 0.2	18.7 ± 0.8	15.6 ± 0.4	2.3 ± 0.1	37.4 ± 1.8
D1CHF	185 ± 9	0.29 ± 0.02	4.3 ± 0.4	15.0 ± 0.7	16.8 ± 0.5	1.4 ± 0.1	17.3 ± 0.6	15.7 ± 0.6	2.5 ± 0.1	32.7 ± 1.3
D2	142 ± 6	<0.25	3.3 ± 0.1	11.4 ± 0.4	12.9 ± 0.5	1.1 ± 0.1	14.0 ± 0.4	9.1 ± 0.2	1.9 ± 0.2	26.9 ± 1.3
D2CHI	156 ± 7	0.26 ± 0.01	4.0 ± 0.2	16.3 ± 0.5	14.0 ± 0.6	1.0 ± 0.0	17.8 ± 0.6	10.4 ± 0.6	1.7 ± 0.1	33.4 ± 1.5
D2CHF	158 ± 5	<0.25	3.7 ± 0.3	13.9 ± 0.7	13.9 ± 0.8	1.2 ± 0.1	14.9 ± 0.5	10.5 ± 0.5	2.3 ± 0.2	28.8 ± 0.8
D3	101 ± 5	<0.25	2.4 ± 0.1	8.0 ± 0.3	9.7 ± 0.3	0.7 ± 0.0	9.8 ± 0.2	8.7 ± 0.3	1.5 ± 0.1	19.9 ± 1.0
D3CHI	164 ± 8	0.25 ± 0.01	4.0 ± 0.3	19.2 ± 0.9	15.0 ± 0.8	1.1 ± 0.1	16.9 ± 0.8	10.4 ± 0.4	2.5 ± 0.3	33.5 ± 1.2
D3CHF	88.0 ± 3.1	<0.25	2.4 ± 0.1	10.9 ± 0.6	8.2 ± 0.2	1.5 ± 0.1	18.7 ± 0.7	4.6 ± 0.1	1.5 ± 0.1	19.8 ± 0.9

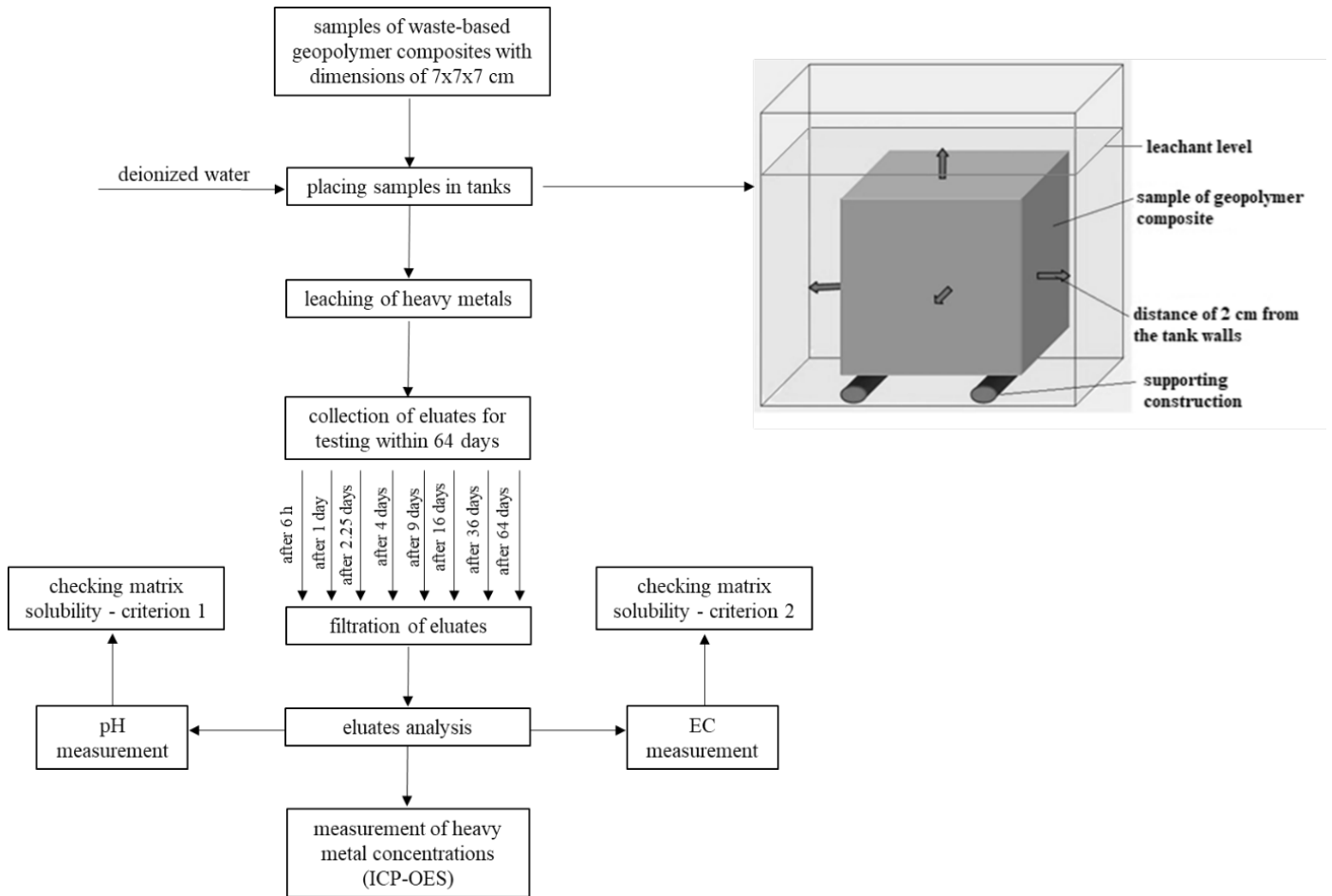


Figure 1. Schematic diagram of the tank test procedure and method of placing the sample according to EA NEN 7375

ICP-OES analysis was within a relative standard deviation (RSD) of less than 5%.

Computational methods and determination of leaching mechanism

The release of heavy metals during the tank test and the cumulative leachability values were calculated according to the guidelines in the EA NEN 7375 standard. The leachability of each heavy metal per unit surface area was determined using the following formula:

$$E_i = \frac{C_i \cdot V}{A} \quad (3)$$

where: E_i – leachability of the heavy metal in eluate i (mg/m^2), C_i – concentration of the heavy metal in eluate i (mg/dm^3), V – volume of eluate (dm^3), A – surface area of the sample (m^2).

The cumulative leachability of individual heavy metals was determined as the sum of the leachability results for the 8 eluates collected over a period of 64 days:

$$E_n = \sum_{i=1}^n E_i \quad \text{for } n = 1 \text{ to } N \quad (4)$$

where: E_n – designated cumulative leachability of the heavy metal over period n , encompassing eluates $i = 1$ to n (mg/m^2), E_i – leachability of the heavy metal in eluate i (mg/m^2).

To determine the mechanisms controlling the heavy metal leaching, the obtained cumulative leachability (ε_n)

was calculated for all leaching periods from $n = 1$ to $N = 8$, according to Equation (5), taking into account the eluate exchange times.

$$\varepsilon_n = E_i \frac{\sqrt{t_i}}{\sqrt{t_i} - \sqrt{t_{i-1}}} \quad (5)$$

where: ε_n – obtained cumulative leachability of the heavy metal over period n , encompassing eluates $i = 1$ to n (mg/m^2), E_i – leachability of the heavy metal in eluate i (mg/m^2), t_i – time of eluate i collection, i.e., time at the end of leaching (s), t_{i-1} – time of collection of eluate $i - 1$, i.e., time at the beginning of leaching (s).

In the next stage, linear regression was applied to the $\log \varepsilon_n - \log t_i$ system (using base-10 logarithms) to determine the slope of the regression curve (r_c). According to the EA NEN 7375 standard, the r_c parameter indicates the dominant leaching mechanism within a given time interval (leaching stage). The heavy metal leaching mechanism was determined across all stages, covering the following sequences: 2-7, 5-8, 4-7, 3-6, 2-5, and 1-4. In each interval, the slope of the linear regression line was calculated (Table 3). Additionally, the concentration factor was calculated as the ratio of the average metal concentration during a specific leaching stage to its detection limit. This factor was checked to ensure it was ≥ 1.5 , as such values allow for determining the leaching mechanism by diffusion. If this criterion was not met, the mechanism could

Table 3. Heavy metal leaching mechanisms based on r_c parameter

Leaching stage (increment)	$r_c \leq 0.35$	$0.35 < r_c \leq 0.65$	$r_c > 0.65$
2 – 7	Surface wash-off (SWO)	Diffusion (DIF)	Dissolution (DIS)
5 – 8	Depletion (DEP)	Diffusion (DIF)	Dissolution (DIS)
4 – 7	Depletion (DEP)	Diffusion (DIF)	Dissolution (DIS)
3 – 6	Depletion (DEP)	Diffusion (DIF)	Dissolution (DIS)
2 – 5	Depletion (DEP)	Diffusion (DIF)	Dissolution (DIS)
1 – 4	Surface wash-off (SWO)	Diffusion (DIF)	Delayed diffusion or dissolution (DD/DIS)

not determine (which occurred in only three cases).

Method for assessing potential ecological risk

The Potential Ecological Risk Index (PERI) provides information about the potential contamination of soils by heavy metals. It was calculated using formula (6) (Kazapoe et al. 2022, Kowalik et al. 2019) for those heavy metals for which toxicity coefficients and reference background values are available in the literature.

(6)

$$PERI = \sum_{i=1}^n E_r^i = \sum_{i=1}^n (C_f^i \cdot T_r^i) = \sum_{i=1}^n \left(\frac{C_D^i}{C_R^i} \cdot T_r^i \right)$$

where: PERI – total potential ecological risk index; E_r^i – potential ecological risk index for the heavy metal i ; C_f^i – contamination factor for the heavy metal i ; T_r^i – toxicity coefficient for the heavy metal i , with values set to 30, 5, 2, 5, 5, 5, and 1 for Cd, Co, Cr, Cu, Ni, Pb, and Zn, respectively (Kazapoe et al. 2022, Liu et al. 2021, Tomczyk et al. 2023); C_D^i – content of the heavy metal i in geopolymer composites (mg/kg); C_R^i – background reference value for heavy metals

(mg/kg), with values set to 0.5, 40, 15, 25, 25, 50, and 70 for Cd, Co, Cr, Cu, Ni, Pb, and Zn, respectively. Heavy metals background levels in soils in Poland were reported by Jaskuła et al. (2021) and Tomczyk et al. (2023).

The E_r^i value is divided into five levels of ecological risk: <40 – low, 40-80 – moderate, 80-160 – considerable, 160-230 – high, and >320 – very high. The PERI index (total risk level) is divided into four levels: <150 – low, 150-300 – moderate, 300-600 – high, and >600 – very high (Movafagh et al. 2020, Tomczyk et al. 2023).

Results and discussion

Analysis of long-term heavy metal release from geopolymer composites

Leaching tests were conducted on monolithic composite samples using the tank test, allowing for the assessment of heavy metal release over an extended period (64 days) and an analysis of cumulative leachability per unit surface area of the composite. This study also facilitated the identification of

Table 4. Cumulative leachability of heavy metal in tank test

Sample	Cumulative leaching E_n (mg/m ²)									
	Ba	Cd	Co	Cr	Cu	Mo	Ni	Pb	Sb	Zn
D1	2.78 ± 0.13	<0.07	<0.08	2.69-2.74 ± 0.13	<0.08	18.83 ± 0.93	<0.08	<0.16	2.18-2.40 ± 0.09	<0.54
D1CHI	5.33 ± 0.27	<0.07	<0.08	2.53-2.58 ± 0.12	<0.08	16.65 ± 0.67	<0.08	<0.16	1.82-2.15 ± 0.07	<0.54
D1CHF	2.64 ± 0.12	<0.07	<0.08	2.72-2.77 ± 0.12	<0.08	19.27 ± 0.92	<0.08	<0.16	2.10-2.43 ± 0.09	<0.54
D2	3.78 ± 0.18	<0.07	<0.08	2.18-2.23 ± 0.10	<0.08	16.49 ± 0.58	<0.08	<0.16	2.80-3.08 ± 0.11	<0.54
D2CHI	3.92 ± 0.20	<0.07	<0.08	2.39-2.44 ± 0.11	<0.08	16.68 ± 0.54	<0.08	<0.16	2.80-3.16 ± 0.12	<0.54
D3CHF	4.84 ± 0.19	<0.07	<0.08	2.31-2.36 ± 0.10	<0.08	15.73 ± 0.48	<0.08	<0.16	2.07- 2.37 ± 0.09	<0.54
D3	11.01 ± 0.55	<0.07	<0.08	2.12-2.17 ± 0.12	<0.08	6.05 ± 0.29	<0.08	<0.16	1.95-2.39 ± 0.08	<0.54
D3CHI	5.88 ± 0.27	<0.07	<0.08	2.72-2.77 ± 0.09	<0.08	19.40 ± 0.73	<0.08	<0.16	2.37-2.88 ± 0.10	<0.54
D3CHF	5.17 ± 0.16	<0.07	<0.08	2.56 ± 0.11	<0.08	18.29 ± 0.65	<0.08	<0.16	2.48 ± 0.12	<0.54

mechanisms governing heavy metal leachability, particularly determining whether diffusion is the dominant process.

Cumulative leaching

Table 4 presents the cumulative leachability for each heavy metal (E_n) measured from the start of the study through to the collection of the final eluate fraction (64 days). When the concentration of a given element was below the detection limit, the cumulative leachability result for the entire test was presented as a range (this applies to Cr and Sb).

Among the analyzed heavy metals, only four showed leaching above the detection limit: Mo, Ba, Cr, and Sb. Molybdenum exhibited the highest leachability, except in the case of sample D3 (50% CFA + 50% GP + 0.5% Gh, which uniquely showed the highest leachability of barium. Although barium is not considered a trace element, it is typically found in higher concentrations in raw materials and waste products and has relatively low toxicity (Overmann et al. 2021).

In prior compressive strength tests of geopolymer composites carried out by the authors (Janowska-Renkas et al. 2023), sample D3 exhibited the lowest compressive strength. This was attributed to the high content of waste glass powder with an amorphous structure, which reacts less readily in a geopolymer matrix. The dissolution of components by OH groups leads to the formation of $\text{Si}(\text{OH})_4^-$, $\text{Al}(\text{OH})_4^-$, $\text{Al}(\text{OH})_6^{3-}$ complexes. Alumina groups form first, as aluminum dissolves more readily than silicon due to weaker Al-O bonds, which break more easily than Si-O bonds. Condensation initially occurs in the presence of aluminates, which are three times more prevalent in fly ash than in glass powder. Increasing the Si/Al ratio significantly enhances the strength and density of the microstructure, with the optimal geopolymer properties achieved at a ratio of 2.5. However, this approach applies at the nanoscale. At the microstructural scale, condensation and crystallization form a dense matrix in the form of the N-A-S-H (sodium aluminosilicate hydrate) phase, which is present in all tested samples. However, in sample D3, a greater quantity of undissolved glass particles was observed, as OH groups failed to break their bonds, indicating the need for further exploration of the appropriate molar ratios to achieve 100% dissolution of the components.

Heavy metals such as Cd, Co, Cu, Ni, Pb, and Zn were fully immobilized in the composite matrices. The immobilization mechanism of these heavy metal cations likely involves chemical binding (through atomic substitutions), physical absorption, or encapsulation (Toniolo et al. 2018). However, the incomplete immobilization of Mo and Cr results from these elements being heavy metal anions, which are not as readily immobilized as cations (Kuterasińska-Warwas and Król 2017). This is attributed to insufficient amounts of calcium sulfoaluminates, which bind Cr and Mo into their structures by replacing sulfate groups (Król 2020). According to Ji and Pei (2020), anion immobilization may also be inhibited by the negative charge of the tetrahedral $[\text{AlO}_4]^-$ structure. Nevertheless, researchers emphasize that the immobilization mechanisms of individual ions are not fully understood, and new geopolymer matrices should be studied in detail.

It is known that hexavalent chromium is permanently bound through the substitution of the sulfate group in ettringite structures ($\text{C}_3\text{A} \cdot 3\text{CaCrO}_4 \cdot 32\text{H}_2\text{O}$). A similar reaction, although

to a lesser extent, can occur with hydrated calcium monosulfate aluminate ($\text{C}_3\text{A} \cdot \text{CaCrO}_4 \cdot 12\text{H}_2\text{O}$) (Takahashi et al. 2003). The substitution of SO_4^{2-} by CrO_4^{2-} is feasible, but it is limited to high oxidation states of chromium. Consequently, chromium immobilization is difficult to achieve and typically does not exceed 80% (Bakhshi et al. 2019, Król 2020). Furthermore, Bakhshi et al. (2019) suggest that the relatively lower binding of hexavalent chromium may be due to its tendency to form highly soluble chromates. In contrast, Cr(III) is incorporated into the structures of C-S-H (hydrated calcium silicate) phases, where it substitutes other elements. This reaction helps stabilize the disturbed structure of the C-S-H phase and inhibits its transition into crystalline phases (Van Der Sloot 2000).

Molybdenum exists only in a single oxidation state, Mo(VI), and forms a stable MoO_4^{2-} group with oxygen. The ionic sizes increase in the order $\text{SO}_4 < \text{CrO}_4 < \text{MoO}_4$. Thus, the replacement of sulfate by MoO_4 in calcium sulfoaluminate structures, such as ettringite, is even less likely than for chromium. The chemical binding of molybdenum is as limited as that of chromium (Van Der Sloot 2000).

Despite the release of metal ions into the aqueous phase, their concentrations are much lower than those observed in studies of traditional cement-based construction materials. In the paper by Overmann et al. (2021), the authors reported cumulative barium leachability levels for these materials in the range of 50-130 mg/m². This applies to monolithic cement-based concrete samples, including CEM I with varying amounts of fly ash or silica fume as additives, as well as CEM II and CEM III. For chromium, researchers observed concentrations similar to those in this study (between 2-3 mg/m²). It is noteworthy, however, that chromium leachability studies on concrete samples were conducted after a preliminary reduction of chromates.

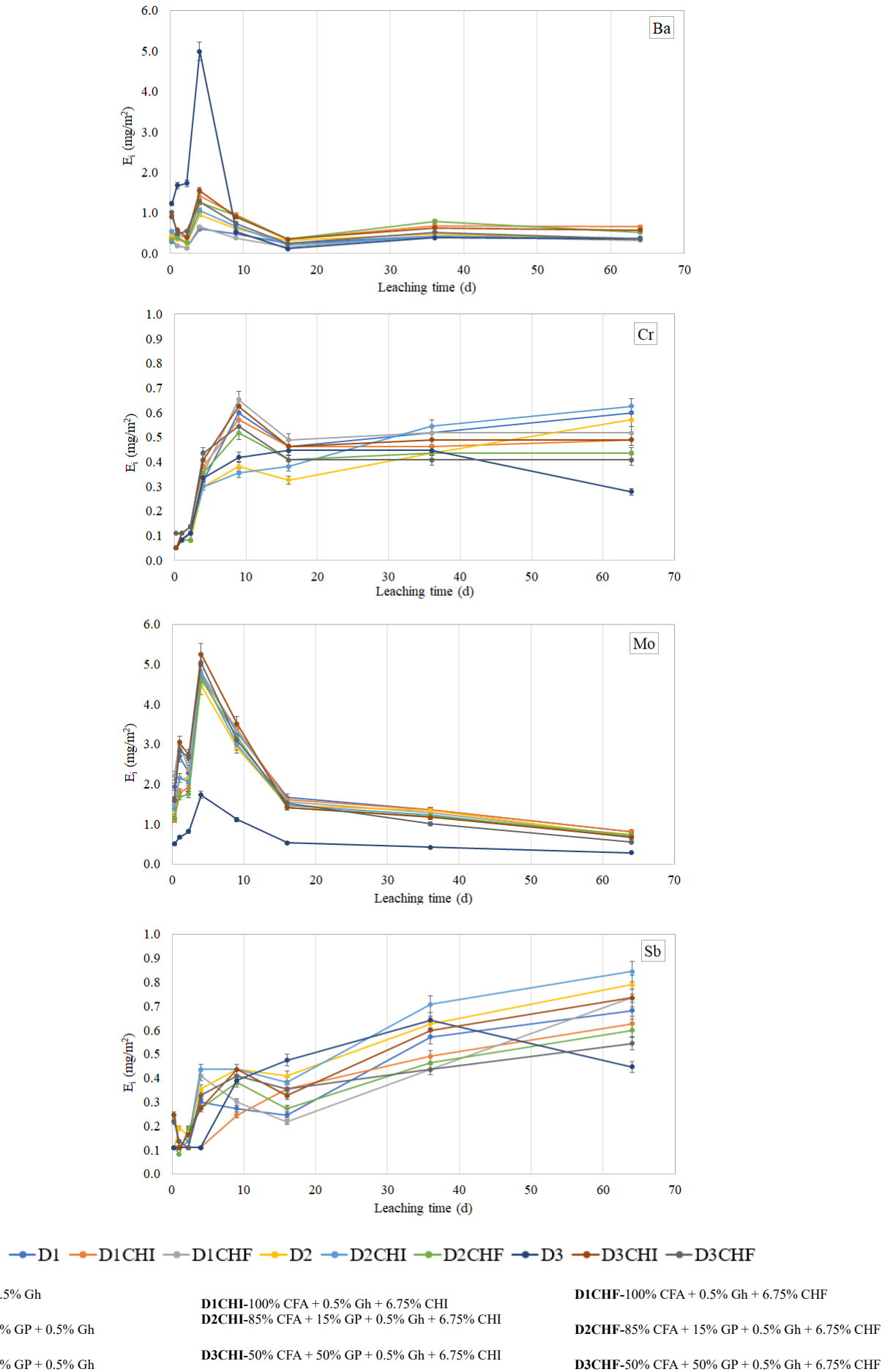
Changes in leachability over time

Four heavy metals that were released from geopolymers in the tank test were analyzed. The results of heavy metal leachability (E_i) in 8 eluates relative to the duration of the test (in days) are presented in the graphs (Figure 2).

A continuous increase in chromium leachability was observed for the D1, D1CHI, D1CHF, D2, D2CHF, D3CHI, and D3CHF samples up to the 9th day of the test, with the highest leached concentrations occurring on that day. After this point, these samples showed stabilized chromium release levels, except for the D2 and D2CHI samples. The D2 composite exhibited a continuous increase in leachability after the 16th day of the test, while D2CHI showed an increase starting on the 4th day. It can be assumed that, if the test duration had been extended, the mobile forms of chromium in these two samples would have been completely released into the aqueous phase.

A significant drop in chromium leachability was observed for the D3 sample after 36 days. This sample was the only one that exhibited a slightly different trend in leaching compared to the other geopolymers. The initial increase in chromium release levels during the early stages of leaching may be attributed to the presence of easily soluble Cr(VI) compounds in the raw materials, particularly in fly ash, such as K_2CrO_4 or $\text{K}_2\text{Cr}_2\text{O}_7$ (Overmann et al. 2021).

For antimony, while other samples showed increased leachability in the later stages of the test, D3 demonstrated a



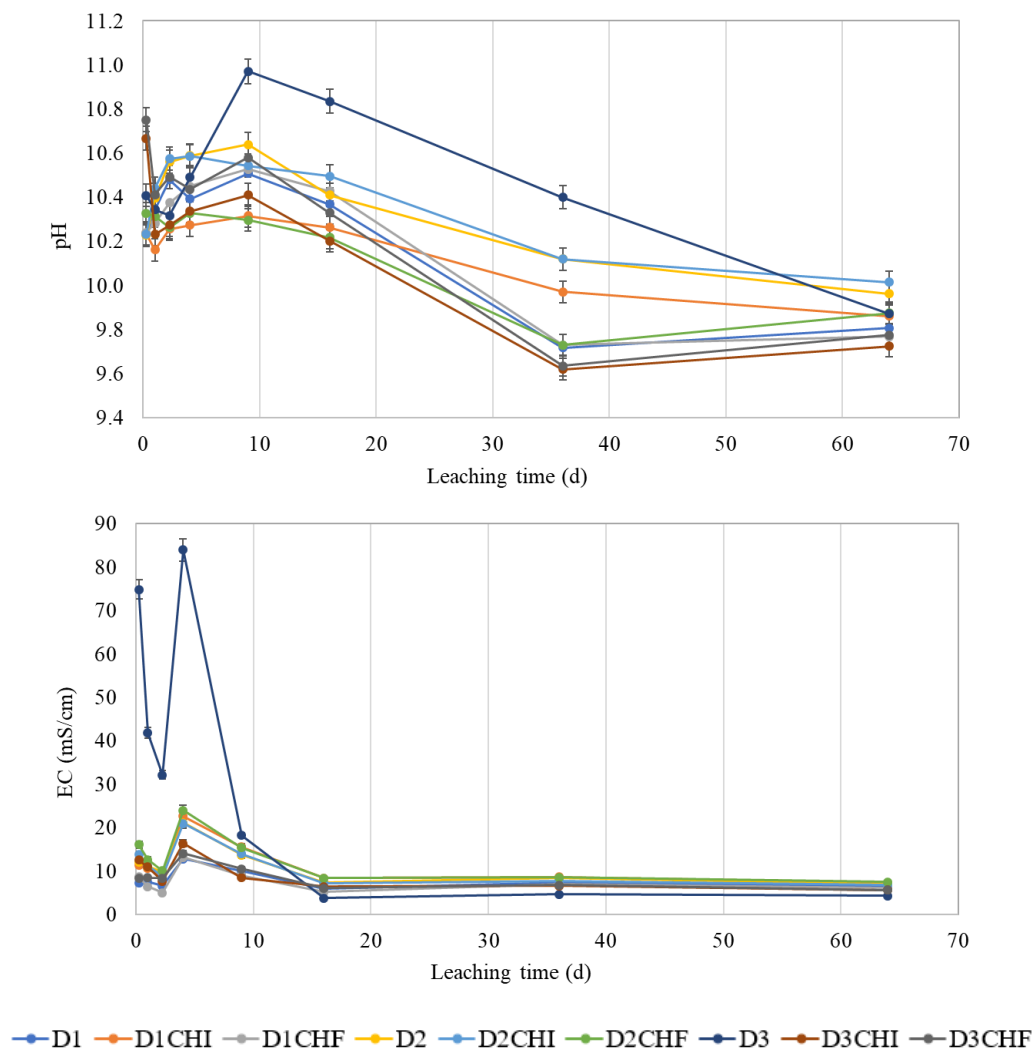


Figure 3. Values of pH and EC of eluates from geopolymers composites

decrease in leachability, indicating that the available Sb ions for leaching were being depleted. A similar depletion process was observed for molybdenum, with concentrations beginning to decrease after reaching a maximum on the 4th stage of the test (the 4th day). By the end of the procedure, sample D3 showed a 30% reduction, while the other samples exhibited a reduction of 78-95%. For barium, leachability decreased during the first three stages of the test, after which the element showed an increase in leachability, reaching its maximum concentrations on the 4th day.

Figure 2 clearly shows that sample D3 exhibited significantly higher leachability of Ba at this stage. The leachability recorded at this stage was 4.99 mg/m², which constitutes 45% of the cumulative leachability. The increase in barium leaching into the solution may be attributed to a decrease in sulfate concentrations, as barium solubility is controlled by BaSO₄ (Vollpracht and Brameshuber 2016). A similar trend of initial decline followed by a temporary increase in barium leaching was also observed in the tank test studies by Müllauer et al. (2015) studies.

Changes in pH and electrical conductivity of eluates

The pH parameter plays a crucial role in the release of anions and cations of heavy metals. Another important parameter in the

evaluation of eluates from leaching tests is electrical conductivity (EC), which indicates the mobility of ions from the monoliths into the aqueous phase. The pH of all eluates obtained from the tested composites was alkaline (Figure 3), with values ranging from 9.62 to 10.97. After the 9th day of the test, a decrease in pH was observed. The highest EC values were recorded on the 4th day of the test for all composites, reaching 84.0 mS/cm for D3, while for the remaining samples, they ranged from 12.8 to 23.9 mS/cm. Over the next 12 days of the tank test, conductivity decreased and stabilized at around 4.3 to 8.5 mS/cm after the 16th day. Notably, the leachability trends for Ba and Mo corresponded with the changes in EC.

The slight increase in pH during the initial stages of leaching is associated with an increase in hydroxide ion concentrations [OH⁻] in the eluates. The subsequent decrease in pH is due to the carbonation process, which is a natural phenomenon and is also influenced by the exchange of leaching solution with neutral pH after each stage of the test. For the same reason, electrical conductivity decreases as well. Similar pH trends were observed in concrete studies by Overmann et al. (2021), though this is not always the case. In instances where the researchers used airtight covers to prevent external air access, pH increased throughout the entire leaching period (Overmann et al. 2021). It should be emphasized that limiting air access

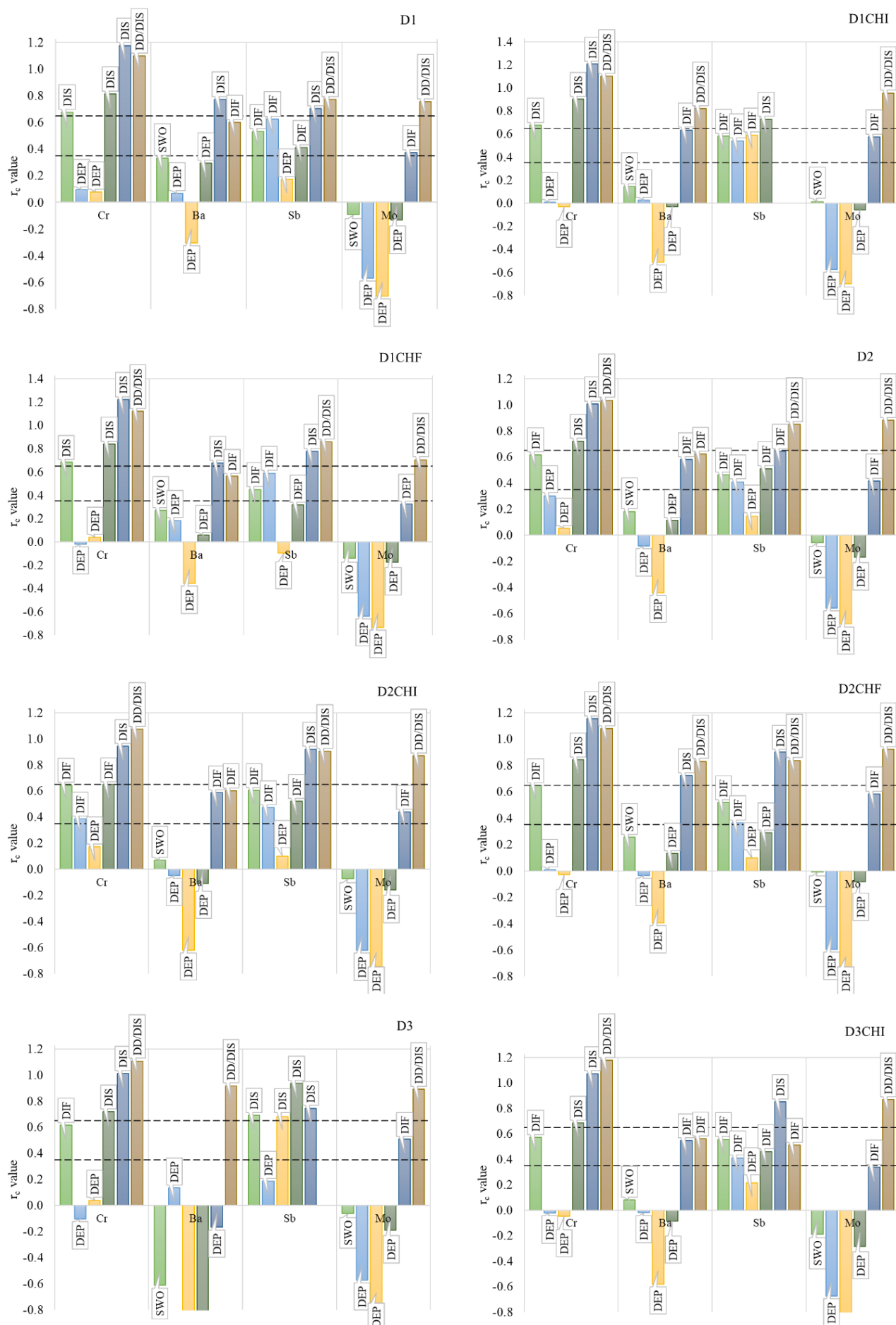
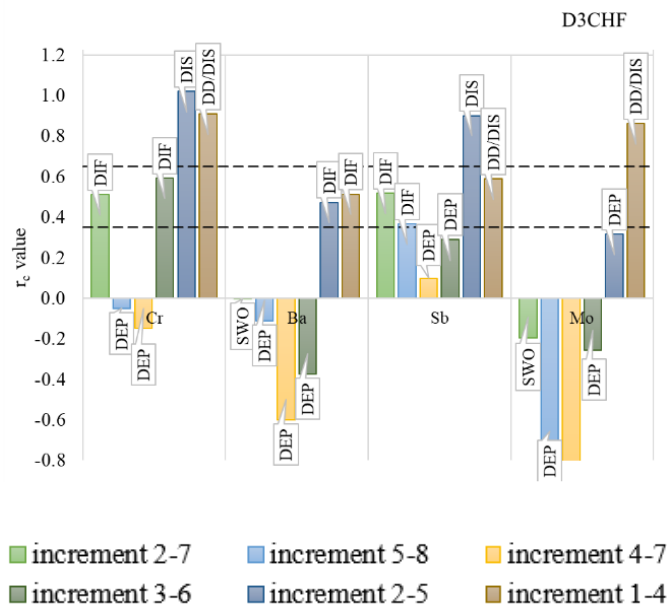


Figure 4.



D1-100% CFA + 0.5% Gh

D2-85% CFA + 15% GP + 0.5% Gh

D3-50% CFA + 50% GP + 0.5% Gh

D1CHI-100% CFA + 0.5% Gh + 6.75% CHI

D2CHI-85% CFA + 15% GP + 0.5% Gh + 6.75% CHI

D3CHI-50% CFA + 50% GP + 0.5% Gh + 6.75% CHI

D1CHF-100% CFA + 0.5% Gh + 6.75% CHF

D2CHF-85% CFA + 15% GP + 0.5% Gh + 6.75% CHF

D3CHF-50% CFA + 50% GP + 0.5% Gh + 6.75% CHF

Figure 4. Leaching mechanisms of heavy metals (DIS – dissolution, DEP – depletion, DIF – diffusion, SWO – surface wash-off, DD/DIS – delayed diffusion or dissolution)

is a forced action, but it provides valuable insight: in an open experimental system, CO₂ absorption from the atmosphere may occur, leading to a decrease in pH and, consequently, an increase in the solubility of heavy metals.

Determination of leaching mechanisms

In addition to evaluating the leaching characteristics of heavy metals from geopolymers, the leaching mechanisms were also investigated. These mechanisms were identified over various time intervals (increments) covering the following sequences: 2-7, 5-8, 4-7, 3-6, 2-5, and 1-4. The 2-7 increment was considered the “total growth interval” for the entire test, excluding the first and last eluate fractions to avoid interpretational errors. The first fraction was eliminated due to the influence of the flushing process, while the last fraction was excluded to prevent potential depletion of the element during the test. This study enabled the determination of whether the leaching process was controlled by diffusion (DIF), dissolution (DIS), surface wash off (SWO), depletion (DEP), or delayed diffusion or dissolution (DD/DIS). The leaching mechanism was determined based on the slope of the regression curve (r_e) obtained, as shown in the graphs (Figure 4).

Due to the low concentrations of the analyzed elements, particularly Sb and Cr, the assessment of the processes controlling leaching was not conclusive. However, an attempt was made to identify the dominant mechanisms. It was observed that the Cr leaching from samples containing 100% fly ash (D1, D1CHI, D1CHF) was dominated by dissolution. For the remaining composites (with reduced CFA content), dissolution was dominant only in the initial stages of leaching, while diffusion played a greater role later on, suggesting that Cr was mainly present in ionic

form. For Ba, leaching in the initial stages was driven by diffusion or dissolution, whereas depletion of available Ba ions occurred toward the end of the test. A similar depletion was observed for Mo, as confirmed by the leaching results in Figure 2. The highest contribution of the diffusion mechanism was recorded for Sb, especially for the D3CHI sample. It was noted that the addition of nanosilica (CHI and CHF) to the geopolymer composites did not significantly affect the leaching mechanisms.

The dominant driving forces for metal migration in aqueous environments stem from the continuous concentration gradient between the material matrix and the surrounding liquid, which leads to a complex leaching mechanism (Cubukcuoglu and Ouki 2019). Initially, heavy metal release is assumed to occur from soluble salts, while the increasing concentrations over extended leaching periods likely result from the dissolution of mineral phases within the matrix. Additionally, leaching mechanisms vary depending on the hydration state of the elements (Overmann et al. 2021).

Analyzing the leaching behavior of heavy metals is crucial, as these elements can gradually accumulate in various environmental components and living organisms once released (e.g., cadmium, lead, zinc, copper, nickel, chromium). Metals such as cadmium and lead are known to be toxic even at minimal concentrations (Zaynab et al. 2022). Notably, no leaching of these elements was observed over the 64-day period, with Cd and Pb concentrations remaining below the detection limit.

Assessment of the potential ecological risk of geopolymer composites

The potential ecological risk indicators E_r^i and PERI for geopolymer composites (Table 5) were calculated based on

equation (6). All tested samples exhibited a low ecological risk, with E_r^i and PERI values below 40. was estimated. The absence of environmental threat is further supported by the total PERI value, which remained below 150 for each geopolymer. The analysis clearly showed that, despite the presence of heavy metals in the tested materials, their concentrations are low. Therefore, monitoring these contaminants after the potential introduction of geopolymer composites into soils is unnecessary.

Comparing the obtained PERI indicator with studies on other geopolymers is challenging, as ecological risk assessments using such indicators are not commonly performed for these materials. However, when compared to soils in Wrocław, Poland (Holtra and Zamorska-Wojdyła 2023), where the average PERI was 2212 (for 66 soil samples), the results of this study confirms that geopolymer composites pose no environmental hazard during their application.

Conclusion

The studies revealed no ecological risk from the heavy metals present in the geopolymer composites. In most cases, the total heavy metal content was lower than the reference background values in soils. The potential ecological risk indicators for the analyzed heavy metals were <40 , indicating a low level of risk. The total ecological risk potential indicator (PERI) ranged from 21 to 35, depending on the geopolymer composition, further confirming the absence of potential harmful effects of heavy metals on the environment. The 64-day leaching study showed the release of Ba, Cr, Mo, and Sb from the geopolymer composites. No strict trend in leaching behavior was observed based on composition. However, the D3 composite (containing 50% fly ash, 50% glass powder, and 0.5% graphene) exhibited distinct leaching behavior, releasing the highest amount of barium. In contrast, for the other geopolymer composites – both those with 100% fly ash and those with a 15% glass powder

- molybdenum was leached in the highest concentration. The concentrations of Cd, Co, Cu, Ni, Pb, and Zn remained below the detection limit, indicating their complete immobilization within the geopolymer matrix. It can be concluded that the immobilization of these elements occurred within the geopolymer matrix. To better understand the leaching process, the mechanisms governing long-term metal release were also examined. The leaching of Ba, Cr, Mo, and Sb was primarily controlled by dissolution and diffusion, while depletion of available ions was observed for barium and molybdenum.

The heavy metals leaching study using the tank test is essential for assessing the environmental compatibility of geopolymer composites for specific applications. As an alternative to cement concrete, these materials may undergo leaching when introduced into soil or aquatic environments. The tank test serves as a practical method for assessing leaching, particularly in fresh construction materials.

The studies presented in this work, along with analyses of consistency, strength, and resistance to aggressive environmental factors, which are the subject of further research by the authors, will provide a comprehensive view of the potential applications of these composites in the construction industry, as well as their safety for the soil-water environment and human health. Further research in this area will help estimate the impact of incorporating nanosilica into geopolymer composites. As shown in this study, the addition of hydrophilic or hydrophobic nanosilica did not significantly affect the leaching behavior of metals. However, given its beneficial effects on mechanical strength, the use of nanosilica in geopolymer formulations remains a viable option.

Conflict of interest

The authors declare no known competing financial interests or personal relationships that could have influenced the work reported in this paper.

Table 5. Potential ecological risk indicator and total PERI for the tested geopolymer composites

Sample	Potential ecological risk indicator							PERI
	Cd	Co	Cr	Cu	Ni	Pb	Zn	
D1	21.60	0.58	2.48	3.88	3.90	1.52	0.53	34.49
D1CHI	19.20	0.60	2.56	3.54	3.74	1.56	0.53	31.73
D1CHF	17.40	0.54	2.00	3.36	3.46	1.57	0.47	28.79
D2	15.00	0.41	1.52	2.58	2.80	0.91	0.38	23.61
D2CHI	15.60	0.50	2.17	2.80	2.80	1.04	0.48	26.15
D2CHF	15.00	0.46	1.85	2.78	2.78	1.05	0.41	24.54
D3	15.00	0.30	1.07	1.94	1.96	0.87	0.28	21.42
D3CHI	15.00	0.50	2.56	3.00	3.00	1.04	0.48	25.96
D3CHF	15.00	0.30	1.45	1.64	1.64	0.46	0.28	22.87

References

- Abbasi, S. M., Ahmadi, H., Khalaj, G. & Ghasemi, B. (2016). Microstructure and mechanical properties of a metakaolinite-based geopolymer nanocomposite reinforced with carbon nanotubes, *Ceramics International*, 42, 14, pp. 15171–15176. DOI:10.1016/J.CERAMINT.2016.06.080
- Bakhshi, N., Sarrafi, A. & Ramezani-pour, A. A. (2019). Immobilization of hexavalent chromium in cement mortar: leaching properties and microstructures, *Environmental Science and Pollution Research*, 26, 20, pp. 20829–20838. DOI:10.1007/S11356-019-05301-Z/FIGURES/7
- Bao, S., Qin, L., Zhang, Y., Luo, Y. & Huang, X. (2021). A combined calcination method for activating mixed shale residue and red mud for preparation of geopolymer, *Construction and Building Materials*, 297, p. 123789. DOI:10.1016/J.CONBUILDMAT.2021.123789
- Baran, A. & Wieczorek, J. (2015). Application of geochemical and ecotoxicity indices for assessment of heavy metals content in soils, *Archives of Environmental Protection*, 41, 2, pp. 54–63. DOI:10.1515/AEP-2015-0019
- Cubukcuoglu, B. & Ouki, S. K. (2019). Comparison of mechanical and leaching behaviours of pulverised fuel ash/low-grade magnesium oxide-cement blended stabilised/solidified baghouse dust, *European Journal of Environmental and Civil Engineering*, 23, 6, pp. 771–788. DOI:10.1080/19648189.2017.1310140
- Faragó, T., Špirová, V., Blažeková, P., Lalinská-Voleková, B., Macek, J., Jurkovič, L., Vítková, M. & Hiller, E. (2023). Environmental and health impacts assessment of long-term naturally-weathered municipal solid waste incineration ashes deposited in soil-old burden in Bratislava city, Slovakia, *Heliyon*, 9, 3, p. e13605. DOI:10.1016/J.HELIYON.2023.E13605
- Hartwich, P. & Vollpracht, A. (2017). Influence of leachate composition on the leaching behaviour of concrete, *Cement and Concrete Research*, 100, pp. 423–434. DOI:10.1016/J.CEMCONRES.2017.07.002
- Holtra, A. & Zamorska-Wojdyla, D. (2023). Application of individual and integrated pollution indices of trace elements to evaluate the noise barrier impact on the soil environment in Wrocław (Poland), *Environmental Science and Pollution Research*, 30, 10, pp. 26858–26873. DOI:10.1007/S11356-022-23563-Y/FIGURES/7
- Janowska-Renkas, E., Zdrojek, M., Koziół, M. & Kaliciak-Kownacka, A. (2023). Effect of composition of geopolymer composites containing fly ash and waste glass powder on their durability and resistivity demonstrated in presence of a nanocarbon additive in a form of graphene, *Measurement*, 211, p. 112616. DOI:10.1016/J.MEASUREMENT.2023.112616
- Jaskuła, J., Sojka, M., Fiedler, M. & Wróżyński, R. (2021). Analysis of spatial variability of river bottom sediment pollution with heavy metals and assessment of potential ecological hazard for the Warta river, Poland, *Minerals*, 11, 3, pp. 1–21. DOI:10.3390/MIN11030327
- Ji, Z. & Pei, Y. (2020). Immobilization efficiency and mechanism of metal cations (Cd^{2+} , Pb^{2+} and Zn^{2+}) and anions (AsO_4^{3-} and $\text{Cr}_2\text{O}_7^{2-}$) in wastes-based geopolymer, *Journal of Hazardous Materials*, 384, p. DOI:10.1016/J.JHAZMAT.2019.121290
- Kazapoe, R. W., Amuah, E. E. Y. & Dankwa, P. (2022). Sources and pollution assessment of trace elements in soils of some selected mining areas of southwestern Ghana, *Environmental Technology & Innovation*, 26, p. 102329. DOI:10.1016/J.ETI.2022.102329
- Kowalik, R., Gawdzik, J. & Gawdzik, B. (2019). Risk Analysis of Accumulation of Heavy Metals from Sewage Sludge in Soil from the Sewage Treatment Plant in Starachowice, *Structure and Environment*, 11, 4, pp. 287–295. DOI:10.30540/SAE-2019-022
- Król, A. (2016). The role of the silica fly ash in sustainable waste management, *E3S Web of Conferences*, 10, 00049. DOI:10.1051/E3SCONF/20161000049
- Król, A. (2020). Mechanisms Accompanying Chromium Release from Concrete, *Materials*, 13, 8, p. 1891. DOI: 10.3390/MA13081891
- Król, A. & Mizerna, K. (2016). Directions of development of research methods in the assessment of leaching of heavy metals from mineral waste, *E3S Web of Conferences*, 10, p. 00050. DOI:10.1051/e3sconf/20161000050
- Król, M., Rožek, P., Chlebda, D. & Mozgawa, W. (2018). Influence of alkali metal cations/type of activator on the structure of alkali-activated fly ash – ATR-FTIR studies, *Spectrochimica Acta Part A: Molecular and Biomolecular Spectroscopy*, 198, pp. 33–37. DOI:10.1016/J.SAA.2018.02.067
- Kuterasińska-Warwas, J. & Król, A. (2017). Leaching of heavy metals from cementitious composites made of new ternary cements, *E3S Web of Conferences*, 19, p. 02019. DOI:10.1051/e3sconf/20171902019
- Liu, D., Wang, J., Yu, H., Gao, H. & Xu, W. (2021). Evaluating ecological risks and tracking potential factors influencing heavy metals in sediments in an urban river, *Environmental Sciences Europe*, 33, 1, pp. 1–13. DOI:10.1186/S12302-021-00487-X/FIGURES/6
- Movafagh, A., Mansouri, N., Moattar, F. & Vafaeinejad, A. R. (2020). Distribution and Ecological Risk Assessment of Heavy Metals in Roadside Soil along the Hemat Highway of Tehran, Iran, *Environment Protection Engineering*, 44, 3, pp. 5-17. DOI:10.37190/EPE180301
- Müllauer, W., Beddoe, R. E. & Heinz, D. (2015). Leaching behaviour of major and trace elements from concrete: Effect of fly ash and GGBS, *Cement and Concrete Composites*, 58, pp. 129–139. DOI:10.1016/J.CEMCONCOMP.2015.02.002
- Nayak, D. K., Abhilash, P. P., Singh, R., Kumar, R. & Kumar, V. (2022). Fly ash for sustainable construction: A review of fly ash concrete and its beneficial use case studies, *Cleaner Materials*, 6, p. 100143. DOI:10.1016/J.CLEMA.2022.100143
- Overmann, S., Lin, X. & Vollpracht, A. (2021). Investigations on the leaching behavior of fresh concrete – A review, *Construction and Building Materials*, 272, p. 121390. DOI:10.1016/J.CONBUILDMAT.2020.121390
- Phoo-ngernkham, T., Chindaprasirt, P., Sata, V., Hanjitsuwan, S. & Hatanaka, S. (2014). The effect of adding nano- SiO_2 and nano- Al_2O_3 on properties of high calcium fly ash geopolymer cured at ambient temperature, *Materials and Design*, 55, pp. 58–65. DOI:10.1016/j.matdes.2013.09.049
- Rachwał, M., Penkała, M., Rogula-Kozłowska, W., Wawer-Liszka, M., Łukaszek-Chmielewska, A. & Rakowska, J. (2024). Influence of road surface type on the magnetic susceptibility and elemental composition of road dust, *Archives of Environmental Protection*, 50, 4, pp. 135–146. DOI: 10.24425/AEP.2024.152903
- Rozpondek, K. & Rozpondek, R. (2019). The Use of Ecological Risk Indicators to Assess the Degree of Pollution of Bottom Sediments of the Dzierżno Duże Water Reservoir, *Engineering and Protection of Environment*, 22, 1, pp. 53–62. DOI:10.17512/ios.2019.1.5. (in Polish)

- Shamsol, A.'lia S., Apandi, N. M., Zailani, W. W. A., Izwan, K. N. K., Zakaria, M. & Zulkarnain, N. N. (2024). Graphene oxide as carbon-based materials: A review of geopolymer with addition of graphene oxide towards sustainable construction materials, *Construction and Building Materials*, 411, pp. 134410. DOI:10.1016/J.CONBUILDMAT.2023.134410
- Sitarz-Palczak, E., Kalembkiewicz, J. & Galas, D. (2019). Comparative study on the characteristics of coal fly ash and biomass ash geopolymers, *Archives of Environmental Protection*, 45, 1, pp. 126–135. DOI:10.24425/AEP.2019.126427
- Sobolev, K., Flores, I., Torres-Martinez, L. M., Valdez, P. L., Zarazua, E. & Cuellar, E. L. (2009). Engineering of SiO₂ Nanoparticles for Optimal Performance in Nano Cement-Based Materials, *Nanotechnology in Construction* 3, pp. 139–148. DOI:10.1007/978-3-642-00980-8_18
- Sun, Z & Vollpracht, A. (2020). Leaching of monolithic geopolymer mortars, *Cement and Concrete Research*, 136, p. 106161. DOI:10.1016/J.CEMCONRES.2020.106161
- Takahashi, S., Daimon, M. & Sakai, E. (2003). Sorption of CrO₄²⁻ for cement hydrates and the leaching from cement hydrates after sorption, Proceedings of the 11th International Congress on the Chemistry of Cement "Cement's Contribution on the Development in the 21st Century," pp. 2166–2172.
- Tan, J., De Vlieger, J., Desomer, P., Cai, J. & Li, J. (2022). Co-disposal of construction and demolition waste (CDW) and municipal solid waste incineration fly ash (MSWI FA) through geopolymer technology, *Journal of Cleaner Production*, 362, p. 132502. DOI:10.1016/J.JCLEPRO.2022.132502
- Tho-In, T., Sata, V., Boonserm, K. & Chindaprasirt, P. (2018). Compressive strength and microstructure analysis of geopolymer paste using waste glass powder and fly ash, *Journal of Cleaner Production*, 172, pp. 2892–2898. DOI:10.1016/J.JCLEPRO.2017.11.125
- Tomczyk, P., Wdowczyk, A., Wiatkowska, B. & Szymańska-Pulikowska, A. (2023). Assessment of heavy metal contamination of agricultural soils in Poland using contamination indicators, *Ecological Indicators*, 156, p. 111161. DOI:10.1016/J.ECOLIND.2023.111161
- Toniolo, N., Rincón, A., Roether, J. A., Ercole, P., Bernardo, E. & Boccaccini, A. R. (2018). Extensive reuse of soda-lime waste glass in fly ash-based geopolymers, *Construction and Building Materials*, 188, pp. 1077–1084. DOI:10.1016/J.CONBUILDMAT.2018.08.096
- Van Der Sloot, H. A. (2000). Comparison of the characteristic leaching behavior of cements using standard (EN 196-1) cement mortar and an assessment of their long-term environmental behavior in construction products during service life and recycling, *Cement and Concrete Research*, 30, 7, pp. 1079–1096. DOI:10.1016/S0008-8846(00)00287-8
- Vollpracht, A. & Brameshuber, W. (2016). Binding and leaching of trace elements in Portland cement pastes, *Cement and Concrete Research*, 79, pp. 76–92. DOI:10.1016/J.CEMCONRES.2015.08.002
- Yandem, G. & Jabłońska-Czapla, M. (2024). Review of indium, gallium, and germanium as emerging contaminants: occurrence, speciation and evaluation of the potential environmental impact, *Archives of Environmental Protection*, 50, 3, pp. 84–99. DOI:10.24425/AEP.2024.151688
- Zaynab, M., Al-Yahyai, R., Ameen, A., Sharif, Y., Ali, L., Fatima, M., Khan, K. A. & Li, S. (2022). Health and environmental effects of heavy metals, *Journal of King Saud University - Science*, 34, 1, p. 101653. DOI:10.1016/J.JKSUS.2021.101653

Wpływ na środowisko kompozytów geopolimerowych na bazie odpadów: wymywalność metali ciężkich i ocena ryzyka

Streszczenie. Geopolimery to stosunkowo nowy rodzaj materiałów, które można wytwarzać z odpadów. Materiały te mogą zawierać niebezpieczne związki, takie jak metale ciężkie, które stwarzają potencjalne zagrożenie dla środowiska w przypadku ich uwolnienia. W pracy przedstawiono wyniki uwalniania metali ciężkich z formowanych kompozytów geopolimerowych w funkcji czasu oraz w celu oceny mechanizmów wymywania poszczególnych pierwiastków. W badaniu oceniono również potencjalne ryzyko ekologiczne badanych materiałów. Kompozyty geopolimerowe wytworzono na bazie popiołu lotnego krzemionkowego (CFA) i odpadowej mączki szklanej (GP). Ponadto skład modyfikowano grafenem i nanokrzemionką. Badaniom poddano materiały o innowacyjnych składach oraz które mogłyby z powodzeniem zastąpić tradycyjny beton na bazie cementu portlandzkiego, którego produkcja jest obecnie uznawana za zwiększającą ślad węglowy. Wymywalność przeprowadzono metodą "tank test". Kompozyty geopolimerowe wykazały uwalnianie Ba, Cr, Mo i Sb spośród dziesięciu metali analizowanych w pracy. Ustalono, że proces wymywania był kontrolowany przede wszystkim przez rozpuszczanie i dyfuzję, jednak w przypadku Ba i Mo zaobserwowano również proces wyczerpywania się jonów dostępnych do wymywania. Analiza ryzyka ekologicznego nie wykazała zagrożenia dla środowiska ze strony badanych kompozytów geopolimerowych.