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Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv



Stabilization and solidification of lead, chromium, and cadmium-contaminated clayey sand using calcium hydroxide and sodium carbonate-activated materials

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HIGHLIGHTS

- Ca(OH)₂ and Na₂CO₃-activated slag and zeolite dramatically improves soil strength
- UCS reaches 4449 kPa, showing a 75fold increase compared to the nonstabilized soil
- Leachability of lead, chromium, and cadmium reduced by 95, 92, and 98 %, respectively
- \bullet LCA results show 75 % lower $\rm CO_2$ emissions compared to OPC-based stabilization methods
- C(A)SH and Ettringite gels significantly enhance soil strength and immobilize *HMs*

ARTICLE INFO

Editor: Antonella Petrillo

Keywords: Stabilization and solidification Sodium carbonate activation Alkali-activated slag Heavy metals immobilization Life cycle assessment Leaching

G R A P H I C A L A B S T R A C T



ABSTRACT

Soil pollution by heavy metals has emerged as a critical environmental issue, posing significant threats to human health, food security, and environmental sustainability. Particularly in industrial and mining areas, the high levels of pollutants such as lead, chromium, and cadmium degrade soil quality and create potential risks for ecosystems. This study explores the stabilization and solidification of lead, chromium, and cadmium-contaminated clayey sand using Ca(OH)₂ and Na₂CO₃-activated waste materials, specifically slag and natural zeolite, aimed at improving mechanical strength, reducing leachability, and enhancing environmental sustainability. Unconfined compressive strength tests revealed that soil samples treated with 20 wt% Ca(OH)₂ and Na₂CO₃-activated slag exhibited a *UCS* of 4449 kPa after 28 days, a 75-fold increase over untreated soil, far exceeding the *U.S. EPA*'s minimum requirement for landfill stabilization. X-ray diffraction (*XRD*) and field emission scanning electron microscopy (*FE-SEM*) analyses confirmed the formation of calcium-aluminosilicate hydrate (C(A)SH) gels and secondary products like ettringite, which contributed to the substantial strength gain. Toxicity characteristic leaching procedure (*TCLP*) leachability tests, showed that Ca(OH)₂ and Na₂CO₃-activated slag achieved up to 95 %, 92 %, and 98 % reductions in lead, chromium, and cadmium leachability, respectively, all below *EPA* regulatory thresholds. Permeability tests demonstrated a significant decrease in

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https://doi.org/10.1016/j.scitotenv.2025.179612

Received 8 January 2025; Received in revised form 27 March 2025; Accepted 4 May 2025 Available online 8 May 2025

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hydraulic conductivity, with up to a 97 % reduction, indicating the treated soil's improved resistance to contaminant migration. Life cycle assessment (*LCA*) showed that the proposed Ca(OH)₂ and Na₂CO₃-activated slag and zeolite-based system reduced CO_2 emissions by 75 % compared to conventional Portland cement stabilization, while maintaining equivalent performance in *UCS* and metal immobilization.

Abbreviations

AAMs Alkali-activated materials AAS Atomic absorption spectroscopy ASTM American society for testing and materials C(A)SH Calcium aluminate silicate hydrate Cd Cadmium CN-AS Calcium hydroxide and sodium carbonate-activated slag Cr Chromium Dr Relative density EC Electrical conductivity EDS Energy-dispersive x-ray spectroscopy FE-SEM Field emission scanning electron microscopy GGBFS Ground granulated blast-furnace slag HMconc. Heavy metal concentration HMs Heavy metals ICP-OES Inductively coupled plasma optical emission spectroscopy k Permeability coefficient LCA Life cycle assessment LCI Life cycle inventory LCIA Life cycle impact assessment OPC Ordinary Portland cement Pb Lead pH Potential of hydrogen S/SStabilization and solidification Sc Slag content TCLP Toxicity characteristic leaching procedure UCS Unconfined compressive strength USCS United States soil classification system WHO World health organization

XRD X-ray diffraction

XRF X-ray fluorescence

1. Introduction

Soil pollution has emerged as a critical environmental issue, primarily caused by improper waste disposal, industrial emissions, excessive pesticide and fertilizer use, as well as mining and fossil fuel combustion activities (Yuanan et al., 2020). These practices have led to widespread soil contamination, presenting significant risks to human health, food security, and environmental sustainability (Hoang et al., 2024). According to the National Survey of Soil Pollution, heavy metals (*HMs*) are among the most pervasive contaminants compromising soil quality.

The issue of *HMs* contamination is not confined to one region but is a global concern, with countries worldwide facing severe consequences. Of particular concern is the accumulation of toxic heavy metals in agricultural soils, as this can lead to environmental degradation and potential contamination of the food chain (Yuksel and Arica, 2018). According to estimates from the World Health Organization (*WHO*) and the Environmental Protection Agency (*EPA*), between 10 % and 20 % of the world's soils may be contaminated with *HMs*, especially in areas impacted by industrial activities and mining operations. Among the most hazardous heavy metals are lead (*Pb*), cadmium (*Cd*), and chromium (*Cr*), which have been identified as widespread pollutants across different regions (Yang et al., 2018).

Highlighting the severity of this problem, the China National Soil Pollution Survey Bulletin (2014) revealed that 16.1 % of the country's soils were contaminated, with certain regions exceeding safety thresholds for specific metals: 7.0 % for *Cd*, 1.5 % for *Pb*, and 1.1 % for *Cr* (Yang et al., 2018). As classified by the *WHO*, *Pb*, *Cd*, and *Cr* are considered major public health threats due to their toxicity and persistence in the environment (Ren et al., 2023).

Contaminated industrial soils often contain *Pb* levels ranging from 500 to 10,000 mg.kg⁻¹ (Alloway, 2013; Wuana and Okieimen, 2011), with specific cases exceeding 12,000 mg.kg⁻¹ (Lamb et al., 2009). *Cr* concentrations have been reported above 2500 mg.kg⁻¹ in heavily polluted sites (Dhal et al., 2013; Kabata-Pendias and Mukherjee, 2007), while *Cd* levels can reach up to 900 mg.kg⁻¹ (Alloway, 2013; Phaenark et al., 2009), demonstrating the severity of industrial contamination.

Given the extreme toxicity of these metals, it is crucial to implement remediation measures once their concentrations surpass safe limits. Various techniques have been developed to address *HMs* contamination, including leaching, phytoremediation, microbial treatment, and stabilization and solidification (S/S) (Luo et al., 2023). Among these, *S/S* is method has gained recognition as an effective, cost-efficient, and practical solution to tackle soil contamination (Zou et al., 2023).

The *S/S* technology is a well-established commercial method that immobilizes contaminants by encapsulating them in a stable matrix, thereby reducing the mobility and volatility of *HMs* and preventing further environmental damage (Komaei et al., 2023a; Ren et al., 2023). The method has distinct advantages over other remediation technologies, including lower costs and faster implementation, with *S/S* typically costing less than \$100 per ton (Ren et al., 2023). Additionally, soils treated through *S/S* can be repurposed for construction, promoting the rehabilitation of contaminated sites (Wei et al., 2015).

In the S/S processes, binders are used to convert polluted soils into a stable solid form. Ordinary Portland cement (OPC) is the most commonly employed stabilizer due to its affordability, availability, and ease of application (Long et al., 2023; Tang, 2022). However, the production of OPC is associated with significant environmental drawbacks, particularly due to its large carbon footprint. Approximately 900 kg of CO2 is emitted for every 1000 kg of OPC produced, contributing to 5-8 % of global carbon emissions (Komaei and Saeedi, 2025; Razeghi et al., 2024b). Global OPC production is currently estimated at 4.6 billion tons annually, with forecasts predicting 6 billion tons by 2050 (Amran et al., 2022). The environmental impact of OPC in S/S is further compounded by the fact that HMs can inhibit the cement hydration reactions, potentially leading to structural degradation and limiting the treated soil's strength development (Weng et al., 2024). Therefore, it is crucial to explore alternative binders that offer both environmental sustainability and long-term stabilization of HMs.

Zeolites represent a promising alternative for the stabilization of heavy metals. These natural minerals function by exchanging cations within their structure for heavy metal ions, thereby effectively immobilizing multiple metals simultaneously (Budianta et al., 2020). However, the efficacy of zeolites can be compromised by impurities like Quartz, which reduce their specific surface area and hinder heavy metal immobilization (Ma et al., 2022). While zeolites alone may not provide sufficient long-term stability, their potential as supplementary materials in S/S processes are still notable.

Recent research has focused on alkali-activated materials (*AAMs*) as alternative binders for *S/S*, which are derived from industrial, agricultural, and mining wastes (Komaei et al., 2023d; Salas et al., 2024). *AAMs* are produced by activating aluminosilicate precursors with alkaline solutions (Razeghi et al., 2024a; Samadi et al., 2023). Research has shown that *AAMs* offer superior mechanical properties, enhanced durability, and a significantly lower carbon footprint compared to

Portland cement (Komaei et al., 2023c; Li et al., 2023). Furthermore, *AAMs* have shown promise in immobilizing *HMs* in treated soils (Komaei et al., 2023b; Molaei et al., 2024; Weng et al., 2024).

Despite their advantages, traditional alkaline activators like sodium hydroxide (NaOH) and sodium silicate (Na₂SiO₃) come with drawbacks, including high corrosivity, shrinkage, and soil pH alterations, which limit their practical use in large-scale remediation projects (Akturk et al., 2019a; Komaei et al., 2024). These activators are also energy-intensive to produce, adding to the overall carbon footprint of AAMs (Komaei et al., 2024; Sengupta et al., 2024). In response to these issues, sodium carbonate (Na₂CO₃) has been proposed as an alternative due to its lower environmental impact, reduced corrosivity, and cost-effectiveness. Na₂CO₃ is abundant, with global reserves estimated at 24 billion tons, and is approximately one-third the cost of NaOH or Na₂SiO₃ (Adesina, 2021; Akturk et al., 2019a). Despite the benefits, Na₂CO₃ as an activator faces some limitations, such as longer setting times and slower strength development (Komaei et al., 2024). To mitigate these challenges, researchers have tested various additives and secondary activators, such as calcium hydroxide, silica fume, and sodium hydroxide, to accelerate the reaction process and improve the performance of Na₂CO₃-activated materials (Adesina, 2020, 2021). Calcium hydroxide (Ca(OH)₂), in particular, has proven effective in enhancing early-age strength while offering environmental advantages over other additives (Adesina, 2020).

While numerous studies have investigated the S/S of contaminated soils with *AAMs*, this research introduces a groundbreaking approach by employing a novel combination of Ca(OH)₂ and Na₂CO₃-activated slag (*CN-AS*) for *S/S*. Unlike conventional techniques that typically utilize NaOH, Na₂SiO₃, and potassium hydroxide (KOH) as activators, this study strategically incorporates *CN-AS* and natural zeolite, which not only enhances the stabilization efficiency but also significantly minimizes both the carbon footprint and environmental impacts associated with these processes. The integration of sodium carbonate and calcium hydroxide offers a dual advantage—reducing *CO*₂ emissions during activation while improving the mechanical integrity and immobilization capacity of the stabilized matrix.

Moreover, this research conducts a detailed life cycle assessment (*LCA*) to determine the sustainability advantages of this approach, demonstrating its superiority over the traditional methods in terms of environmental impact reduction. This innovative methodology presents a transformative alternative to conventional S/S practices, paving the way for more sustainable soil remediation technologies.

The effectiveness of this approach was assessed through a series of tests, including unconfined compressive strength (*UCS*), toxicity characteristic leaching procedure (*TCLP*), and permeability tests. Additionally, microstructural analyses were conducted to explore the underlying mechanisms of metal stabilization.

2. Material and methods

2.1. Material

2.1.1. Soil

In this study, natural soil was collected from a site in the southern region of Tehran, Iran., at a depth of 1 m. The physical characteristics of the soil sample are detailed in Table 1. According to the United States Soil Classification System (*USCS*), the soil is categorized as clayey sand (*SC*). The soil's particle size distribution curve is provided in Fig. 1. The chemical composition was analyzed through X-ray fluorescence (*XRF*), and the results are presented in Table 2.

2.1.2. Alkali-activated material (AAMs)

The alkali-activated materials (*AAMs*) were prepared using two components: an aluminosilicate source and an activator. The aluminosilicate source consisted of ground granulated blast-furnace slag (*GGBFS*), obtained from Isfahan Steel Company, and natural zeolite,

Table 1Specifications of the natural soil.

Characteristics	Value
Specific gravity, G_s	2.67
Liquid Limit, LL (%)	24.63
Plastic Index, PI (%)	8.09
Maximum void ratio, e_{max}	0.99
Minimum void ratio, e_{min}	0.78
D ₁₀ (mm)	0.02
D ₅₀ (mm)	0.42
D ₉₀ (mm)	2.55
USCS	SC



Fig. 1. Particle size distribution of natural soil, slag, and zeolite.

sourced from Semnan Negin Powder Company. The chemical compositions of the slag and zeolite used in this study were identified through *XRF* analysis and are given in Table 2. Moreover, the particle size distribution curves for both slag and zeolite are illustrated in Fig. 1.

To activate the aluminosilicate sources (the combination of slag and zeolite), a mixture of Na_2CO_3 powder and $Ca(OH)_2$ powder was employed. The sodium carbonate powder had a purity of 99 % and a density of 2.53 g.cm⁻³, while the calcium hydroxide powder also had a purity of 99 % and a density of 2.21 g.cm⁻³.

2.2. Methods

2.2.1. Sample preparation

The natural soil was sieved through a #4 mesh and dried at 105 °C for one day. To artificially contaminate the soil, solutions of Pb(NO₃)₂, K₂CrO₄, and CdSO₄ were prepared, targeting concentrations of 5000 mg. kg⁻¹ for *Pb*, 2500 mg.kg⁻¹ for *Cr*, and 1000 mg.kg⁻¹ for *Cd*, representing 25 % of the total soil weight. The prepared solutions were then mixed with the dry natural soil, which was stored in sealed bags for 7 days. Afterwards, the contaminated soil was dried at 105 °C for one day. To verify the contamination levels, the concentrations of various metals in contaminated soil samples were measured using inductively coupled plasma-optical emission spectroscopy (*ICP-OES*) and atomic absorption spectroscopy (*AAS*). The results are provided in Table 3.

The stabilization of samples in this study was conducted using a onepart method. The soil, aluminosilicate precursors (the combination of slag and zeolite), and activators (the combination of Na_2CO_3 powder and $Ca(OH)_2$ powder) were mixed using a HOBART N50 mechanical mixer with a flat beater attachment at a speed of 125 rpm (medium setting) for 5 min to achieve a homogeneous mixture. After obtaining a

Table 2

Chemical composition of natural soil, slag and zeolite.

Composition	SiO_2	CaO	Al_2O_3	MgO	TiO ₂	MnO	SO_3	Fe ₂ O ₃	K ₂ O	Na ₂ O
Soil [wt%] Slag [wt%] Zeolite [wt%]	58.00 35.92 77.63	11.60 38.45 5.44	12.72 10.16 12.54	3.48 8.91 0.89	0.67 1.51 0.24	0.20 1.58 0.01	0.30 1.28 0.08	5.47 0.87 1.09	5.06 0.75 1.01	1.29 0.57 1.07
Element		Ni	Cr	Cu	Zn	Sr	Zr	Cd	Ba	Pb
Contaminated soil [wt%]		0.009	0.246	0.005	0.013	0.045	0.015	0.093	0.064	0.504

Table 3.Metals identified in prepared contaminated soil.

uniform mixture, water was added to the blend.

For this purpose, the amount of contaminated soil was specified based on a relative density (D_r) of 90 %. The weighed contaminated soil samples were then mixed with 5 %, 10 %, 15 %, and 20 % by weight (*wt* %) of slag. Natural zeolite was added to each contaminated soil and slag mixture at a rate of 10 *wt*% of the slag. Additionally, Na₂CO₃ and Ca (OH)₂ were added to activate the aluminosilicate precursors at a rate of 5 *wt*% of the slag. Finally, after thoroughly mixing the contaminated soil, aluminosilicate precursors, and activators to achieve a homogeneous dry mixture, 20 *wt*% of water was added. The soil samples stabilized with 5 %, 10 %, 15 %, and 20 % by weight of slag are referred to as S_5 , S_{10} , S_{15} , and S_{20} , respectively.

To evaluate the effectiveness of the investigated method, a batch of samples was prepared using 5 %, 10 %, 15 %, and 20 % by weight of ordinary Portland cement (*OPC*), stabilized with 20 % by weight of water. These samples were designated as C_5 , C_{10} , C_{15} , and C_{20} , respectively. Additionally, a non-stabilized contaminated soil sample was compacted with 20 wt% of water and referred to as the C_S sample.

The mixture was placed in cylindrical molds (38 mm diameter, 76 mm height) in three layers. Each layer was compacted using a plastic hammer in accordance with ASTM D2166, which is the standard for preparing soil samples for unconfined compressive strength tests. After one hour, the samples were removed and stored in sealed bags at ambient temperature (20 ± 4 °C) and 70 ± 5 % relative humidity for 3, 7, and 28 days to maintain moisture. A summary of the sample preparation is shown in Table 4.

2.2.2. Unconfined compressive strength (UCS)

The *UCS* of both non-stabilized and stabilized samples was measured in accordance with ASTM D2166. In this test, the prepared samples were subjected to loading with a constant strain rate of 1 mm.min⁻¹. The average *UCS* values for both non-stabilized and stabilized samples were reported after conducting three repetitions for each sample to ensure the reproducibility of the results.

Table 4

Summary of specimen preparation.

Characteristics	Value
Soil type	Clayey sand
Relative density of sample preparation, D_r [%]	90
Heavy metal concentration [mg/kg of soil]	
Lead (Pb (II))	5000
Chromium (Cr (VI))	2500
Cadmium (Cd (II))	1000
Binder type	Alkali-activated materials
Aluminosilicate sources	Slag + Natural zeolite
Slag content [wt% of soil]	0, 5, 10, 15, and 20
Zeolite content [wt% of slag]	10
Activator type	$Na_2CO_3 + Ca(OH)_2$
Na ₂ CO ₃ content [wt% of slag]	5
Ca(OH) ₂ content [wt% of slag]	5
Water [wt% of soil]	20
Testing age [day]	3, 7, and 28
Curing condition	Ambient
Temperature [°C]	20 ± 4
Relative humidity [%]	70 ± 5

2.2.3. Leachability test

The toxicity characteristic leaching procedure (*TCLP*) test, based on U.S. Environmental Protection Agency (*USEPA*) method 1311 (Agency, 1992), was utilized to assess the leaching of lead, chromium, and cadmium from the samples. As per this test, the maximum diameter of the solid sample (both non-stabilized and stabilized contaminated soil) should not exceed 9.5 mm, with a weight ratio of the extraction solution to the solid sample of 20:1. The extraction solution was prepared using acetic acid at a *pH* of 2.88 \pm 0.05. Following the test protocol, the bottle containing the extraction solution and solid sample was rotated for 18 h at a rotation speed of 30 rpm using the *TCLP* test device. Subsequently, the solution was filtered using Whatman #42 filter paper to separate the solids from the leaching solution. The final liquid (leachate) obtained was analyzed for heavy metal concentrations (*Pb*, *Cd*, and *Cr*) using *ICP-OES* analysis. This experiment was conducted in triplicate to confirm the results.

2.2.4. Permeability test

The permeability of both non-stabilized and stabilized soil was evaluated with the constant head method as per ASTM D2434–68. In this experiment, soil samples were saturated, and water flow through the samples under a constant differential head was measured (Cheng et al., 2024). The permeability coefficient (k) was calculated using Eq. (1):

$$k = \frac{QL}{Aht}$$
(1)

where Q is the water volume, L is the sample length, A is the crosssectional area, h is the head difference, and t is the time taken for water to pass through.

2.2.5. EC and pH

To conduct the pH and electrical conductivity (EC) measurements, a 50 g soil sample was initially ground and passed through a No. 10 sieve, ensuring that the maximum particle diameter was no greater than 2 mm. Then the sieved soil was mixed with 50 ml of distilled water, ensuring thorough dispersion, and the mixture was stirred for 15 min to achieve uniform soil-water suspension. Following this, pH and EC values were measured. The procedures outlined in this study accord with the guidelines specified in ASTM D4972 for pH and EC determination.

2.2.6. Microstructural characteristics

Minerals in the contaminated soil and products formed by aluminosilicate precursors activated by Ca(OH)₂ and Na₂CO₃ were identified using X-ray diffraction (*XRD*). Energy dispersive X-ray spectroscopy (*EDS*) was conducted to assess and identify the chemical elements in contaminated soil samples. Additionally, field emission scanning electron microscopy (*FE-SEM*) was used to analyze the morphology of the non-stabilized contaminated soil particles and the amorphous structures formed in the stabilized contaminated soil. Furthermore, optical microscopy with $220 \times$ magnification was performed to allow a closer inspection of the void spaces between particles in the non-stabilized soil and the bonds formed between particles and cementation in the stabilized soil, contributing to a deeper understanding of the changes in soil structure due to the stabilization process.

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2.2.7. Statistical analysis

Data analysis was performed using ORIGIN PRO 2024 software. Regression analysis was conducted to establish correlations between the percentage of *CN-AS* and parameters such as *UCS*, leachability, permeability, *pH*, and *EC*. The significance of these correlations was evaluated using an F-test at a significance level of 0.05.

3. Results

3.1. Unconfined compressive strength (UCS)

Fig. 2 presents the *UCS* for both non-stabilized and stabilized samples treated with varying Ca(OH)₂ and Na₂CO₃-activated slag (*CN-AS*) contents (*S_c*). A power function, represented as $UCS = a \times (S_c + b)^c$, was employed to model the relationship between *UCS* and *S_c*. Statistical analysis confirmed that the fitted models yielded statistically significant *p*-values and *R*² values, suggesting a strong correlation. The stabilization approach employed in this study proved to be highly effective. After the 28th day of curing, all stabilized soil samples containing 5–20 wt% *CN-AS* surpassed the *U.S. EPA*'s recommended *UCS* threshold of 350 kPa for landfill soil stabilization (depicted as a dashed line in Fig. 2) (Phojan et al., 2023; Yang et al., 2024). Notably, the *UCS* of the sample stabilized with 20 wt% *CN-AS* (*S*₂₀) increased more than 75-fold compared to the non-stabilized sample, demonstrating the effectiveness of the stabilization method.

Furthermore, *UCS* continued to increase with higher slag content. After the 28th day of curing, *UCS* values of 771, 1596, 2738, and 4449 kPa recorded for samples stabilized with 5 %, 10 %, 15 %, and 20 % *CN*-*AS* (S_5 , S_{10} , S_{15} , and S_{20}), respectively. The *UCS* values also improved progressively with curing time, particularly for the S_{20} sample, where *UCS* values after 3, 7, and 28 days of curing were 972, 1846, and 4449 kPa, respectively.

The *UCS* enhancement achieved in this study significantly outperforms several alternative stabilization techniques applied to heavy metal-contaminated soils. For instance, microbial induced calcium precipitation (*MICP*) has been shown to improve *UCS* in the range of 539–1130 kPa, depending on bacterial activity and curing conditions (Hadi and Saeed, 2022; Xing et al., 2023; Yin et al., 2022). Similarly, biopolymer-based methods, such as xanthan gum, guar gum, gellan gum, and chitosan, have reported *UCS* improvements ranging from 337 kPa to 2900 kPa (Ayeldeen et al., 2016; Chang et al., 2016; Cha



Fig. 2. UCS of non-stabilized and stabilized samples with varying CN-AS contents.

2015; Hataf et al., 2018). *OPC*-based stabilization has shown *UCS* improvements from 1500 to 6900 kPa, depending on curing conditions (Chen et al., 2023; Yin et al., 2022). In comparison, the *CN-AS* method in this study achieved a peak *UCS* of 4449 kPa. Unlike alternatives that require precise control of biological activity or polymer concentrations, *CN-AS* offers a chemically stable, scalable, and cost-effective solution. It also avoids the environmental drawbacks of Portland cement, providing sustained strength development and superior performance for stabilizing heavy metal-contaminated soils.

3.2. Leachability

Fig. 3 demonstrations the results from the *TCLP* test and *ICP-OES* analysis, which evaluated the leaching behavior of lead (*Pb*), chromium (*Cr*), and cadmium (*Cd*) from both untreated and stabilized contaminated soil samples with different *CN-AS* contents (S_c) after 3, 7, and 28 days of curing. The leachability data were modeled using an exponential function, with statistical analysis confirming a robust correlation. The *EPA's* regulatory limits for leaching of *Pb*, *Cr*, and *Cd* from contaminated waste in landfills are 5, 5, and 1 ppm, respectively, and this limits are depicted as reference lines in the Fig. 3 (EPA EPA, 1991).

The *TCLP* test and *ICP-OES* results demonstrate that after 28 days, the leached concentrations of lead ($Pb_{conc.}$), chromium ($Cr_{conc.}$), and cadmium ($Cd_{conc.}$) from contaminated soil samples stabilized with 10 %, 15 %, and 20 % by weight of *CN-AS* were below the *EPA* regulatory allowable limits, indicating the significant efficiency of the stabilization and solidification method for contaminated clayey sand soil. Specifically, the $Pb_{conc.}$, $Cr_{conc.}$, and $Cd_{conc.}$ from the S_{20} sample after 28 days of curing were reduced by 95 %, 92 %, and 98 %, respectively, compared to the non-stabilized contaminated soil sample.

Additionally, as shown in Fig. 3, increasing the *CN-AS* content to 5 wt % (S_5) significantly reduced the $Pb_{conc.}$, $Cr_{conc.}$, and $Cd_{conc.}$, with further increases in S_c leading to additional reductions. However, there was no significant variance in leached concentrations between S_{10} , S_{15} , and S_{20} . For example, after 28 days of curing, $Pb_{conc.}$ values were 8.67, 3.55, 2.99, and 2.79 ppm for samples S_5 , S_{10} , S_{15} , and S_{20} , respectively, representing reductions of 83 %, 93 %, 94 %, and 95 % compared to the non-stabilized sample. Moreover, the leached concentrations of *HMs* decreased with increased curing age. For example, in the case of Cr, the S_{20} sample showed leached concentrations of 6.28, 4.21, and 2.57 ppm after 3, 7, and 28 days of curing, respectively, resulting in reductions of 81 %, 87 %, and 92 % compared to the non-stabilized sample.

The saturation effect observed in Fig. 3, where the increase in *CN-AS* beyond 15 % results in minimal additional reductions in heavy metal leaching, is attributed to the saturation of available binding places within the soil-stabilizer system. At lower *CN-AS* dosages (5 %, 10 %, and 15 %), heavy metals are effectively immobilized through precipitation, adsorption, and encapsulation. However, beyond 15 % *CN-AS*, the binding capacity of the system reaches its limit, and the excess stabilizer does not significantly contribute to further leachability reduction. This behavior is consistent with similar findings, such as those of Estabragh et al. (2018), who observed comparable saturation effects in cement-based stabilization systems.

This immobilization performance surpasses several alternative stabilization techniques. For instance, MICP has been reported to achieve Pb and Cr(VI) immobilization efficiencies of up to 92 % and 94 %, respectively, in contaminated biocemented sand (Sharma et al., 2022). Similarly, Portland cement-based stabilization can effectively immobilize heavy metals in contaminated sands, with up to 97 % retention (Al-Kindi, 2019), while biochar-supported nanoscale iron sulfide composites have demonstrated a 94.7 % reduction in Cr(VI) leachability in TCLP tests (Lyu et al., 2018). Additionally, limestone calcined clay cement has been shown to immobilize up to 88 % of Pb in contaminated soils (Reddy et al., 2020), while chitosan-stabilized FeS composites have achieved Cr (VI) immobilization of up to 96.6 % (Shang et al., 2023). Other biopolymerand nanomaterial-based approaches, such as



Fig. 3. Leached concentrations of lead, chromium and cadmium from non-stabilized and stabilized contaminated soil samples after 3, 7, and 28 days of curing: (a) *Pb*, (b) *Cr*, and (c) *Cd*.

hydroxyapatite derived from flue gas desulfurization gypsum, have immobilized up to 93.8 % of *Pb* and 65.4 % of Cd (Yan et al., 2020), whereas sulfidated nanoscale zero-valent iron has demonstrated over 97.6 % immobilization of exchangeable cadmium (Guo et al., 2021).

Compared to these methods, the *CN-AS* stabilization system not only achieves comparable or superior immobilization efficiencies but also provides additional advantages such as chemical stability, scalability, and cost-effectiveness. The significant reduction in *Pb*, *Cr*, and *Cd* leachability highlights its potential as an effective solution for stabilizing heavy metal-contaminated soils, offering both environmental and engineering benefits over conventional stabilization approaches.

3.3. Permeability

The permeability coefficients (k) of non-stabilized contaminated soil samples and stabilized samples are shown in Fig. 4.

A power function of the form $k = a \times (S_c + b)^c$ was fitted to model the relationship between S_c and k after 3, 7, and 28 days of curing. Statistical analysis confirmed a high degree of accuracy, with significant R^2 values and p-values, as presented in Fig. 4.

According to the results shown in Fig. 4, the permeability of the non-stabilized soil was $1.61 \times 10^{-2} \text{ cm.s}^{-1}$. After stabilization and 28 days of

curing, the *k* for the soil stabilized with 5 %, 10 %, 15 %, and 20 % *CN*-*AS* were 2.05×10^{-3} , 9.29×10^{-4} , 5.92×10^{-4} , and 4.89×10^{-4} cm. s⁻¹, respectively, representing reductions of 87 %, 94 %, 96 %, and 97 % compared to the non-stabilized sample. As expected, permeability decreased with increasing *CN-AS* content; however, the rate of permeability reduction diminished with further increases in slag content.

When compared to previous studies, this permeability reduction demonstrates a significant improvement. For instance, a study on alkaliactivated slag stabilization using 5 % NaOH-activated slag reported a 57 % reduction in permeability, with stabilized soil exhibiting an average hydraulic conductivity of 6.54×10^{-3} cm.s⁻¹, compared to 1.52×10^{-2} cm.s⁻¹cm/s for untreated soil (Komaei et al., 2025). In contrast, *CN-AS* stabilization in the present study achieved up to 87 % permeability reduction with the same 5 wt% binder, indicating superior compaction and enhanced potential for contaminant containment. This substantial reduction highlights the effectiveness of *CN-AS* in reducing soil permeability, making it a promising candidate for geoenvironmental applications requiring low hydraulic conductivity, such as landfill liners and contaminant barriers.



Fig. 4. Permeability coefficients of non-stabilized and stabilized samples with varying *CN-AS* contents.

3.4. EC and pH

The results of electrical conductivity (*EC*) and *pH* measurements for both non-stabilized and stabilized contaminated soil with varying *CN-AS* contents (S_c) after 28 days are provided in Fig. 5.

A power function was fitted to describe the relationship among S_c and *EC*, while a linear function was used to model the relationship between S_c and *pH*. Statistical analysis indicated that both models had acceptable accuracy and significance.

The results show that EC decreased after stabilization and with increasing *CN-AS* content. Specifically, the *EC* values for the non-stabilized sample and the samples stabilized with 5 %, 10 %, 15 %, and 20 % by weight of *CN-AS* were 4.37, 3.16, 2.47, 2.13, and 1.91 mS. cm⁻¹, respectively. Thus, the *EC* of the soil stabilized with 20 wt% *CN-AS* decreased by more than 55 %.

Conversely, the *pH* increased after stabilization and with increasing *CN-AS* content. The *pH* of the non-stabilized contaminated soil was 7.91, which increased by 7.6 %, 16.2 %, 28.8 %, and 33.9 % for the samples stabilized with 5 %, 10 %, 15 %, and 20 wt% *CN-AS*, respectively,



Fig. 5. EC and pH of non-stabilized and stabilized soil after the 28th day of curing.

reaching a maximum of 10.59 for the S_{20} sample. This moderate alkalinity increase suggests effective stabilization without excessive *pH* elevation, which could otherwise lead to secondary environmental concerns.

For comparison, soil stabilized with alkaline-activated materials using conventional activators such as sodium hydroxide exhibits a much higher *pH*. As demonstrated by (Komaei et al., 2025), the *pH* of non-stabilized soil in their study was 7.77, whereas stabilization with slag activated by 2 *M* sodium hydroxide resulted in a significantly higher *pH* of 12.28. This indicates that while *CN-AS* effectively increases *pH* to enhance heavy metal immobilization, it does so within a controlled range, potentially reducing the risk of excessive alkalinity that may negatively impact soil ecosystems.

3.5. XRD analysis

Fig. 6 displays the *XRD* results for non-stabilized and stabilized samples with different *CN-AS* percentages.

The non-stabilized soil primarily contained Quartz (SiO₂), Calcite (CaCO₃), and Illite (KAl₂(Si, Al)₄O₁₀(OH)₂·nH₂O). Following stabilization with sodium carbonate-activated materials at varying dosages (5 %, 10 %, 15 %, and 20 % CN-AS) and 28 days of curing, additional crystalline phases were observed, including Anorthite (CaAl₂Si₂O₈), Ettringite (Ca₆Al₂(SO₄)₃(OH)₁₂·26H₂O), and Dolomite (CaMg(CO₃)₂).

These newly formed phases play a crucial role in the stabilization process. Specifically, Ettringite has been widely recognized for its role in heavy metal immobilization, while Anorthite contributes to the enhancement of *UCS* by forming a rigid matrix within the soil structure. Notably, the consistent presence of these phases at all tested *CN-AS* dosages (5 %, 10 %, 15 %, and 20 %) suggests that the stabilization mechanisms are active across a broad range of *CN-AS* contents. This finding aligns well with the observed improvements in both *UCS* and leachability, further reinforcing the efficacy of *CN-AS* in soil stabilization.

3.6. EDS, FE-SEM, and optical microscopy analysis

Fig. 7 displays the results of *EDS* analysis on both non-stabilized and stabilized soil samples. The primary elements identified in both cases were similar. However, following the treatment of contaminated soil with *CN-AS*, the weight percentages of *Ca*, *Si*, and *Al* increased by over 100 %, 55 %, and 35 %, respectively. Additionally, the weight percentages of the *Pb*, *Cr*, and *Cd* decreased by 45 %, 25 %, and 30 %, respectively.

Fig. 8 shows the results of *FE-SEM* analysis performed on two soil samples: (a) non-stabilized contaminated soil and (b) contaminated soil stabilized with 20 wt% of *CN-AS*.

In Fig. 8-a, compacted sand, silt, and clay particles are visible, but there is an absence of effective bonding, with a noticeable presence of void spaces that indicate mechanical weakness. However, after stabilizing the sample with *CN-AS*, dense, compact gels were formed between the soil particles, significantly enhancing strength and reducing leaching in the stabilized sample.

FE-SEM analysis also revealed the formation of aluminosilicate gel, specifically calcium aluminate silicate hydrate (CASH) in its amorphous form, between the soil particles. Additionally, Calcite (CaCO₃) crystals were observed, along with Ca(OH)₂ and Mg(OH)₂ crystals in the stabilized sample. A considerable amount of Ettringite crystals were also present, forming among and on the soil particles, which greatly improved the mechanical properties of the soil.

The results from *FE-SEM*, including the identified amorphous phases and crystals, align well with the *XRD* and *EDS* analysis results, further confirming their presence and the overall effectiveness of the stabilization process.

To gain deeper insights into the microstructural modifications of the stabilized soil samples, optical image analysis was employed. This



Fig. 6. XRD pattern of non-stabilized soil sample and stabilized soil sample.



Fig. 7. EDS analysis results: (a) non-stabilized soil sample and (b) Stabilized soil sample with 20 wt% of CN-AS.

technique offers a detailed understanding of the structural changes within the soil and the impact of varying *CN-AS* dosages on the soil's microstructure. High-resolution optical microscopy was used to capture the images, facilitating an in-depth examination of surface morphology and the degree of void reduction in the stabilized specimens.

Fig. 9 presents the optical images of the soil samples, with each panel corresponding to a different stabilization level. Figs. 9-a and 9-b display the non-stabilized soil sample, while Figs. 9-c, 9-d, 9-e, and 9-f showcase the soil samples stabilized with 5 %, 10 %, 15 %, and 20 % *CN-AS*,

respectively. As the *CN-AS* content increases, notable changes in the microstructure are observed. In the non-stabilized sample (Figs. 9-a and 9-b), large, visible voids are evident, which progressively decrease as the *CN-AS* content rises. At 5 %, 10 %, and 15 % *CN-AS* (Figs. 9-c, 9-d, and 9-e), the stabilizer effectively fills the voids, leading to a more compact and denser microstructure. However, at 20 % *CN-AS* (Fig. 9-f), the reduction in voids becomes minimal, indicating that the soil-stabilizer system has reached its saturation poly in the term of void-filling capacity.

These images provide valuable microstructural evidence that



Fig. 8. FE-SEM results: (a) non-stabilized sample and (b) Stabilized sample with 20 wt% of CN-AS.



Fig. 9. Optical image results: (a) and (b) non-stabilized sample, (c) Stabilized sample with 5 wt% of *CN-AS*, (d) Stabilized sample with 10 wt% of *CN-AS*, (e) Stabilized sample with 15 wt% of *CN-AS*, and (f) Stabilized sample with 20 wt% of *CN-AS*.

corroborates the findings from the previous sections, including permeability and unconfined compressive strength tests. The progressive reduction of voids with increasing *CN-AS* content aligns with the observed improvements in both strength and permeability, up to the optimal stabilizer dosage of 15 % *CN-AS*. Beyond this threshold, as shown in Fig. 9-f, the addition of further stabilizer has a diminished effect on the microstructure, highlighting the point of diminishing returns in the stabilization process.

3.7. Life cycle assessment (LCA)

This research has compared the evaluation of the environmental impact assessment of soil stabilization and solidification using *OPC* versus a mixture of calcium hydroxide and sodium carbonate-activated slag (*CN-AS*), employing a *LCA* methodology in line with ISO 14040 standards (*Environmental management-Life cycle assessment-Requirements and guidelines*, 2006). The analysis is limited to the production and

transportation of the requisite materials for stabilizing and solidifying one cubic meter of clayey sand contaminated with lead, chromium, and cadmium. The quantities of *OPC* and *CN-AS* were selected based on achieving equivalent *UCS* for both treatments.

3.7.1. Goal and scope definition

This study adopts a cradle-to-gate approach, encompassing all processes from raw material extraction to the point where the stabilizers are ready for use at the project site. The system boundary encompasses the extraction, processing, manufacturing, and transportation of raw materials required for producing *OPC* and *CN-AS*. Specifically, it considers raw material extraction and processing for slag, natural zeolite, Na₂CO₃, Ca(OH)₂, and *OPC*, along with their subsequent grinding and production. The transportation phase assumes a uniform distance of 100 km for all materials. These processes are marked with solid lines in Fig. 10, highlighting their inclusion in the system boundary.

Processes such as site preparation, mixing, compaction, and long-

Fig. 10. The system boundary considered for this LCA.

term service life considerations are excluded from this study, as they are identical for both stabilization methods and do not significantly impact the comparative results. These exclusions are denoted by dotted lines in Fig. 10. The focus of this *LCA* is on the upstream environmental burdens associated with producing *CN-AS* and *OPC*. The decision to exclude service life aspects aligns with prior *LCA* studies in construction materials (e.g. Ghadir et al. (2021)). The boundaries and assumptions used in this study are in line with common *LCA* practices to maintain a consistent and relevant comparison of material production impacts.

3.7.2. Life Cycle Inventory (LCI) data

In the *LCI* phase, all relevant inputs and outputs associated with the production processes were identified and quantified. Data for the materials used in the study—slag, natural zeolite, and *OPC*—were directly sourced from the manufacturers. For data gaps, the Ecoinvent v.3.11 database was referenced, and SimaPro input data was utilized for Na₂CO₃ and Ca(OH)₂. The detailed inventory, including material consumption and corresponding impact factors, is provided in Table S1 in the Supplementary Material to ensure transparency and reproducibility. The comparative analysis was conducted on samples prepared to achieve similar *UCS* levels.

After the 28th day of curing, the *UCS* of soil stabilized with varying amounts of *OPC* (5 %, 10 %, 15 %, and 20 %) were measured as 1439, 2803, 4555, and 6787 kPa, respectively. The *UCS* of a sample with 15 % *OPC* was found to be nearly equivalent to that of a sample stabilized with 20 % *CN-AS*.

The maximum dry density of the soil was determined according to ASTM D698, with a relative density of 90 %, resulting in a dry density of 1482.5 kg.m⁻³. Consequently, 222.4 kg of *OPC* was used to stabilization and solidification 1 m³ of lead, chromium, and cadmium-contaminated clayey sand, using 15 % OPC. Similarly, for stabilization and solidification with 20 % *CN-AS*, 296.5 kg of slag, 29.65 kg of natural zeolite, 14.82 kg of Na₂CO₃, and 14.82 kg of Ca(OH)₂ were used.

3.7.3. Life cycle impact assessment (LCIA)

A life cycle impact assessment (*LCIA*) was carried out to assess the potential environmental impacts using SimaPro software. The assessment compared two methods of stabilization and solidification contaminated soil: one using *OPC* and the other using *CN-AS*, across 11 environmental impact categories. The evaluation was conducted using the problem-oriented (mid-point) methodology, specifically the ReCiPe midpoint (H) method version 1.12. The results of this assessment are shown in Table 5, which presents the ReCiPe midpoint (H) outcomes for the production and transportation of the components used in *CN-AS*

(slag, natural zeolite, Na₂CO₃, Ca(OH)₂) and *OPC* across various impact categories. Based on these results, Fig. 11 illustrates the percentage contribution of different life cycle inventory for both methods.

In terms of climate change impact, *OPC* accounted for 75 % of the total contribution, while *CN-AS* contributed 25 %. The significant climate change impact from *OPC* is primarily due to the decomposition of calcium carbonate during cement clinker production, which releases up to 65 % of the total CO_2 emissions (Antunes et al., 2021). For freshwater ecotoxicity and marine eutrophication, both methods showed nearly equal and minimal contributions.

However, *CN-AS* had a higher impact than *OPC* in two environmental categories: particulate matter formation and ozone layer depletion, with *CN-AS* contributing approximately 60 % to these effects. In contrast, *OPC* showed a greater contribution to land acidification, anthropogenic toxicity, photochemical oxidant formation, and marine ecotoxicity accounting for approximately 60 %, indicating that *OPC* may be more polluting in these categories. Additionally, *OPC* contributed about three times more to terrestrial ecotoxicity compared to *CN-AS*. Lastly, in terms of metal depletion, *OPC* had a significantly higher impact, contributing 92 %, compared to just 8 % for *CN-AS*.

4. Discussion

This study provides a comprehensive evaluation of using $Ca(OH)_2$ and Na_2CO_3 -activated slag (*CN-AS*), in combination with natural zeolite, as a stabilization and solidification agent for lead, chromium, and cadmium-contaminated clayey sand. The experimental findings demonstrate significant advancements in mechanical strength, heavy metal immobilization, permeability reduction, and environmental impact mitigation.

The *UCS* results are particularly promising, demonstrating a substantial improvement in the mechanical integrity of the stabilized soils. The *UCS* of samples stabilized with 5–20 wt% *CN-AS* exceeded the minimum threshold of 350 kPa required by the *U.S. EPA* for landfill stabilization after 28th day of curing (Phojan et al., 2023; Yang et al., 2024). Specifically, the *UCS* of the sample containing 20 wt% *CN-AS* (S_{20}) reached 4449 kPa, over 75 times greater than the non-stabilized sample. This notable increase in strength can be attributed to the formation of C(*A*)SH gels and other durable reaction products. These reaction products act as binding agents within the soil matrix, enhancing its mechanical properties (Komaei et al., 2023a; Madadi and Wei, 2022). The results clearly show that the higher the *CN-AS* content, the more pronounced the strength improvement, indicating that slag plays a critical role in the development of the reaction products (Ghanbari et al.,

Table 5

ReciPe midpoint (*H*) method results for stabilization and solidification of 1 m³ functional unit of contaminated-clayey sand using 15 % of ordinary Portland cement and 20 % of Ca(OH)₂ and Na₂CO₃-activated slag with similar UCS.

Impact categories	$Ca(OH)_2$ and Na_2CO_3 -activated slag (<i>CN-AS</i>), combined with natural zeolite										Ordinary Portland cement (OPC)	
	Total	Slag production (296.5 kg)	Transport of slag (100 km)	Zeolite production (29.65 kg)	Transport of zeolite (100 km)	Na ₂ CO ₃ production (14.82 kg)	Transport of Na ₂ CO ₃ (100 km)	Ca(OH ₎₂ production (14.82 kg)	Transport of Ca (OH) ₂ (100 km)	Total	OPC production (222.4 kg)	Transport of OPC (100 km)
Climate change (kg CO ₂ eq)	58.71	44.47	2.28	1.42	0.23	6.82	0.11	3.26	0.11	190.75	189.04	1.71
Ozone layer depletion (kg CFC-11 eq)	$\begin{array}{c} 5.78 \times \\ 10^{-7} \end{array}$	3.60×10^{-7}	9.49 imes 10 ⁻⁸	6.52×10^{-8}	$9.49 imes$ 10^{-9}	2.67×10^{-8}	4.74 imes 10 ⁻⁹	1.63×10^{-8}	$\textbf{4.74}\times \textbf{10}^{-9}$	4.05×10^{-7}	3.34×10^{-7}	7.12×10^{-8}
Terrestrial acidification (kg SO ₂ eq)	$\begin{array}{c} 2.44 \times \\ 10^{-1} \end{array}$	1.78×10^{-1}	$rac{1.04}{10^{-2}} imes$	$\textbf{8.90}\times 10^{-3}$	$rac{1.04}{10^{-3}} imes$	2.37×10^{-2}	$5.19 imes$ 10^{-4}	2.07×10^{-2}	5.19×10^{-4}	$3.41 imes 10^{-1}$	$\textbf{3.34}\times \textbf{10}^{-1}$	$\textbf{7.78}\times 10^{-3}$
Marine eutrophication (kg N eq)	$2.13 imes 10^{-2}$	1.19×10^{-2}	$rac{1.19 imes}{10^{-3}}$	5.93×10^{-4}	$1.19 imes$ 10^{-4}	5.93×10^{-3}	5.93×10^{-5}	1.48×10^{-3}	$\textbf{5.93}\times 10^{-5}$	$rac{1.87 imes}{10^{-2}} imes$	1.78×10^{-2}	8.90×10^{-4}
Human toxicity (kg 1,4-dB eq)	8.18	5.93	7.71 imes10 ⁻⁴	2.97×10^{-2}	7.71 imes 10 ⁻⁵	1.78	3.85×10^{-5}	$\textbf{4.45}\times \textbf{10}^{-1}$	$\textbf{3.85}\times \textbf{10}^{-5}$	13.34	13.34	$\textbf{5.78}\times \textbf{10}^{-4}$
Photochemical oxidant formation (kg NMVOC)	$\begin{array}{c} 1.16 \times \\ 10^{-1} \end{array}$	1.01×10^{-1}	$rac{6.23}{10^{-3}} imes$	5.93×10^{-3}	$rac{6.23 imes}{10^{-4}} imes$	1.48×10^{-3}	$3.11 imes$ 10^{-4}	5.93×10^{-4}	3.11×10^{-4}	$rac{1.83}{10^{-1}} imes$	$\textbf{1.78}\times \textbf{10}^{-1}$	$\textbf{4.67}\times \textbf{10}^{-3}$
Particulate matter formation (kg PM10 eq)	$1.88 imes 10^{-1}$	1.80×10^{-1}	$2.08 imes$ 10^{-3}	2.97×10^{-3}	$rac{2.08 imes}{10^{-4}}$	2.96×10^{-3}	1.04 imes 10 ⁻⁴	1.48×10^{-3}	1.04×10^{-4}	$1.35 imes 10^{-1}$	1.33×10^{-1}	1.56×10^{-3}
Terrestrial ecotoxicity (kg 1.4-dB eq)	3.25×10^{-3}	2.97×10^{-3}	3.26×10^{-6}	1.19×10^{-4}	3.26×10^{-7}	1.19×10^{-4}	1.63×10^{-7}	$\textbf{4.45}\times 10^{-5}$	1.63×10^{-7}	8.90×10^{-3}	8.90×10^{-3}	$\textbf{2.45}\times 10^{-6}$
Freshwater ecotoxicity	7.57×10^{-3}	5.93×10^{-3}	4.45×10^{-6}	2.97×10^{-4}	4.45×10^{-7}	1.04×10^{-3}	2.22×10^{-7}	$\textbf{2.96}\times \textbf{10}^{-4}$	2.22×10^{-7}	8.90×10^{-3}	$\textbf{8.90}\times \textbf{10}^{-3}$	3.34×10^{-6}
Marine ecotoxicity	3.09×10^{-3}	$\textbf{2.97}\times \textbf{10}^{-3}$	1.48×10^{-6}	8.90×10^{-5}	1.48×10^{-7}	2.37×10^{-5}	7.41×10^{-8}	5.93×10^{-6}	$\textbf{7.41}\times 10^{-8}$	4.45×10^{-3}	$\textbf{4.45}\times 10^{-3}$	1.11×10^{-6}
Metal depletion (kg Fe eq)	1.60×10^{-2}	1.48×10^{-2}	3.85×10^{-4}	$\textbf{2.97}\times \textbf{10}^{-4}$	3.85×10^{-5}	$\textbf{2.96}\times 10^{-4}$	1.93×10^{-5}	1.48×10^{-4}	1.93×10^{-5}	1.78×10^{-1}	1.78×10^{-1}	2.89×10^{-4}

Fig. 11. The percentage contributions of each impact category are derived from the data shown in Table 5.

2024). Additionally, the increase in strength over time suggests ongoing pozzolanic reactions, which continue to enhance soil stability even beyond the initial curing period (Mozejko and Francisca, 2020).

The introduction of natural zeolite into the *S/S* system further enhances its performance, particularly in terms of heavy metal immobilization. Zeolite's high cation exchange capacity and porous structure provide additional binding sites for *Pb*, *Cr*, and *Cd*, facilitating their adsorption and subsequent immobilization within the soil matrix (Cadar et al., 2021). This is corroborated by the *TCLP* test results, which showed a substantial reduction in leached concentrations of these heavy metals. For instance, in the S_{20} sample, the reduction in *Pb* leachability reached 95 %, demonstrating the combined effectiveness of slag and zeolite in trapping these contaminants and preventing their release into the environment. Microstructural analyses, confirmed the formation of stable crystalline phases, such as ettringite, which contribute to the encapsulation and immobilization of heavy metals (Fan et al., 2023).

The study also highlights the role of Na₂CO₃ as an alkali activator. While Na₂CO₃ is effective in initiating the activation of slag, it is known for slower strength development compared to stronger activators like NaOH or Na₂SiO₃, primarily due to its lower *pH* (Akturk et al., 2019b; Komaei et al., 2024). However, the inclusion of Ca(OH)₂ in the mix significantly accelerates the strength gain. Calcium hydroxide raises the *pH*, facilitating faster aluminosilicate dissolution and enhancing the formation of C(*A*)SH and other binding phases (Xie and Liu, 2022). This dual-activator system creates an optimal environment for both rapid strength development and long-term stability, offering a sustainable alternative to more aggressive alkali activators.

The reduction in *EC* with increasing slag content is another critical finding, as it indicates successful heavy metal immobilization. As the heavy metals become chemically bound within the soil matrix, their mobility decreases, reducing the soil's conductivity (Caporale and Violante, 2016). This reduction in *EC* is a direct reflection of the stabilization process, where ionic species become less mobile, contributing to the long-term stability of the treated soil (Xia et al., 2019). Additionally, the rise in *pH*, reaching 10.59 in the *S*₂₀ sample, promotes the formation of stable mineral phases such as C(*A*)SH, further driving the immobilization of contaminants and the overall effectiveness of the *S/S* process (Chen et al., 2024).

curing age. The S_{20} sample exhibited reductions in *Pb*, *Cr*, and *Cd* leachate concentrations by 95 %, 92 %, and 98 %, respectively, compared to the non-stabilized soil. These reductions were well below the *EPA* regulatory limits, confirming the effectiveness of *CN-AS* in immobilizing these heavy metals.

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The observed exponential relationship between leachate concentrations and slag content is indicative of the critical role played by C(A)SH gels and mineral phases like Ettringite in binding heavy metals (Liu et al., 2024). These reaction products encapsulate the contaminants, preventing their migration and ensuring long-term containment. The cation exchange capacity of natural zeolite further enhances this immobilization by providing additional sites for heavy metal adsorption, making the overall system highly effective in reducing environmental risk (Senila and Cadar, 2024).

The permeability tests further validate the effectiveness of *CN-AS* in stabilizing contaminated soils. The permeability of the untreated clayey sand was 1.61×10^{-2} cm.s⁻¹, while the permeability of the S_{20} sample was reduced to 4.89×10^{-4} cm.s⁻¹, marking a 97 % reduction. This substantial decrease in permeability is essential for landfill and waste containment applications, as it minimizes the risk of contaminant migration and leachate escape. The continued reduction in permeability with time underscores the role of ongoing pozzolanic reactions, which progressively densify the soil matrix, enhancing its long-term impermeability (Das et al., 2022).

The slowing of permeability reduction beyond 15 % *CN-AS* is attributed to the diminishing effect of void filling in the soil-stabilizer matrix. Optical image analysis of the stabilized specimens (see Fig. 9) shows that, at lower *CN-AS* dosages (e.g., 10 % and 15 %), the stabilizer effectively fills the voids, leading to a denser microstructure and significantly reducing permeability. However, at higher dosages (e.g., 20 % *CN-AS*), the additional stabilizer contributes marginally to void reduction, as most of the available voids have already been filled. This microstructural evidence supports the observed trend in permeability reduction and aligns with the trends in *UCS* and leachability, where significant improvements were observed up to 15 % *CN-AS*, after which the rate of improvement diminished.

Microstructural analyses provide insight into the chemical and physical transformations occurring in the stabilized soil. The formation of new crystalline phases including Anorthite, Dolomite, and Ettringite, was identified via *XRD*. These phases contribute to the structural integrity and immobilization capacity of the soil (Reddy et al., 2020). *EDS* analysis revealed an increase in the concentrations of key elements like calcium, silicon, and aluminum, further supporting the formation of C(*A*)SH gels. *FE-SEM* images confirmed a denser soil matrix, with reduced void spaces between soil particles, leading to enhanced mechanical strength and reduced permeability.

From an environmental perspective, the LCA results underscore the sustainability advantages of using CN-AS over OPC. The climate change impact, as measured by CO2 emissions, was significantly lower for alkali-activated materials compared to OPC (Fernando et al., 2021; Segura et al., 2023). For example, the total climate change impact of the CN-AS was 58.71 kg CO₂ eq., compared to 190.75 kg CO₂ eq. for OPC. This reduction is largely due to the avoidance of the high-energy clinker production process required for OPC, highlighting the environmental benefits of using industrial by-products like slag (Sbahieh et al., 2023). Nevertheless, the LCA also identified some areas where CN-AS had higher environmental impacts, such as particulate matter formation and ozone depletion. These impacts can be mitigated by optimizing material sourcing and transport logistics. Overall, the use of CN-AS, combined with natural zeolite, offers a more sustainable alternative to OPC for contaminated soil stabilization, particularly in terms of carbon footprint reduction.

5. Conclusions

The *TCLP* and *ICP-OES* results demonstrate a noteworthy reduction in the leachability of *Pb*, *Cr*, and *Cd* with increasing *CN-AS* content and

The present study provides a detailed investigation into the

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stabilization and solidification of lead, chromium, and cadmiumcontaminated clayey sand with Ca(OH)₂ and Na₂CO₃-activated slag (*CN-AS*), combined with natural zeolite. The experimental results highlight substantial improvements in mechanical strength, reduced heavy metal leachability, lower permeability, and significant environmental benefits when compared to conventional methods. The key findings are summarized as follows:

- The UCS of stabilized soil showed a significant increase, with all treated samples exceeding the minimum requirement of 350 kPa for landfill stabilization set by the U.S. EPA. Specifically, samples treated with 20 wt% CN-AS achieved a UCS of 4449 kPa after 28 days of curing—over 75 times higher than the untreated soil. This strength enhancement is attributed to the formation of C(A)SH gels and secondary reaction products such as ettringite, as confirmed by XRD and FE-SEM analyses. The continuous strength gains suggest ongoing pozzolanic and alkali activation reactions.
- Leachability tests demonstrated the high efficiency of the stabilization/solidification (*S/S*) process in immobilizing heavy metals. Samples stabilized with 20 wt% *CN-AS* exhibited reductions in *Pb*, *Cr*, and *Cd* leachability of up to 95 %, 92 %, and 98 %, respectively, ensuring compliance with *EPA* regulatory limits. The addition of natural zeolite further enhanced heavy metal retention and reduced contaminant release.
- Permeability tests revealed a significant reduction in hydraulic conductivity, with up to 97 % lower values in samples stabilized with 20 wt% *CN-AS*. This suggests improved resistance to contaminant migration, reinforcing the suitability of the stabilized soil for containment applications.
- The *EC* of the soil decreased after stabilization and with increasing *CN-AS* content. The non-stabilized soil exhibited an *EC* of 4.37 mS. cm⁻¹, which progressively reduced to 1.91 mS.cm⁻¹ for the soil stabilized with 20 wt% *CN-AS*—a decrease of over 55 %. In contrast, the *pH* increased following stabilization, with higher *CN-AS* dosages resulting in a more pronounced rise. The *pH* of the non-stabilized soil was 7.91 and increased by 33.9 % in the sample with 20 wt% *CN-AS*, reaching a maximum value of 10.59.
- The *LCA* analysis indicated that the proposed *CN-AS* and zeolitebased stabilization method is an environmentally sustainable alternative to conventional *OPC*-based stabilization. This system reduced *CO*₂ emissions by 75 % while maintaining comparable *UCS* and heavy metal immobilization performance.

In conclusion, the application of *CN-AS* combined with natural zeolite proves to be a highly effective and sustainable method for the stabilization and solidification of *HMs*-contaminated soils. The significant improvements in mechanical strength, heavy metal immobilization, and permeability reduction, alongside the lower environmental impact demonstrated by *LCA*, position this technique as a promising alternative to traditional cement-based stabilization methods.

Supplementary data to this article can be found online at https://doi. org/10.1016/j.scitotenv.2025.179612.

CRediT authorship contribution statement

Alireza Komaei: Writing – original draft, Visualization, Resources, Project administration, Methodology, Investigation, Data curation, Conceptualization. Arman Moazami: Resources, Methodology, Investigation. Pooria Ghadir: Writing – review & editing, Validation, Investigation. Abbas Soroush: Writing – review & editing, Supervision. Akbar A. Javadi: Writing – review & editing, Supervision.

Consent for publication

All authors have read and approved this manuscript.

Ethics approval and consent to participate

This research work does not involve human participants, human data, or human tissue.

Funding

Open access funding was provided by the University of Strathclyde.

Declaration of competing interest

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

Data availability

Data will be made available on request.

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