Cadmium Pollution in Mining Areas: Chemical Stabilization Strategies and Their Performance

Xiaodi Sun

The experimental high school attached to Beijing normal university, Beijing, China

xiaodisun873@outlook.com

Abstract. Cadmium (Cd) contamination in mining regions poses significant risks to ecological security and human health. This study synthesizes evidence from more than thirty field investigations, laboratory experiments, and theoretical analyses published between 2000 and 2024. The objectives are fourfold: (i) to summarize the primary sources, concentration ranges, and spatial distribution of Cd in mining environments; (ii) to examine the health hazards associated with Cd exposure; (iii) to compare the mechanisms, removal efficiencies, applicability, and remediation outcomes of various chemical stabilization agents; and (iv) to evaluate monitoring indicators and assessment methods for restoration effectiveness. Literature retrieval focused on mine pit case studies, concentration data, pollutant migration patterns, health risk documentation, and remediation practices, with extracted data subjected to comparative analysis. Results indicate that Cd is highly enriched in tailings, smelting areas, and downstream plains, exhibiting both vertical and horizontal migration. Human exposure occurs mainly through inhalation and food chain accumulation, leading to multiple health disorders. Among chemical stabilizers, $CaCO_3$ provides rapid but temporary immobilization, $FeCl_3$–citric acid achieves efficient Cd extraction but may impair soil quality, while natural zeolite offers moderate yet durable stabilization at relatively low cost. Overall, no single technique satisfies all sustainability requirements. Site-specific strategies, such as combining zeolite treatment with controlled pH adjustment, coupled with long-term monitoring, are essential to ensure the safe management and potential reuse of Cd-contaminated mining land.

Keywords: Mining areas, Cadmium, Chemical stabilization, Immobilization, Soil remediation.

1. Introduction

Mining and smelting activities constitute the dominant anthropogenic source of cadmium (Cd) in terrestrial environments, with global mine-waste inventories reporting soil Cd totals ranging from 0.8 to 1 969 mg kg^{-1} [1−3]. Once released, Cd migrates vertically via "surface enrichment–deep precipitation" and spreads horizontally through fluvial transport and dust [4-6]. Chronic exposure through inhalation and the food chain has been epidemiologically linked to renal tubular dysfunction, cardiovascular morbidity, reproductive disorders and skeletal demineralisation [7-9].

Despite these case-specific findings, three persistent knowledge gaps in the systematic understanding of Cd contamination remain. First, quantitative, high-resolution maps of Cd concentration ranges and spatial patterns across contrasting mining environments remain scarce [10-12]. Second, field evidence comparing the long-term performance of alkaline amendments, $FeCl_3$–citric acid washing and natural-zeolite stabilisation across variable soil chemistries is limited to short-term studies with durations of less than 90 days [13-16]. Third, no integrated monitoring and decision framework currently links source identification, pathway analysis and technology selection for Cd-contaminated mine soils [17-21].

To address these gaps, this study pursues three measurable objectives: (i) quantify the occurrence ranges and spatial patterns of Cd in mining-impacted soils and sediments through high-resolution sampling and geostatistical mapping; (ii) compare the effectiveness of alkaline amendments, $FeCl_3$–citric acid washing and zeolite stabilisation by reductions in DTPA (diethylenetriaminepentaacetic acid, molecular formula $C_{14}H_{23}N_3O_{10}$), a chelating agent widely applied in industry and medical diagnostics, - and TCLP (Toxicity Characteristic Leaching Procedure), a standard method for evaluating the leachability and mobility of heavy metals in solid media or waste.extractable Cd, concurrent changes in soil pH, CEC (cation-exchange capacity, which refers to the number of

millimoles of cations adsorbed or exchanged by each kilogram of soil or colloid from the surrounding solution.) and nutrient status, as well as recording the treatment cost per cubic meter of soil, and (iii) propose and validate a monitoring and decision framework that guides sustainable remediation of Cd-polluted mining land [22-26].

2. Methodology

Following a concise PRISMA workflow, we searched SCI-hub and Google Scholar (2000–2024) with the string: ("cadmium" OR "Cd") AND ("mine*" OR "tailings" OR "smelting") AND ("soil*" OR "sediment*") AND ("immobilization" OR "stabilization" OR "washing"). Studies were included only if they (i) reported field data from mining areas, (ii) provided total Cd and DTPA-Cd or TCLP-Cd, and (iii) evaluated at least one of the following: $CaCO_3$, $FeCl_3$–citric acid, or natural zeolite ($n \geq 6$). Three fully documented cases were selected: $CaCO_3$ stabilization (Chang 2013, rapid fixation in highly acidic soils), $FeCl_3$–citric acid washing (Gao 2017, high removal of acid-soluble Cd), and natural zeolite stabilization (Zhang 2021, low-cost, high-CEC sediment passivation). Data on pH, CEC (cmol kg^{-1}), Cd (mg kg^{-1}), biomass and cost (USD t^{-1}) were harmonized; values below the detection limit were set to LOD/2. A random-effects model was applied to compare the technologies [27-31].

However, the literature data may have limitations in comparability due to experimental year differences, regional differences, and differences in testing methods.

3. Current status of Cd pollution in mining areas

3.1. Data on heavy metal concentration in mine pits

Mining and smelting activities are the main sources of cadmium pollution in mining areas. During ore extraction and processing, cadmium may be released during crushing, beneficiation, and the treatment of cadmium-bearing minerals. High-temperature smelting of metals such as zinc and lead generates cadmium-containing flue gas, wastewater, and slag. These pollutants may enter the environment through flue gas emissions, waste water emissions or improper solid waste treatment. For example, during zinc smelting, cadmium may be released in gaseous or liquid form and enter the environment with flue gas or wastewater.

Heavy metals are discharged into adjacent environmental media during mining and subsequently accumulate in soils, leading to elevated Cd concentrations in mining districts [14].

Soil contaminants are commonly used to assess environmental pollution. The mean Cd concentration reached 1.08 ± 0.36 mg kg^{-1} in smelting zones, significantly higher than the 0.84 ± 0.63 mg kg^{-1} observed in mining areas. All sampled soils in the mining area had Cd concentrations exceeding the background value of 0.108 mg kg−1 in the soil A horizon of Jiangxi Province and the Soil Environment Quality Risk Control Standard for Soil Contamination of Agricultural Land. In the smelting area, 89.3% of samples surpassed the standard threshold values. These findings indicate severe Cd contamination in both areas [26].

Table 1. Summary statistics for Cd in several mine districts (in-situ analyses)

mg/kg Cd	All sites	Clare Pb	Glendalough	Silvermines	Tynagh
n	1432	47	381	24	134
Minimum	0.0	0.0	0.0	0.0	0.0
Maximum	1969	505	660	728	1969
Median	0.0	0.0	51	89	57
Mean	39	59	63	198	129

Note: A value of 0.0 indicates result < detection limit

The table summarizes the Cd statistics of different mining areas (such as Claire lead mine, Glendaloch, etc.), with the with a maximum value of 1969 mg kg^{-1}, a median of 0.0 mg kg^{-1}, and a

mean of 39 mg kg^{-1} respectively, indicating that there are significant differences in the Cd concentration in the mine pit or related areas [15].

3.2. Spatial Distribution of Cd in mining areas

The spatial distributions of Cd in sediments and soils are influenced by topography, water velocity, channel width, changes in physicochemical properties, and mineral composition of the upper river.

The migration path of Cd is controlled by the river flow rate, river channel width and terrain undulation. For example, the influx of the Taiping River changes the hydrodynamic conditions, resulting in the reactivated Cd in the sediments of the alluvial plain [17]. Surface runoff is the main driving force for Cd migration, especially in northern arid mining areas (such as North China), where Cd spreads along the downstream of the river [18].

Dabaoshan Mine is one of China's major large−scale polymetallic sulfide mines. Cd is the most prominent polluting element in this mining area, showing significant enrichment and spatial heterogeneity in both soil and sediments. The concentration of Cd in the soil: decreases from the tailings pond toward the river terrace, but rebounds slightly after the Taiping River flows through the alluvial plain. Cd concentration in sediments: In contrast to soil, it increases from tailings burrows to the river terrace, and slightly decreases after the Taiping River flows into the alluvial plain [17].

3.3. The migratory route of Cadmium in mining areas

During the mining of polymetal sulfides, metal sulfides oxidize and release a large amount of cadmium ions and acidic wastewater, and enter natural water bodies (such as rivers) through surface runoff or percolation [19]. Then, acid wastewater causes soil acidification in downstream rice fields, promotes the dissolution and migration of cadmium, and causes soil cadmium accumulation (such as the cadmium concentration in the topsoil of lead-zinc mining areas is as high as 23.36 mg/kg) [20]. Last, mining waste slag and dust spread to surrounding soil through atmospheric sedimentation or sewage irrigation, forming a pollution gradient of "mining area → smelting area → riparian zone" [21].

Vertical migration:

Cadmium shows a bimodal distribution of "surface enrichment, deep precipitation" in the soil profile [20]: Topsoil (0–20 cm) is activated due to biological activities (such as plant roots, microorganisms). Cadmium is mainly reductible (Fe/Mn oxide binding), oxidizable state (organic matter binding) and acid exchangeable state (accounting for 60–80%), and has strong mobility [20]. Deep (>0.8 m), Due to leaching, cadmium seeps down with water and precipitates and enriches in clay or carbonate cracks due to leaching [20].

Organic matter (OC) is a key factor driving migration: high OC content enhances the complexing capacity of cadmium and promotes its leaching migration in lead-zinc mining areas; low OC areas (such as uncontaminated areas) cadmium is prone to retention in the clay layer [20].

Horizontal diffusion:

Cadmium in the soil in the riparian zone migrates to the river through hydraulic action, polluting downstream farmland [21]. The geological background source of cadmium (such as cadmium-rich shale weathering) is transported through rivers to pollute downstream soil, and is adsorbed and retained by iron sediments during migration [22].

Plant absorption and food chain transmission. Certain tolerant plant species, such as Buddleja davidii and Artemisia lavandulaefolia, absorb cadmium primarily through their root systems. Although transfer coefficients may approach or slightly exceed unity (e.g., TF = 0.97), enrichment coefficients remain below one, suggesting that Cd is largely retained in roots rather than undergoing strong translocation to aboveground tissues [23]. Within plants, the distribution generally follows the order: roots > stems > leaves. Cadmium fixed in roots may subsequently return to soils via leaf fall or litter decomposition [23,24]. In agricultural systems, Cd readily enters the food chain. It is absorbed by rice and vegetables, ultimately reaching humans through dietary intake [19]. In pasture environments, Cd is transferred along the "soil → forage → milk" pathway, while the application of

phosphate fertilizers further exacerbates this process [25]. The overall migration and transmission routes of cadmium in mining areas are illustrated in Fig. 1.

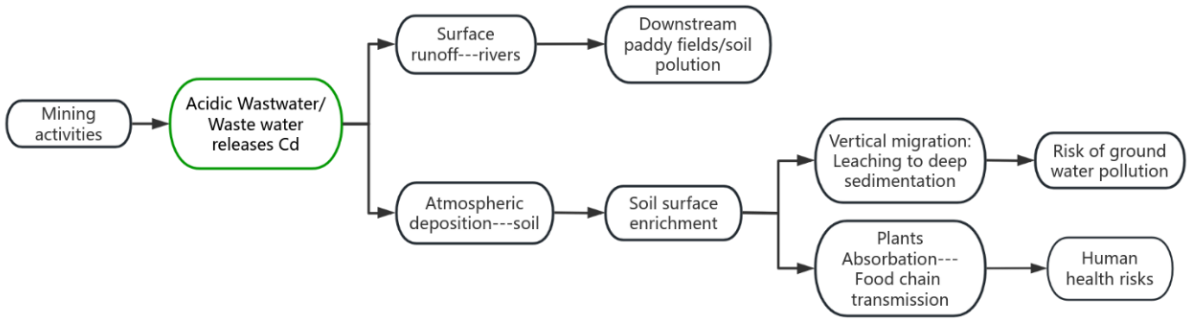


Fig. 1 The Migration Routs of Cadmium

4. Environmental health impacts of Cd pollution

Environmental Cd poses significant risks to human health due to its widespread presence in the environment and its ability to accumulate in the body over time. Cd exposure can occur through various routes, including inhalation of contaminated air, ingestion of contaminated food and water, and dermal contact with contaminated soil or dust. Once absorbed, Cd can accumulate in various organs and tissues, exerting toxic effects on multiple physiological systems [27].

One of the primary targets of Cd toxicity is the kidneys, where Cd accumulates and causes damage to the renal tubular epithelial cells, leading to kidney dysfunction and ultimately renal failure. Cd-induced kidney damage is thought to result from oxidative stress, inflammation, and disruption of cellular signaling pathways [27].

Cadmium induces the alteration of steroidogenesis, disorders of the menstrual cycle and reproductive hormones, delay in puberty and menarche, pregnancy loss, premature birth, and reduced birth weight [27].

Epidemiological evidence indicates that cadmium (Cd) exposure is positively correlated with increased morbidity and mortality from cardiovascular diseases. The strongest associations are with hypertension and atherosclerosis, although a few studies have reported potential anti-atherosclerotic effects. Cd exposure generally elevates blood pressure—particularly diastolic pressure—and reduces heart rate. The cardiotoxic mechanisms are multifactorial. Cd induces oxidative stress by inhibiting key antioxidant enzymes such as glutathione peroxidase (GSH-PX) and superoxide dismutase (SOD), thereby promoting lipid peroxidation. It disrupts calcium homeostasis and causes intracellular Ca^{2+} overload, leading to endothelial dysfunction and enhanced sodium and water retention. Furthermore, Cd damages red blood cells and directly injures cardiomyocytes, altering their electrophysiological properties. These alterations include reduced action potential amplitude, decreased maximum depolarization rate, and shortened action potential duration, which together impair cardiac performance through interference with Na^{+}, Ca^{2+}, and K^{+} ion channel activity [27].

The harmful effect of Cd on the human body was discovered as lung damage. Symptoms of respiratory tract irritation, such as dry throat, pain, runny nose, dry cough, chest tightness, difficulty breathing, dizziness, fatigue, and joint aches, can appear within 4 to 10 h after inhaling large amounts of Cd vapor. These symptoms are like influenza. In severe cases, individuals may develop bronchopneumonia lung edema (alveolar distention, alveolar wall hypertrophy), as well as the filling of swollen alveoli with a large amount of protein slurry. Additionally, bronchial mucosal epithelial cells may undergo degeneration, necrosis, and shedding, accompanied by pulmonary capillary dilation, congestion, and high levels of pulmonary interstitial water, which can eventually lead to death [27].

Cadmium damages bone health by triggering oxidative stress, interfering with calcium metabolism, activates inflammatory signals and causes cellular dysfunction, impairs bone health and increases the risk of fractures. Specifically, cadmium exposure promotes excessive production of reactive oxygen

species (ROS), causing oxidative damage to bone cells, destroying proteins, lipids and DNA, and inhibiting the activity of antioxidant enzymes such as superoxide dismutase and glutathione peroxidase, thereby aggravating oxidative stress. At the same time, cadmium causes mitochondrial dysfunction, leading to insufficient energy and apoptosis of osteoblasts, further weakening bone formation. In addition, cadmium may also affect the expression of bone metabolism-related genes through epigenetic changes such as DNA methylation and histone modification [27].

5. Chemical stabilization and remediation strategies

5.1. Alkaline materials($CaCO_3$)

Alkaline stabilization mechanism. Alkaline materials, such as $CaCO_3$, are commonly used to immobilize Cd in soils of mining areas. The key mechanism involves the reaction of Cd^{2+} with CO_3^{2-} to form the insoluble precipitate $CdCO_3$ [28].

The principal reaction can be expressed as:

$$Cd^{2+} + CO_3^{2-} \rightarrow CdCO_3 \quad (1)$$

Carbonate ions are typically derived from the partial dissolution of $CaCO_3$ or related carbonate compounds. Dissolved $CaCO_3$ releases CO_3^{2-}, but the dissolution process is strongly pH-dependent. Under acidic conditions, $CaCO_3$ tends to dissolve to form Ca^{2+} and HCO_3^-, whereas under alkaline conditions (pH > 10), HCO_3^- is converted to CO_3^{2-}, favoring $CdCO_3$ precipitation [32]. In addition, HCO_3^- may indirectly contribute through equilibrium conversion or react directly with Cd^{2+} to yield $CdCO_3$, with concomitant release of water and CO_2 [28]:

$$Cd^{2+} + 2HCO_3^- \rightarrow CdCO_3(s) + H_2O + CO_2 \quad (2)$$

A basic environment (pH > 8) is therefore critical for $CdCO_3$ formation. Elevated pH promotes the shift from HCO_3^- to CO_3^{2-} ($HCO_3^- \rightleftharpoons CO_3^{2-} + H^+$ (3)), thereby enhancing the precipitation efficiency [32]. When $CaCO_3$ acts as the carbonate donor, the released Ca^{2+} may compete with Cd^{2+} for CO_3^{2-}, forming $CaCO_3$ rather than $CdCO_3$. However, due to the much lower solubility product (Ksp) of $CdCO_3$ ($\approx 3.3 \times 10^{-12}$), Cd^{2+} preferentially precipitates [28]. The initial concentrations of Cd^{2+} and CO_3^{2-} further influence the precipitation kinetics: higher concentrations accelerate the reaction but may result in finer precipitate particles [28].

Thermodynamically, the precipitation of $CdCO_3$ is an exothermic process ($\Delta H < 0$), with negative Gibbs free energy, indicating spontaneity and irreversibility under environmental conditions [28]. The actual rate of immobilization is further modulated by stirring intensity, temperature, and ionic strength [32].

An experiment of stabilizing Cd from contaminated soil by using $CaCO_3$ was conducted in Taiwan [29]:

Contaminated topsoil (0–20 cm) from northern Taiwan (pH 4.42, Cd 8.71 mg kg^{-1}) was air-dried, sieved to <2 mm, and amended with 0.1 % (w/w) $CaCO_3$. The mixture was incubated at 25°C, 70–90 % RH, 12 h light for 90 days. pH and DTPA-Cd were measured every 15 days; Tessier fractionation was performed on day 90. Chinese cabbage was then grown for 40 days before Cd analysis [29].

The results come out that shoot Cd concentration decreased significantly from 24 mg kg^{-1} in the control to 16 mg kg^{-1} with amendment (33 % reduction), while plant biomass increased, demonstrating effective mitigation of Cd phytotoxicity [29]. In addition, $CaCO_3$ treatment quickly increased soil pH from 4.40 ± 0.12 to 6.35 ± 0.21[29] within 15 days, with significant results, but pH declined steadily thereafter, reaching 5.35 ± 0.14 [29] by day 90, indicating rapid neutralization in strongly acidic soil [29]. Moreover, DTPA-extractable Cd dropped from 4.35 ± 0.11 mg kg^{-1} [29] in the control to 2.12 ± 0.14 mg kg^{-1} [29] at 15 days (51 % reduction) and was lowest at 30 days (1.96 ± 0.10 mg kg^{-1}, 54 % reduction [29]). However, By day 90 the value had rebounded to 3.43 ± 0.12 mg kg^{-1} [29]—only 11 % below the control, with statistical significance greatly diminished [29].

In conclusion, $CaCO_3$ rapidly raises soil pH and reduces available Cd by ~50 % within weeks through $CdCO_3$ precipitation, thereby lowering plant uptake. However, its buffering capacity is rapidly exhausted in strongly acidic soils, leading to a pronounced rebound in available Cd by 90 days. $CaCO_3$ is thus suitable as an emergency immobilization agent; for long-term stability, other stabilization strategies should be proposed [29].

5.2. Chemical washing with $FeCl_3$and citric acid (CTA)

5.2.1. Mechanism

$FeCl_3$- citric acid (CTA) chemical washing has two steps: First, Acid solution - complexation synergy. $FeCl_3$ hydrolysis produces H^+, lowers pH, and promotes the dissolution of Acid-soluble Cd ($CdCO_3$, $Cd(OH)_2$) [30]:

$$CdCO_3 + 2H^+ \rightarrow Cd^{2+} + CO_2 \uparrow + H_2O \quad (3)$$

Cl^- and Cd^{2+} form stable soluble ions $[CdCl_4]^{2-}$ to inhibit readsorption. Second, Citric acid (CTA) synergy. CTA further provides H^+ and forms a Cd-citrate complex with Cd^{2+}, enhancing dissolution; CTA can also reduce and dissolve the Fe-Mn oxide envelope to release the encapsulated Cd. Finally, Cd is transferred from the "acid-soluble + reducing" state to the eluent and discharged from the soil with the liquid phase [30].

5.2.2. Data analysis

The test soil was collected from Zhuzhou, Hunan Province, air-dried, sieved (2 mm), and characterized by an initial Cd concentration of 3.1 mg kg^{-1} and pH 5.22. Three eluents were applied: (i) 10 mmol L^{-1} $FeCl_3$, (ii) 20 mmol L^{-1} citric acid (CTA), and (iii) 10 mmol L^{-1} $FeCl_3$ + 20 mmol L^{-1} CTA. Soil suspensions were prepared at a liquid-to-solid ratio of 2:1 and shaken at 25 °C (180 rpm) for 6 h. Elution was carried out either once or in three successive steps, with eluents replaced after centrifugation and filtration, followed by two rinses with deionized water. After washing, Cd concentrations in the filtrates were determined to calculate removal efficiency. Soil samples were further analyzed by the BCR sequential extraction procedure to assess Cd speciation, while soil pH, cation exchange capacity (CEC), and exchangeable Ca and Mg were also measured [30].

Results. Under the “three washes and two rinses” protocol, single-dose $FeCl_3$ achieved a Cd removal rate of 60.3%, which increased to 64.0% with the addition of CTA. When multi-step elution was applied, Cd removal rose to 79.6% with $FeCl_3$ and 78.9% with the combined $FeCl_3$–CTA system. Acid-soluble Cd accounted for 77.3% of the total removal. However, soil quality declined after treatment: pH decreased from 5.22 to 3.71, CEC from 11.4 to 8.1 cmol kg^{-1}, and exchangeable Ca and Mg were substantially reduced [30].

Discussion. The $FeCl_3$–CTA washing method achieved high Cd removal (≈79%), primarily due to the high proportion (74%) of acid-soluble Cd that is readily mobilized by acid-complexing reactions [30]. Combined eluents performed better than single agents, and multi-step elution proved more effective than single-step treatment. Nevertheless, adverse effects were evident, including soil acidification, reduced CEC, and nutrient depletion. Post-treatment management should therefore involve lime application, organic amendments, and supplementation with Ca, Mg, and P fertilizers to restore soil fertility. This approach represents a chemical “removal” rather than “stabilization” strategy and is most suitable for Cd-contaminated soils dominated by the acid-soluble fraction where rapid remediation is required [30].

5.3. Mineral stabilizer (natural zeolite)

5.3.1. Mechanism

The mineral stabilizer is composed of natural zeolite crystals supplemented with small amounts of Ca-, Mg-, Fe-, and Mn-bearing salts and clay. Zeolite possesses a high cation-exchange capacity

(CEC ≈ 140 meq 100 g^{-1}) and a negatively charged aluminosilicate framework. At near-neutral pH (~7.5), Cd^{2+} readily exchanges with charge-balancing cations within the lattice [31]:

$$Cd^{2+}\ (solution) + 2Na^{+}\ (zeolite) \rightarrow Cd^{2+}\ (zeolite) + 2Na^{+}\ (solution) \quad (4)$$

Due to its stronger affinity for the zeolite surface compared with Ca^{2+}, Mg^{2+}, Fe^{2+}, or Mn^{2+}, Cd^{2+} is immobilized through formation of sparingly soluble surface complexes and interlayer precipitates. This process substantially decreases Cd mobility and bioavailability in soils [31].

5.3.2. Data analysis

To test this method, an experiment had been conducted: First, take the sample of Cd exceeding the standard (Cd >4 mg kg^{-1}, above the risk control value), and air-dry it through a 2 mm screen. Second, set up 10 groups, each group of 200 g, and mineral stabilizers of 0 %, 2 %, and ≥2 % were added, respectively. Then, add deionized water at a solid-liquid ratio of 1:0.5, mix well and seal at room temperature (15 °C) for 7 days, and air-dry naturally for later use. Last, the leaching rate of Cd was measured by toxic leaching test (TCLP similar method), and the mean was taken after repeated 3 times [31].

Table 2. Comparison of three cadmium remediation techniques for contaminated soils.

Treatment	Technology type	Primary mechanism	Applicable conditions	Removal / stabilization efficiency	Cost	Limiting factors
$CaCO_3$	Chemical stabilization (immobilization)	pH increase → negative surface charge ↑ → Cd^{2+} adsorption & $CdCO_3$/$Cd(OH)_2$ precipitation	Strongly acidic soils (pH < 5.5) with acid-soluble Cd	~50 % DTPA-Cd reduction in 90 d; plant Cd ↓ 24 → 16 mg kg^{-1}	Low (~50–200 USD t^{-1} soil)	Low buffering capacity; rapid acid neutralization; no total-Cd reduction; long-term instability
$FeCl_3$–CTA	Chemical washing (extraction)	Acid dissolution + chloride complexation + citrate chelation: $FeCl_3$ hydrolysis → H^+ dissolves Cd; Cl^- forms $[CdCl_4]^{2-}$; CTA reduces Fe-Mn oxides and chelates Cd	Soils where acid-soluble / reducible Cd ≥ 70 %	78.9 % Cd removed (TFA soil); 45 % (THA); 5.3 % (TGA)	Moderate (~500–800 USD t^{-1} soil)	Soil acidification (pH ↓ 1.5–2.5); CEC & nutrient loss; secondary waste stream requiring treatment
Zeolite mineral	Physico-chemical stabilization (immobilization)	High-CEC (140 meq/100 g) cation exchange: Cd^{2+} replaces Ca^{2+}/Mg^{2+}/Fe^{2+} on zeolite surface forming insoluble surface complexes	River sediments or soils with high clay/OM and Cd > risk control value	Cd leaching rate ↓ 33.3 → 14.3 % at 2 % dose; further increments ineffective	Low (~50–200 USD t^{-1} soil)	Total-Cd unchanged; efficiency capped by high organic matter/clay; requires nutrient replenishment

The result comes out that without the curing agent, the Cd leachate rate was 33.3%. After adding 2% stabilizer, the leachate rate dropped to 14.3%, a decrease of 57%. The doping amount was continued to increase (2.5%–6%), and the leaching rate was stable at around 14%, with no significant difference ($P>0.05$). It shows that 2% is a cost-effective dose, and over-administration contributes limited to further reduction of leaching rates [31].

In conclusion, through the cation exchange-surface complexing mechanism, mineral stabilizers can reduce the Cd leachate by >50% under low doping (2%) conditions, significantly reducing their environmental mobility and biotoxicity. Granted, since river sediments are rich in organic matter, some Cd co-precipitated with soluble organic complexes, limiting the space for further decline, but overall it can still meet the risk control requirements. To sum up, The technique is low-cost, easy to apply, rapid (7 days), and preserves soil nutrients, making it suitable for rapid stabilization of Cd-contaminated dredged sediment; subsequent lime pH adjustment and organic-fertilizer amendment can ensure safe site reuse [31].

Table 2 compares those three Cd stabilization and remediation methods' mechanisms, applicable conditions, removal/stabilization rate, costs, and their limiting factors and situations.

5.4. Common monitoring techniques

All three studies adopted an integrated framework covering "soil basic properties – total metals – bioavailable fraction – speciation – plant verification". Soil pH was measured at a 1:2.5 soil-to-water ratio, CEC by the NH_4OAc–NaOAc method, and organic matter by dichromate oxidation. Total Cd was determined after HCl/HNO_3 or aqua regia digestion, followed by FAAS, GF-AAS or ICP-AES quantification; bioavailable Cd was extracted with DTPA (pH 7.3) or TCLP. Sequential extraction (Tessier or BCR) was used to distinguish exchangeable, carbonate-bound, Fe-Mn oxide-bound, organic and residual fractions. Plant validation used pot-grown or field crops such as Chinese cabbage, rice or tall fescue to measure shoot Cd concentration, biomass and BCF (Bioconcentration Factor, used to quantify the ability of chemicals to be absorbed and enriched by organisms simply from water or soil)/TF. Blanks, duplicates and certified reference materials (recovery 95–105 %) were included, with statistical analysis by one-way ANOVA plus LSD/Tukey tests [28-31].

5.5. Common data interpretation method

The unified interpretation is based on the three-step method of "baseline-change-compliance": the risk screen value or control group concentration C_0 is used as the baseline to calculate the removal rate $(C_0-C_1)/C_0\times100\%$[28-31] or the stability rate $(L_0-L_1)/L_0\times100\%$[28-31]; the pH-effectiveness index graph or exponential attenuation/dose-effect curve is used to describe the biological effectiveness relationship of pH-Cd; the morphological changes are considered to be successful in passivation, and the acid-soluble state is considered to be successful in leaching; the plant verification is based on crop Cd<0.4 mg kg^{-1}, and BCF>1 is considered to be super enriched [28-31].

6. Conclusion

Published data indicate that total Cd concentrations in mine-affected soils range from 0.8 to 1 969 mg kg^{-1}, with elevated levels typically observed in tailings, smelting sites, and downstream alluvial plains. Cd exhibits distinct migration patterns, including vertical redistribution characterized by "surface enrichment–deep precipitation" and horizontal dispersion through fluvial transport and atmospheric deposition. Human exposure occurs predominantly via dust inhalation and dietary intake, contributing to renal, cardiovascular, reproductive, and skeletal disorders. In terms of chemical remediation, $CaCO_3$ provides rapid but short-lived immobilization; $FeCl_3$–citric acid washing achieves efficient Cd extraction but may induce secondary soil degradation; while natural zeolite offers moderate yet long-lasting immobilization at relatively low cost. Compared to previous studies, this review systematically synthesizes the origin, partial distribution, migration pathways, and health effects of Cadmium from former research papers; It also analyses and compares three of chemical stabilization and remediation methods mentioned in previous literature. The limitations are: evidence is overwhelmingly laboratory-scale or greenhouse; field performance under realistic mining areas, climate and redox fluctuations is scarce. Microbial activity, enzyme profiles and food-chain risk indices are rarely reported, while cost analyses are restricted to unit amendment prices and ignore full life-cycle or waste-stream disposal expenses. Last, inter-study comparability is hampered by

divergent extraction methods, risk thresholds, detection limits, and experimental locations and time. Future research should conduct field tests in multi-climate mining areas for ≥3 years to monitor Cd re-migration; synchronously measure microbial communities and enzyme activities; use a unified extraction method and reference soil to achieve comparable data; and calculate the entire life cycle costs such as raw materials-transport-waste liquid treatment to support sustainable restoration decisions.

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