2026, 66(6): 2634-2656
CSTR: 32112.14.j.AMS.20250998
DOI: 10.13343/j.cnki.wsxb.20250998

微生物学报
*Acta Microbiologica Sinica*
http://journals.im.ac.cn/actamicrocn

Review 综述

# 微生物诱导碳酸钙沉淀技术协同固碳与重金属固定的研究进展

罗正宇[1,2*]，何煌[1]，韩洪瑞[1]，倪文[1]，范翼寻[1]，郭玉[1]，黄昕昳[1]，刘振宇[1]，童晋[1]，陈志[1]

1 成都理工大学 生态环境学院，四川 成都
2 生态环境部水土污染协同控制与联合修复重点实验室，四川 成都

罗正宇, 何煌, 韩洪瑞, 倪文, 范翼寻, 郭玉, 黄昕昳, 刘振宇, 童晋, 陈志. 微生物诱导碳酸钙沉淀技术协同固碳与重金属固定的研究进展[J]. 微生物学报, 2026, 66(6): 2634-2656.
LUO Zhengyu, HE Huang, HAN Hongrui, NI Wen, FAN Yixun, GUO Yu, HUANG Xinyi, LIU Zhenyu, TONG Jin, CHEN Zhi. Research progress in microbially induced carbonate precipitation technology for synergistic carbon sequestration and heavy metal immobilization[J]. Acta Microbiologica Sinica, 2026, 66(6): 2634-2656.

**摘　要：**全球气候变化与土壤重金属污染对传统治理技术的协同适配性提出了更高要求。微生物诱导碳酸钙沉淀(microbially induced carbonate precipitation, MICP)技术凭借其独特的生物代谢与环境交互特性，为固碳与重金属稳定化的协同治理提供了新途径。该技术通过脲酶与碳酸酐酶(carbonic anhydrase, CA)介导的 2 条核心酶促路径诱导碳酸钙生成，可同步实现 $CO_2$ 矿化封存与重金属固定。在固碳场景中，MICP 技术可通过岩性改良提升碳封存场地的地质稳定性，并依托高效矿化反应增强固碳效率；在重金属修复场景中，其可通过吸附、共沉淀及表面络合等多重机制实现重金属稳定化，且不同碳酸钙晶型可适配差异化的污染特征。然而，当前 MICP 技术的规模化应用仍面临三大瓶颈：功能菌株对极端环境的耐受性不足、外源菌株与土著生态系统的适配性冲突，以及固碳与重金属固定代谢路径的耦合障碍。针对上述问题，本文提出“重金属固定-矿化固碳-长期监测”3 阶段协同工艺流程假设，通过时序性切换代谢路径，从理论上解决了重金属固定与矿化固碳的 pH 需求矛盾，为 MICP 技术的工程化应用提供了新思路。未来研究应聚焦极端生境功能菌株的适应性改造、外源-土著微生物互作机制调控及工艺参数精准优化，推动该协同模式从理论设计走向现场验证，为碳中和目标实现与污染土壤安全利用提供技术支撑。

**关键词：**微生物诱导碳酸钙沉淀(MICP)；固碳减排；重金属污染修复；岩性改良；协同治理

---

资助项目：国家科技重大专项(2025ZD1205103, 2025ZD1205105)
This work was supported by the National Science and Technology Major Project (2025ZD1205103, 2025ZD1205105).
*Corresponding author. E-mail: luozhengyu15@cdut.edu.cn
Received: 2025-12-30; Accepted: 2026-03-25; Published online: 2026-04-01

# Research progress in microbially induced carbonate precipitation technology for synergistic carbon sequestration and heavy metal immobilization

**LUO Zhengyu[1,2*], HE Huang[1], HAN Hongrui[1], NI Wen[1], FAN Yixun[1], GUO Yu[1], HUANG Xinyi[1], LIU Zhenyu[1], TONG Jin[1], CHEN Zhi[1]**

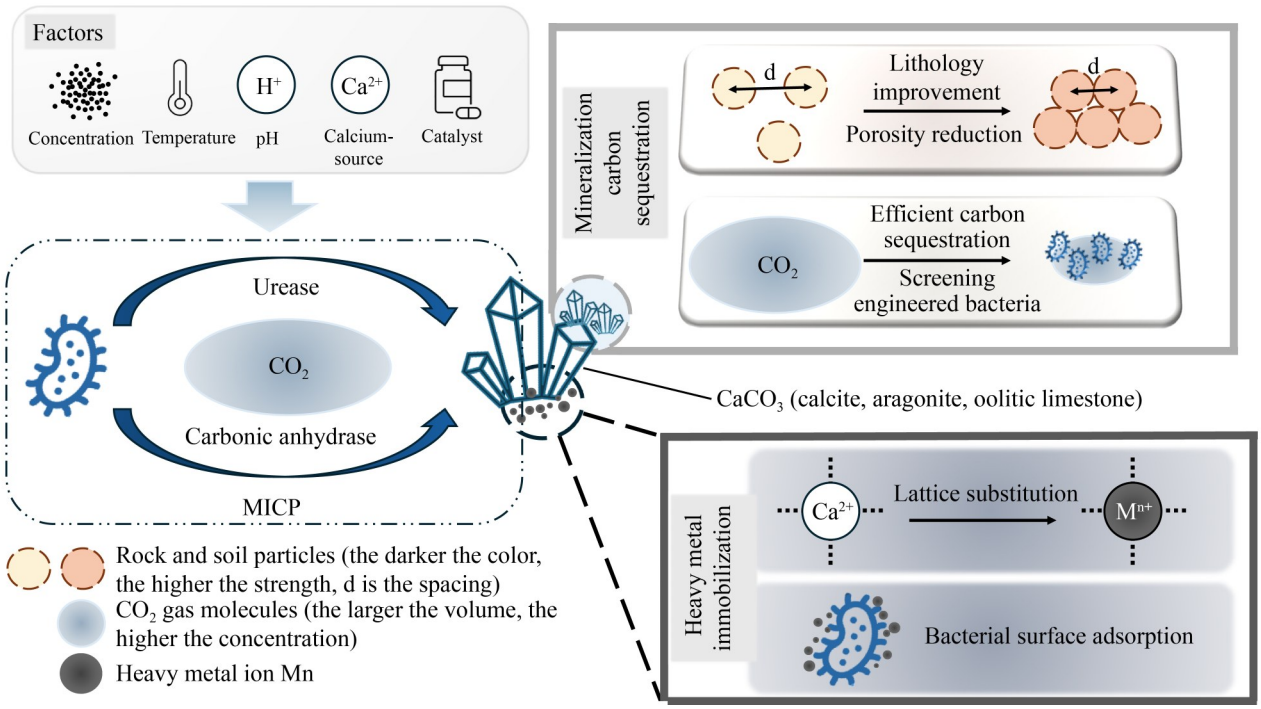
1 College of Ecology and Environment, Chengdu University of Technology, Chengdu, Sichuan, China

2 Key Laboratory of Synergistic Control and Joint Remediation of Soil and Water Pollution, Ministry of Ecology and Environment, Chengdu, Sichuan, China

**Abstract:** Global climate change and soil heavy metal pollution have raised higher requirements for the synergistic adaptability of conventional remediation technologies. Microbially induced carbonate precipitation (MICP) technology, with its unique biological metabolism and environmental interaction characteristics, provides a new pathway for the synergistic management of carbon sequestration and heavy metal stabilization. This technology induces calcium carbonate formation through two core enzyme-mediated pathways involving urease and carbonic anhydrase, enabling simultaneous $CO_2$ sequestration by mineralization and heavy metal immobilization. In carbon sequestration scenarios, MICP technology can enhance the geological stability of carbon sequestration sites through lithological improvement and strengthen carbon sequestration efficiency through high-efficiency mineralization reactions. In heavy metal remediation scenarios, it can achieve heavy metal stabilization through multiple mechanisms such as adsorption, co-precipitation, and surface complexation, and different calcium carbonate crystal forms can adapt to varied pollution scenarios. However, the large-scale application of MICP technology currently faces three major bottlenecks: insufficient tolerance of functional strains to extreme environments, compatibility conflicts between exogenous strains and native ecosystems, and coupling barriers between metabolic pathways for carbon sequestration and heavy metal immobilization. To address these issues, this paper proposes a three-stage synergistic process flow hypothesis for heavy metal immobilization, carbon sequestration by mineralization and long-term monitoring. Sequentially switching metabolic pathways theoretically resolves the pH requirement conflict between heavy metal immobilization and carbon sequestration by mineralization, providing new solutions for the engineering application of MICP technology. Future research should focus on the modification of functional strains for extreme habitats, regulation of interactions between exogenous and native microorganisms, and precise optimization of process parameters, to advance this synergistic model from theoretical design to on-site validation, providing technical support for achieving carbon neutrality and the safe utilization of polluted soils.

**Keywords:** microbially induced carbonate precipitation (MICP); carbon sequestration and emission reduction; heavy metal pollution remediation; lithological improvement; synergistic management

Graphical abstract:




全球气候变化已成为制约人类社会可持续发展的核心挑战。政府间气候变化专门委员会(Intergovernmental Panel on Climate Change, IPCC)第 6 次评估报告第一工作组指出，2011–2020 年全球地表平均温度较工业化前升高 1.09 ℃，其中约 1.07 ℃归因于人类活动影响；同时，为实现《巴黎协定》的温控目标，全球碳排放需在 2025 年前达到峰值；然而 2019 年全球温室气体总排放量达 590 亿 t $CO_2$ 当量，较 2010 年增长约 12%，较 1990 年增长 54%[1]。现有减排技术尚难以填补目标路径下的排放缺口，因此亟需发展并推广新型碳捕集与封存技术[2]。

与此同时，土壤重金属污染形势严峻，与气候变化共同制约全球环境可持续发展。农业及矿区土壤污染尤为突出，镉(Cd)、铅(Pb)、砷(As)、锌(Zn)等污染物在全球广泛分布，主要源于工业排放及农业化学品施用[3-5]。这些污染物不仅抑制土壤酶活性、干扰作物根系养分吸收，导致生态系统功能退化，还会通过土壤-作物-人体的食物链富集作用对人类健康和生命安全构成潜在威胁[6-7]。

当前，固碳与重金属污染叠加的双重压力日益凸显，传统治理技术的局限性随之暴露。在重金属固定方面，传统的物理化学方法治理对象单一，且对当地生态扰动较大，存在二次污染风险，难以作为长期治理手段[8-9]。在固碳方面，碳捕集与封存(carbon capture and storage, CCS)技术成本高、存在泄漏风险，而植物固碳周期长、受地域制约，难以适应当下的治理需求[10-12]。近年来，随着绿色生物技术的快速发展，微生物修复技术因其环境友好特性逐渐成为污染治理领域的研究热点。其中，微生物诱导碳酸钙沉淀(microbially induced carbonate precipitation, MICP)技术凭借其独特的生物代谢机制与环境交互产生的多重效应展现出突出的协同调控潜力，为同步实现碳固定与重金属钝化提供了创新途径[13-15]。基于此，本文系统综述 MICP 技术在减污固碳领域的基本原理、协同

机制及应用前景，以期为相关领域的研究与实践提供理论参考。

# 1 MICP 技术的发展历程与基本原理

## 1.1 发展历程

MICP 技术的研究发端于 20 世纪初，其发展历程可划分为 5 个阶段(图 1)[16-24]。1911 年，Drew[16]首次从自然环境中分离出可诱导碳酸盐沉淀的细菌，奠定了该技术的微生物学基础。1973 年，Boquet 等[17]通过实验室研究证实土壤细菌具备生成 $CaCO_3$ 晶体的能力，标志着 MICP 技术概念的正式确立。1992–2001 年间，Kantzas 等[18]发现该技术可显著降低砂岩孔隙度，首次揭示其在岩土工程领域的应用潜力；Bang 等[19]继而将聚氨酯固定的巴氏芽孢八叠球菌(*Sporosarcina pasteurii*)用于混凝土裂缝修复，证实该方法可提升混凝土抗压强度，拓展了 MICP 技术在建筑材料领域的研究方向。2004–2010 年为技术成熟化阶段，Whiffin[20]正式提出“微生物诱导碳酸钙沉淀”的学术术语，Mitchell 等[21]推动了 MICP 在岩土工程中的系统化应用，Jonkers 等[22]则提出基于 MICP 的微生物自修复混凝土设计理念，促使相关研究在生物建筑材料领域形成热潮。2010 年以来，MICP 技术进入协同发展新阶段，Okyay 等[23]对不同脲解菌群的固碳能力开展系统研究，发现其固碳效率存在显著差异，推动了该技术在 $CO_2$ 矿化固碳领域的应用；Gilmour 等[24]近期通过基因工程手段改造枯草芽孢杆菌(*Bacillus subtilis*)，构建出可同时利用碳酸酐酶(carbonic anhydrase, CA)路径的新型工程菌株，实现了矿物产量提升与 $CO_2$ 捕获的协同增效，为固碳与重金属修复的联合治理提供了新的技术途径。

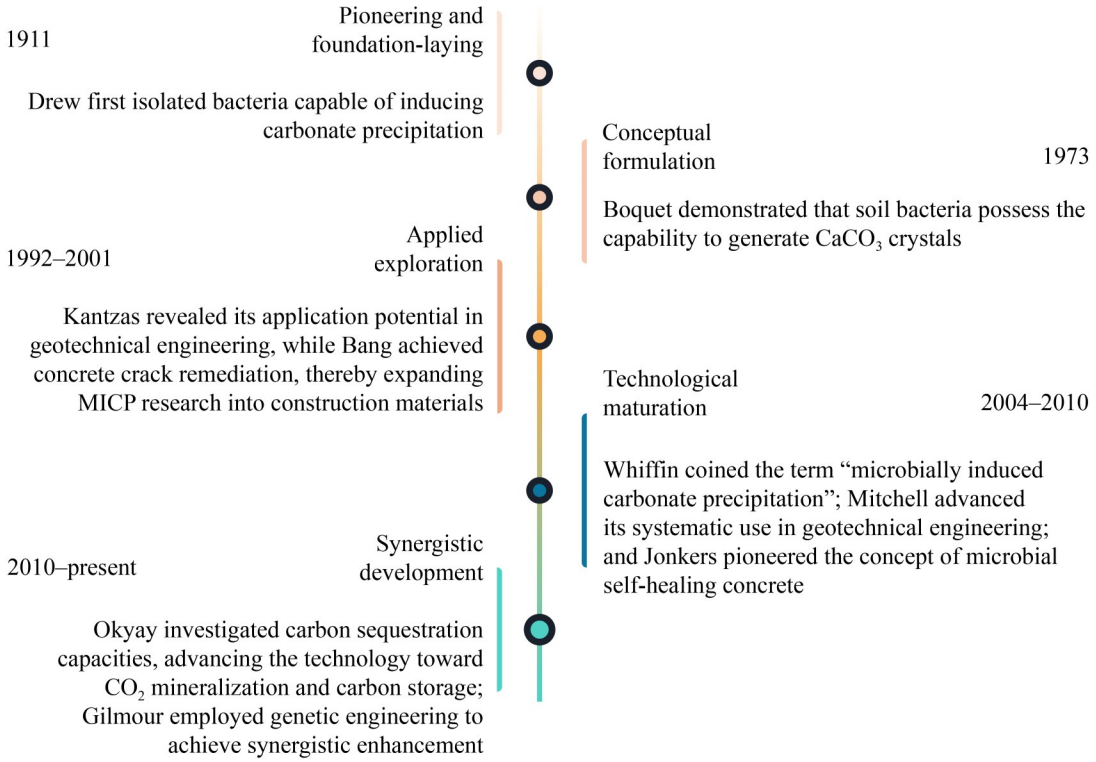



**图1 MICP技术发展时序[16-24]**

Figure 1 MICP technology development timeline[16-24].

## 1.2 核心代谢路径

MICP 技术的核心在于利用微生物代谢作用介导碳酸钙沉淀，该过程主要通过脲酶与碳酸酐酶(carbonic anhydrase, CA) 2 条关键酶促路径实现。

脲酶路径以双 $Ni^{2+}$为核心的活性中心催化尿素的水解反应，以典型的球状脲酶为例，核心催化步骤是双 $Ni^{2+}$极化尿素羰基启动水解，释放氨和碳酸，氨进一步水解提升溶液 pH，促使碳酸分解为 $CO_3^{2-}$，最终 $CO_3^{2-}$与 $Ca^{2+}$结合形成 $CaCO_3$ 沉淀[25]，详见图 2[26]。

CA 则通过以 $Zn^{2+}$为核心的活性中心催化 $CO_2$ 的水合反应，在自然状态下，$CO_2$ 的水合反应速率十分缓慢，一级反应速率常数仅为 0.13 $s^{-1}$，CA 作用下其反应速率常数可高达 $1.4\times10^7$ $s^{-1}$，达到自然情况下的 $10^8$ 倍左右[27]。以最常见的 α 型 CA 为例，其核心催化步骤是 $Zn^{2+}$极化水分子生成亲核羟基，最终将 $CO_2$ 转化为 $HCO_3^-$，随后 $HCO_3^-$分解与 $Ca^{2+}$结合生成沉淀[28-30]，详见图 3[31]。

上述路径形成的 $CaCO_3$ 存在方解石、文石、球霰石 3 种晶型，三者可相互转化，转化速率受 pH 及温度显著影响[32-33]。目前 MICP 技术的应用可大致分为矿化固碳和重金属固定 2 个方向。

在矿化固碳领域，MICP 的核心机制在于通过微生物代谢活动直接驱动 $CO_2$ 实现永久碳封存，CA 在将 $CO_2$ 水合为碳酸氢根过程中发挥作用，脲酶则有助于维持促进钙化的碱性 pH 值[34]。除了上述 2 种路径外，硝酸盐还原及光合作用等代谢途径也可驱动碳酸盐沉淀。例如，硝酸盐还原菌通过产生碱性环境促进碳酸盐形成，而光合微生物通过消耗 $CO_2$ 提高 pH，间接诱导沉淀[35-36]。MICP 产生的 $CaCO_3$ 沉淀能够填充岩石孔隙、封堵裂缝，并使岩体抗压强度提升，有效增强地质封存安全性[37-38]。

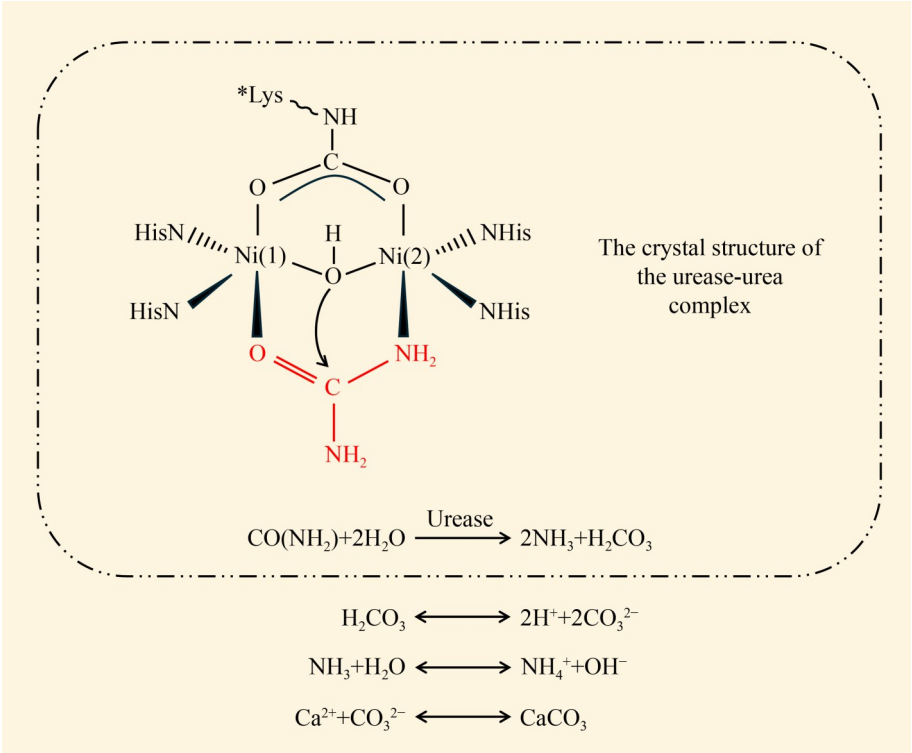



**图2 脲酶机制示意图[26]**

Figure 2 Schematic diagram of the urease mechanism[26].

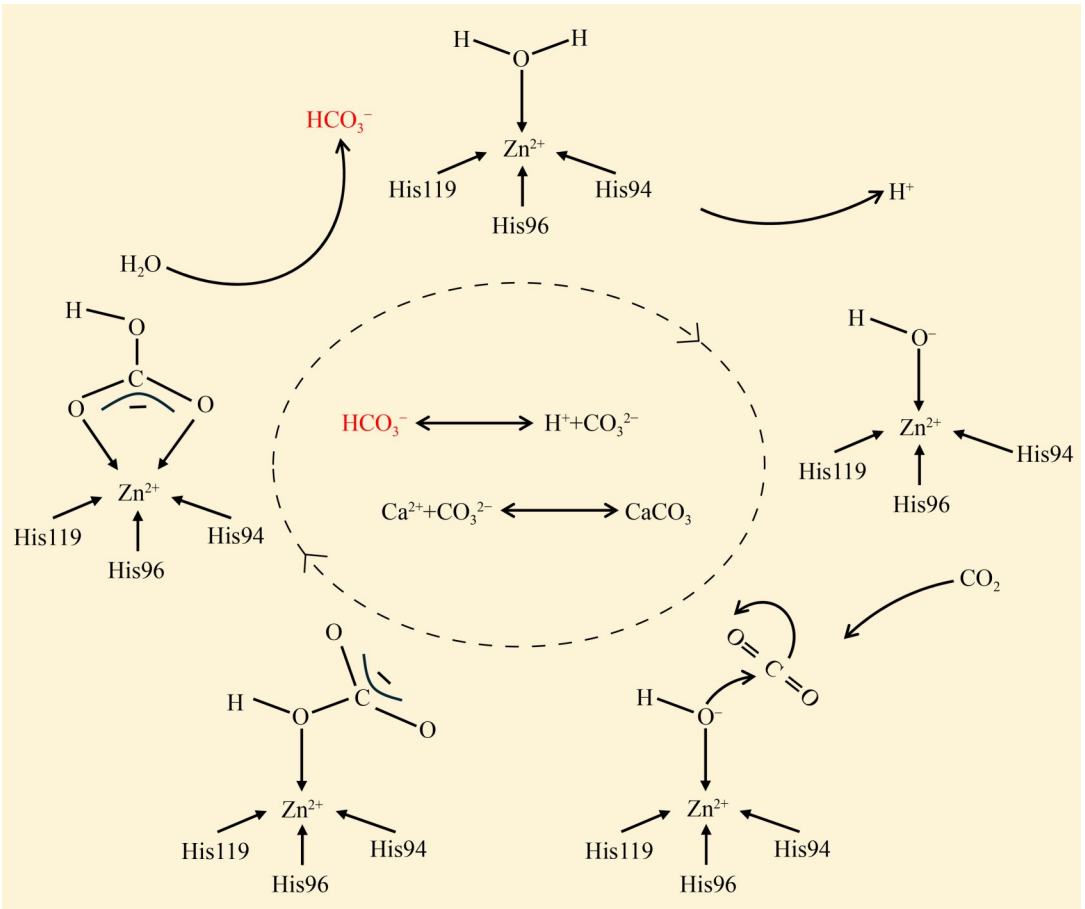



**图3 CA机制示意图[31]**

Figure 3 Schematic diagram of the CA mechanism[31].

在重金属固定领域，MICP 技术主要通过微生物表面吸附和晶格取代等机制实现对重金属的固定。微生物分泌的胞外聚合物(extracellular polymeric substances, EPS)富含羟基、羧基和氨基等官能团，可通过静电引力、离子交换或络合作用吸附重金属离子，降低其生物有效性[15]；同时，对于离子半径与 $Ca^{2+}$ 相近的重金属，可通过同晶替换取代方解石晶格中的 $Ca^{2+}$ 形成固溶体，实现稳定包埋[39]；此外，MICP 过程中生成的碳酸盐矿物可通过共沉淀将还原后的金属离子固定于晶格或包裹于矿物内部，形成低迁移性的碳酸盐结合态[14]。

## 1.3 碳酸钙晶型的形成与环境行为

自然环境中，碳酸钙结晶常以无定形碳酸钙(amorphous calcium carbonate, ACC)为前驱相，而非直接从溶液中成核[40]。纯 ACC 遵循“脱水-溶解-再结晶”的机制，首先通过快速脱水转化为球霰石，再经溶解-沉淀转化为热力学更稳定的方解石，后者速率约为前者的十分之一[41]。然而，该结晶路径并非固定不变，可受诸多外界因素调控，如在富镁条件下镁离子可诱导 ACC 使其直接脱水形成文石而非球霰石[42]。类似地，富含羧基的多肽类有机添加剂，如聚天冬氨酸，同样能调控结晶路径，通过稳定 ACC 前驱相、抑制其快速脱水，从而阻断 ACC 到球霰石的转化，并诱导 ACC 采用非经典颗粒附着机制形成特定形貌的方解石[43]。值得注意的是，ACC 在生物体系中并非简单的不稳定中间产物，而是

具有特定短程有序结构的瞬时前驱相；这种结构可预设最终晶型，并通过与有机基质的相互作用形成复杂形态的生物矿物[40]。

不同碳酸钙晶型的重金属固定能力差异显著。球霰石因比表面积大、表面能高，对铜、镉等重金属离子表现出优异的吸附去除效率，文石具有针状或柱状晶体结构能提供一定吸附位点，但受限于比表面积较小，主要通过表面络合作用实现固定，方解石作为最稳定晶体，吸附速率较快但容量较低，常通过晶格取代形成稳定的重金属碳酸盐固溶体[44-46]。

# 2 MICP 技术的关键影响因素

MICP 技术的效能取决于微生物-环境-底物之间的复杂相互作用，其关键影响因素可分为生物因素与非生物因素两大类[47]。

## 2.1 生物因素

MICP 技术的效能受多种生物因素调控，主要包括微生物的种间差异和极端环境适应性。

微生物的种间差异显著影响 MICP 的效率与路径，Okyay 等[23]对 15 个不同的脲解菌群进行了研究，结果表明所有群落都表现出不同的固碳能力和 MICP 速率，固碳率在 0−86.4% 之间不等。另有研究表明，脲酶基因的丰度与碳酸钙沉淀量呈正相关，这可能是导致不同菌种间 MICP 能力存在显著差异的原因之一[48]。此外，不同种类的微生物具有多样的代谢途径，它们可通过改变微环境 pH 值与离子浓度，间接调控碳酸钙沉淀的动力学过程，如产甲烷古菌的乙酸降解途径、酿酒酵母的有机酸盐呼吸代谢途径和厌氧甲烷氧化群落的甲烷氧化路径，均能够在升高微环境 pH 的同时，产生溶解性无机碳(dissolved inorganic carbon, DIC)，促进 MICP 作用[49-51]。

不同种类的微生物在不同极端环境下的适应机制是决定 MICP 技术广泛应用可行性的关键。MICP 微生物可通过多种策略适应盐度、温度及重金属胁迫等极端环境。在 Hu 等[52]的研究中，芽孢八叠球菌属(*Sporosarcina*)、节杆菌属(*Arthrobacter*)可与其他嗜盐微生物构成稳定的脲解菌群，这种群落协同作用可共享抗盐物质或代谢产物，分散高盐胁迫对单一菌种的压力并提升整体系统的稳定性，使其在总溶解固体约 65 g/L 的高盐环境下仍保持 96% 的 $Ca^{2+}$去除率。Shen 等[53]从红球菌属(*Rhodococcus*)中分离的菌株具有显著的耐寒性，在低至 5 ℃的温度下仍保持生长和脲酶活性，展现出在寒冷环境中实施 MICP 技术的潜力。Jiang 等[54]研究表明，在重金属胁迫下，路氏肠杆菌(*Enterobacter ludwigii*)将优先激活胞外多糖屏障、硫化物沉淀和硫醇螯合三重防御系统对抗毒性，再通过脲酶驱动的 MICP 进程固定重金属，同时保留吲哚-3-乙酸合成、铁载体产生和磷溶解等植物促生能力，这种协同机制使其在重金属胁迫下仍能高效发挥作用。

## 2.2 非生物因素

MICP 技术的效能还受非生物因素制约，这些因素可系统归纳为生长环境因素与外源调控因素两大类。生长环境因素指微生物生存与代谢依赖的全部基础条件，其中可分为原生环境条件和生长营养基质 2 个核心部分。外源调控因素则是人为施加的、非菌株生长必需的辅助性物质或工艺调控手段，包括生物炭、壳聚糖等改性添加剂。具体研究如表 1 所示。

深入分析表 1 可以发现，生长环境因素是 MICP 进程的核心决定因子，其中 pH、盐度、温度等原生环境条件直接作用于菌株的酶活性，决定代谢反应的基础速率[55-58]；而钙、磷、有机碳源等生长营养基质则从反应底物供给层面出发，通过不同营养元素的代谢响应机制调控反应动力学进程，2 类生长环境因素的适配是 MICP 反应高效发生的前提[35,59-60]。外源调控因素则通过施加辅助性物质改善微生物所处的微环境，缓解其逆境胁迫，进而间接影响微生物酶活性、调控其代谢反应路径，同时为碳酸钙结晶提供额外吸附成核点位，进一步强化 MICP 反应的效率与产物稳定性[36,61-66]。

**表1 影响MICP技术的非生物因素**

Table 1 Abiotic factors affecting MICP technology

| Abiotic factor categories | Factors | Impact effect | References |
|---|---|---|---|
| Growth environmental factors | Phosphorus | In the low-phosphorus environment of crystal fountains, the microbial phosphorus starvation response reshapes community structure and functions, indirectly regulating photosynthesis-driven MICP, optimizing the precipitation microenvironment, and enhancing $CaCO_3$ precipitation | [35] |
| | pH | Urease conformation and microbial activity are optimal at pH 7.0−8.0, yielding maximum $CaCO_3$ precipitation. Extreme pH causes urease unfolding, suppressed metabolism, and reduced precipitation. $CO_2$ solubility and capture peak near pH 8.9 in urea-regulated systems, but decline sharply at higher pH from excess urea | [55-56] |
| | Salinity and pH | In intertidal microbialites, salinity significantly affects dominant bacteria, while pH correlates more strongly with community composition. Under high salinity, microbes prefer urea and amino acids over nitrogen fixation; urea decomposition and amino acid ammonification increase pH and DIC, synergistically driving carbonate precipitation | [57] |
| | Temperature | In the 10−30 ℃ range, higher temperatures yield greater but faster-declining urease activity. Low-temperature MICP performs better: at 10 ℃, $CaCO_3$ precipitation in aqueous solution and sand column experiments is 92% and 37% higher than at 30 ℃, respectively | [58] |
| Growth environmental factors | Urea and $Ca^{2+}$ | Extra urea in complex media slows nitrate reduction, and increased $Ca^{2+}$ significantly inhibits it. Optimizing the coupling of urea hydrolysis and nitrate reduction enhances $CaCO_3$ precipitation by 20%−30%, up to 14 g/L | [59] |
| | Organic carbon source | Molasses and distiller's grains as organic carbon sources effectively promote MICP-related $CaCO_3$ formation, stabilize soil surfaces, and significantly reduce wind erosion-induced soil loss, with distiller's grains performing better. When tryptone soya broth is used, high sodium input disperses soil aggregates and increases soil loss | [60] |
| Exogenous regulatory factors | Trehalose and sodium alginate | Trehalose promotes calcite formation, and sodium alginate stabilizes vaterite *via* carboxyl groups. Both enhance calcium ion removal, with higher concentrations yielding higher precipitation rates. Sodium alginate also induces $CaCO_3$ nucleation on its surface, improving MICP efficiency | [61] |
| | Chitosan (CTS) | Chitosan adsorbs $Mn^{2+}$, alleviates its toxicity to *Sporosarcina pasteurii*, improves bacterial adaptability and urease activity, regulates $MnCO_3$ crystal morphology to finer and more uniform particles, and significantly enhances MICP efficiency for Mn removal while shortening remediation time | [62] |
| | Artificial humic acid and biochar | Biochar provides pores for *Sporosarcina pasteurii* adhesion, and artificial humic acid acts as a nutrient to stimulate urease secretion. Their synergy promotes coprecipitation of cadmium with calcium carbonate, significantly improving Cd immobilization efficiency by MICP. They also enhance soil fertility and ecological functions, with a notable increase in the relative abundance of dominant phyla such as *Proteobacteria* | [63] |

(待续)

(续表 1)

| Abiotic factor categories | Factors | Impact effect | References |
|---|---|---|---|
| | Chitosan and carboxymethyl chitosan (CMCS) | CTS and CMCS enhance MICP efficiency by providing nucleation sites, promoting $Ca^{2+}$ binding, and synergistic mineralization with bacteria. CMCS shows higher compatibility and stronger promotion than CTS, raising calcium precipitation up to 28.28% and driving the transformation from calcite to aragonite | [64] |
| | Magnesium polypeptide (MP) | MP elevates soil pH and organic matter, provides adsorption sites, and synergizes coprecipitation of $Ca^{2+}$, $Mg^{2+}$, and $Pb^{2+}$, significantly enhancing MICP remediation efficiency for lead-contaminated soil. It reduces exchangeable Pb by 30.37% and increases carbonate-bound Pb by 40.82%, while mitigating MICP's negative impacts on soil microbial communities | [65] |
| | Sand surface modification | Modifying sand grain mineral composition, specific surface area and charge promotes bacterial adsorption and growth, accelerates nitrate reduction-mediated $CaCO_3$ precipitation and optimizes crystal morphology and distribution, thus improving MICP solidification efficiency and engineering performance for sandy soil | [36] |
| | Modified organic materials (MOM) | For severely disturbed soil in degraded meadows of the Qinghai-Xizang Plateau, MOM significantly increases urease and phosphatase activities, accelerating soil nutrient cycling. It also boosts microbial abundance, stabilizes community structure, and enhances the environmental adaptability of MICP | [66] |

总体而言，生长环境因素与外源调控因素均未改变菌株的核心矿化代谢机制，仅通过直接或间接途径调控酶活性与代谢进程，是基于菌株固有特性的效能优化手段。

# 3 MICP 技术的应用前景

## 3.1 MICP 技术在矿化固碳中的应用

MICP 技术通过微生物代谢活动催化 $CO_2$ 与钙源反应生成碳酸钙矿物，实现 $CO_2$ 的矿化固定，其应用可大致分为岩性改良与高效固碳 2 类方向。不同菌株与技术路径在性能表现、适用场景及技术局限上呈现差异化特征(表 2)。

在岩土体改良领域，功能特异性菌株针对极端地层环境、特殊介质及工程材料损伤展现出定制化修复潜力。地衣芽孢杆菌(*Bacillus licheniformis*)可耐受 50 ℃高温、10 MPa 高压及高 $CO_2$ 分压的极端地层条件，通过诱导碳酸盐沉淀使砂岩孔隙率从 12.61% 降至 10.75%，同时分泌 EPS 形成生物膜封堵孔隙通道，降低 $CO_2$ 泄漏风险，但该研究仅基于砂岩-超临界 $CO_2$-卤水静态水溶液体系开展，与原位动态地层环境存在显著差异，长期原位应用中微生物营养供给的持续性、非目标岩性的适配性及地质尺度的封存效应仍待系统验证[67]；本土耐盐木糖葡萄球菌(*Staphylococcus xylosus*)在高盐岩土介质改良中表现出独特适配性，可在可溶性盐含量为 8%−12% 的高离子强度环境中稳定诱导球霰石型碳酸钙沉淀，1 周内使高盐粉质土柱孔隙率降低 6.12%、内聚力增加 39.5%，并在不同围压条件下显著提升抗剪强度，但其当前研究仅完成单轮短期注浆实验，技术的长期稳定性、循环处理效能及对其他类型高盐岩土的适用性尚未明确，菌株性能定向优化与现场应用技术体系仍需完善[69]；巴氏芽孢八叠球菌与胶胨样类芽孢杆菌(*Paenibacillus mucilaginosus*)的协同矿化体系在岩土防风蚀与结构强化方面成效显著，

**表2 MICP技术在$CO_2$矿化固碳中的应用前景**

Table 2 Application prospects of MICP technology in $CO_2$ mineralization and carbon sequestration

| Microbial categories | Application prospects | Application effect | Key conditions | References |
|---|---|---|---|---|
| Recombinant engineered *Bacillus subtilis* expressing carbonic anhydrase | Lithological improvement and efficient carbon sequestration | $CO_2$ concentration decreased from 3 800 to 820 mg/L. Synergistic action with *Sporosarcina pasteurii* increased mineral production by 1.5 times while preserving $CO_2$ sequestration capacity | An aqueous solution system was adopted with initial $CO_2$ concentration of 3 800 mg/L and mineralization pH of 8.0. No cycle number was set, and no long-term leaching test was performed | [24] |
| *Sporosarcina pasteurii* | Lithological improvement | It readily forms stable calcite crystals accounting for 73%. Synergy with engineered bacteria significantly increases crystal size, raising mineral yield from 0.2 to 0.3 g | | |
| *Bacillus licheniformis* | | Porosity decreased from 12.61% to 10.75%, and core mass increased by 0.23%. Meanwhile, secreted EPS forms biofilms to reduce $CO_2$ leakage | A static sandstone-supercritical $CO_2$-brine aqueous system was used, with initial pH 7.2, salinity 10 000 mg/L, no dynamic circulation, and reaction period of 60 days | [67] |
| *Sporosarcina pasteurii* | | At 1.0 mol/L $Ca^{2+}$, precipitate mass increased from 0.5 to 3.5 g. Samples with calcium acetate rapidly sealed cracks within 48 h, achieving 87.43% calcium removal | The experiment was conducted in an aqueous system with initial $Ca^{2+}$ concentration of 1.0 mol/L and pH 7.5−9.5 (optimum 9.5), without circulation or long-term leaching tests | [68] |
| *Staphylococcus xylosus* | Lithological improvement | Porosity decreased by 6.12%, cohesion increased by 39.5%. Shear strength rose by 22.1%, 13.5%, and 8.3% under confining pressures of 100, 200, and 300 kPa, respectively | A high-salt silty soil column was employed with 0.5 mol/L urea and 0.5 mol/L $CaCl_2$ solution at pH 7.58, 8%−12% soluble salt, and high ionic strength. Single grouting was used without circulation or long-term leaching | [69] |
| *Sporosarcina pasteurii*+ *Paenibacillus mucilaginosus* | | $CaCO_3$ precipitation rate was 60%, wind erosion efficiency reached 95% at 10 m/s wind speed, with dense pores and stable structure | Aqueous system, initial pH 8.0, cementing solution of 0.6 mol/L urea and 0.6 mol/L $CaCl_2$ (ionic strength provided by 0.6 mol/L $Ca^{2+}$ and urea). No fixed cycles, only 5 min short-term simulated rainfall, no long-term leaching | [70] |
| *Priestia megaterium* | Efficient carbon sequestration | It reduced initial 3 800 to 699 mg/L $CO_2$. The BmCAb1/BmCAb2 carbonic anhydrases encoded by this strain exhibited high $CO_2$ hydrolysis activity in engineered bacteria | Aqueous system with initial $CO_2$ concentration of 3 800 mg/L and mineralization pH 8.0. No cycles and no long-term leaching tests | [24] |
| *Paenibacillus mucilaginosus* | | At 0.3% bacterial dosage, $CO_2$ uptake increased from 0.18 to 0.34 kg/m$^2$; 1 g of bacteria absorbed 567 g $CO_2$ | A 1 g/kg *Paenibacillus mucilaginosus* solution was prepared in an aqueous system and added to shotcrete at 0.3% mass fraction. No data on pH, ionic strength, cycles, or long-term leaching was reported | [71] |
| *Chlorella vulgaris* | | The measured maximum $CO_2$ fixation rate was 97.73 mg/(L·d), with total carbon capture up to 1 943.37 mg/L | Modified artificial seawater system with 60 mmol/L $NaHCO_3$ and 18 mmol/L $CaCl_2$. pH rose to 9.5−9.8 during culture. Initial algal concentration, cycles and long-term leaching were not reported | [72] |

可实现 60% 的 $CaCO_3$ 沉淀率，对 10 m/s 风速的风蚀抑制效率达 95%，形成的矿化层致密稳定，兼具抗雨水侵蚀、低蒸发损耗及高抗压强度特性，然而该体系目前仅在实验室水溶液环境中完成验证，未开展原位长期应用测试，大规模现场推广时的微生物活性维持、矿化效率稳定性等核心技术仍待深入研究[70]；巴氏芽孢八叠球菌介导的矿化技术在水泥基材料裂缝修复中展现出快速响应能力，当 $Ca^{2+}$ 浓度为 1.0 mol/L 时，沉淀质量较对照组提升 6 倍，钙去除率达 87.43%，可在 48 h 内快速闭合裂缝，且通过硫铝酸盐水泥包裹细菌可将修复反应精准限定在裂缝区域，但 $Ca^{2+}$ 浓度超过 1.0 mol/L 时会显著抑制细菌代谢活性，生成的方解石晶体与裂缝面缺乏有效黏结力，无法恢复材料力学性能，且未开展长期稳定性验证实验[68]。

在高效固碳领域，多菌株协同体系与特定微生物介导的矿化技术已展现出显著的固碳应用潜力。巴氏芽孢八叠球菌与工程化改造的枯草芽孢杆菌可协同将初始浓度 3 800 mg/L 的 $CO_2$ 降至 820 mg/L，并将矿物沉淀产量提升 1.5 倍，具备高效 $CO_2$ 封存能力，然而该体系仅在 pH 8.0 的水溶液体系中表现最优，对复杂岩土环境的兼容性及长期封存稳定性尚未得到充分验证[24]；普通小球藻(*Chlorella vulgaris*)介导的矿化体系表现出优异的固碳效率，实测最高 $CO_2$ 固定率达(97.73±1.17) mg/(L·d)，总碳捕获量最高为(1 943.37±12.53) mg/L，该体系可避免传统 MICP 技术的氨副产物问题，且生成的文石与方解石晶体结构更利于碳的长期封存，但高浓度 $NaHCO_3$ 或 $CaCl_2$ 会显著抑制微藻生长活性，且现有研究仅在受控改良人工海水体系中开展，其在自然海水环境中的适应性及规模化应用的成本控制仍需进一步研究[72]；胶胨样类芽孢杆菌在喷射混凝土固碳领域展现出协同优化效应，当细菌添加量为 0.3% 时混凝土 $CO_2$ 吸收量从 0.18 kg/m$^2$ 提升至 0.34 kg/m$^2$，同时 14 d 单轴抗压强度可达 25.19 MPa，实现了碳减排与力学性能提升的双重目标，但高细菌添加量会降低混凝土早期强度，且硝酸钙虽可提升早期强度却会抑制碳化反应进程，该体系在实际工程中的长期应用效果及最优添加剂量仍需系统验证[71]。

整体而言，MICP 技术在矿化固碳领域已展现出显著的应用前景，功能菌株及其协同体系表现出差异化的 $CO_2$ 矿化固定潜力。少数极端环境适配菌株可耐受高温高压、高 $CO_2$ 分压地层条件，通过孔隙封堵与岩性强化实现地质尺度碳封存，而微藻及工程菌体系能突破传统固碳效率瓶颈，部分技术还可规避氨副产物风险，为地质封存、海洋固碳、混凝土固碳等场景提供了绿色低成本的技术路径。然而，当前研究仍以实验室可控水溶液体系为主，原位动态地层环境模拟与长期封存效应数据严重不足，多数菌株对极端环境适配性仍较差，工程化应用中微生物的活性维持、矿化产物的长期稳定性等缺乏系统验证，部分技术还存在高浓度抑制剂限制、副产物环境污染等问题。MICP 矿化固碳技术的研究成熟度仍较低，距离规模化工程落地仍存在显著差距。

## 3.2 MICP 技术在重金属污染土壤修复中的应用

在重金属污染修复领域，MICP 技术通过微生物代谢催化生成碳酸盐矿物实现重金属固定。针对不同污染介质、重金属类型及场地条件，该技术展现出定制化修复潜力：单一功能菌株凭借对特定重金属的定向转化能力，在精准修复场景中表现突出；复合修复体系则通过多机制协同作用适配复杂污染环境的综合修复需求，为重金属污染土壤、尾矿及矿山废弃物的绿色修复提供了新路径(表 3)。

**表3 MICP技术在重金属污染土壤修复中的应用前景**

Table 3 Prospects of MICP technology in the remediation of heavy metal contaminated soils

| Abiotic materials/ microorganisms | Application effect | Key conditions | References |
|---|---|---|---|
| *Sporosarcina pasteurii* | $Sr^{2+}$ is efficiently removed *via* isomorphic substitution into calcite or strontianite, with maximum removal rates of 99.8% and 99.9% under static and shaking conditions, respectively | Aqueous system was used with initial $Sr^{2+}$ 10 mmol/L and pH 7.2; ionic strength, cycles, and long-term leaching design were not reported | [39] |
| | Pb immobilization efficiency in aqueous solution reached 78.6%. In loess surface layers, exchangeable Pb decreased from 9.7%−17.4% to 0.4%−2.3%, while carbonate-bound Pb increased to 65.1%−66.2%. In long-term leaching tests, Pb leaching rate was 90.36% at pH 3.5 in 7 days and 12.96% at pH 5.5 in 27 days | Both aqueous and soil column systems were involved. Initial Pb was 1 mmol/L at pH 6.0 in solution, and 4 000 mg/kg in loess for column tests. Long-term leaching was performed; ionic strength and cycle number were not reported | [73] |
| | $Cd^{2+}$ is immobilized by biosorption and lattice doping. After 12 cementation cycles, leachate $Cd^{2+}$ decreased from 7.695 to 2.356 mg/L. Under optimal conditions, it reached 2.439 mg/L, an 88.39% reduction | Soil column system was adopted with initial Cd 400 mg/kg. Optimal conditions: 30 ℃, $OD_{600}$ 1.5, cementation solution 0.5 mol/L. Ionic strength, cycles and long-term leaching design were not reported | [74] |
| | Optimal pH for *Sporosarcina pasteurii* MICP is 6.0−8.0. Cd immobilization reached 99.9% in solution, with calcite dominating $CaCO_3$ for Cd adsorption. In column/*in situ* systems, exchangeable Cd fell by 41.6%, leached Cd by 45.6%. Soil $CaCO_3$, aggregates, and urease activity increased; carbonatebound Cd rose by 40.1%, reducing Cd mobility and bioavailability while enhancing soil stability | Three systems were included. In aqueous solution, initial Cd was 50−200 mg/L; ionic strength was controlled by 0.4−0.6 mol/L urea-$CaCl_2$ cementation solution, without cycling or long-term leaching. In column/*in situ* systems, initial Cd in loess was 50 mg/kg at pH 7.0−8.0, using 0.4−0.6 mol/L cementation solution without cycling; long-term leaching experiments were conducted to assess Cd leaching risk and soil stability | [75] |
| | The $CaCl_2$ group achieved nearly 100% Pb removal efficiency. $Ca(CH_3COO)_2$ was slightly less efficient but much better than the calcium-free control. High inoculation produced only $PbCO_3$ monolayer, while low inoculation formed mixed $(PbCl)_2CO_3$ and $PbCO_3$. $Ca^{2+}$ enhanced remediation by reducing $Pb^{2+}$ toxicity and improving urea hydrolysis | Aqueous system was used with initial $Pb^{2+}$ up to 50 mmol/L, calcium sources 0.25 mol/L $CaCl_2$ or $Ca(CH_3COO)_2$, urea 0.5 mol/L, initial pH 8.8, and inoculation ratios 1:3 and 1:9. No cycling or long-term leaching was performed | [76] |
| Humic acid (HA) | Increased maximum Cd adsorption capacity of $CaCO_3$ from 120.1 to 184.4 mmol/g; relieved Cd inhibition on urease, with Cd fixation rate rising from 3.7% to 19.1% at 96 h; suppressed calcite formation and promoted metastable aragonite | In aqueous solution at pH 7.0, initial $Cd^{2+}$ concentration 10−500 mg/L, HA 0.5−2.0 mg/L, $Ca^{2+}$ 1−5 mmol/L, ionic strength 1 mmol/L ($CaCl_2$ or $CdCl_2$). No cycles or long-term leaching | [77] |

(待续)

(续表3)

| Abiotic materials/ microorganisms | Application effect | Key conditions | References |
|---|---|---|---|
| Urease-producing carbonate-mineralizing functional bacterial community | Under optimal parameters, leachable Cd decreased by 80.7% after 7 mineralization cycles. Cd coprecipitated with Ca as $(Ca_{0.67},Cd_{0.33})CO_3$ and calcite, with tailings encapsulated to inhibit Cd release | In the soil column system, the initial leachable Cd concentration in tailings was 3.11 g/L, the optimal conditions were $1\times10^8$ CFU/mL bacteria, 3 h retention time, 0.25 mol/L mineralization solution and 1.5 mL/min flow rate, and 7 mineralization cycles were performed without long-term leaching | [78] |
| Urea-decomposing community dominated by *Sporosarcina* | Leachable Cd, Pb, and Zn in most samples decreased by at least one order of magnitude. The pH of acidic effluent rose from 1.2−1.3 to 5.7−8.5, and reduced slag permeability enhanced heavy metal immobilization | This technique used a soil column system. Acidic waste samples with initial pH 1.2−1.3 were treated with 15 applications of MICP reagents and *Sporosarcina* urea-decomposing bacteria. Long-term leaching was evaluated by humidity cell method, with no mention of mineralization solution ionic strength | [79] |
| Urea-acclimatized soil microorganisms | MICP-treated soil had a maximum $Cd^{2+}$ adsorption capacity of 8.92 mg/g. The Thomas model for column tests fitted 6.11 mg/g. Fractionation showed exchangeable Cd decreased by 21% and carbonatebound Cd increased by 22% | The technology used a soil column system with initial $Cd^{2+}$ concentration 10 mg/L, pH 6.5, column test breakthrough volume 49 L; ionic strength, cycle times and long-term leaching evaluation were not mentioned | [80] |

其中，巴氏芽孢八叠球菌作为核心模式菌株，针对不同重金属展现出定向修复能力，但也存在诸多场景适配局限。例如，在黄土 Pb 污染修复中，该菌株在弱酸性条件下可实现 78.6% 的 Pb 固定率，将易迁移的交换态 Pb 占比从 9.7%−17.4% 降至 0.4%−2.3%，碳酸盐结合态 Pb 占比提升至 65.1%−66.2%，兼具环境友好性与原位潜力；然而，该菌株对强酸性环境耐受性差，pH 3.5 时 Pb 沉淀 7 d 的浸出率高达 90.36%，且尿素水解产生的 $NH_4^+$易引发氮污染风险[73]。在 Sr 污染修复中，该菌株通过同晶替代机制将 $Sr^{2+}$稳定嵌入方解石或锶白云石晶格，静态与摇动条件下 $Sr^{2+}$最大去除率分别达 99.8% 和 99.9%，兼具高效固锶与固碳双重价值；但该体系存在初期优先沉淀 $Ca^{2+}$的离子选择性，沉淀矿物的长期稳定性及现场规模化应用潜力仍需系统验证[39]。在 Cd 污染尾矿修复中，该菌株通过生物吸附、晶格掺杂等多重机制钝化 $Cd^{2+}$，12 轮胶结后 $Cd^{2+}$浸出浓度从 7.695 mg/L 降至 2.356 mg/L，最优条件下浸出浓度较未处理降低 88.39%；但多次胶结后尾矿孔隙易被碳酸钙充填，导致后续修复效果提升有限，且高浓度胶结液会促进 Cd 的释放并抑制脲酶活性，当温度偏离 30 ℃时还会降低修复效率，此外细菌与土著微生物的竞争也可能影响长期稳定性[74]。在镉污染黄土原位修复中，该菌株在水溶液体系下对 Cd 的固定率最高达 99.9%，在土柱或原位体系下修复后可交换态 Cd 含量降低 41.6%，Cd 浸出浓度降低 45.6%；同时显著提升土体 $CaCO_3$ 含量、水稳定团聚体占比与脲酶活性，有效降低 Cd 的生物有效性与迁移风险，同步抑制水土流失，但该技术的应用受镉浓度、环境含氧量、胶结液浓度等条件限制，高浓度的污染物或胶结液会抑制细菌活性，副产物氨氮存在潜在环境风险，且菌株对深部低氧环境的适配性较差[75]。在高浓度的 Pb 污染修复中，最优接种比

例与钙源类型可实现接近 100% 的 $Pb^{2+}$修复效率；但低接种比例下修复效率随 $Pb^{2+}$浓度升高显著下降，且该研究仅在受控水溶液体系开展，缺乏复杂实际污染环境的长期应用验证[76]。

复合修复体系通过微生物与材料、土著菌群的协同作用增强修复效能，但也存在诸多瓶颈。腐殖酸-微生物复合体系可缓解 Cd 对脲酶的抑制，使 $CaCO_3$ 对 Cd 的最大吸附容量从 120.1 mmol/g 增至 184.4 mmol/g，96 h 内 Cd 固定率由 3.7% 提升至 19.1%，同时促进亚稳态文石形成以增强吸附能力；但该技术仅在实验室水溶液体系验证效果，未开展原位土壤应用，实际污染场地的长期稳定性与适配性仍待验证[77]。产脲酶碳酸盐矿化功能细菌群落经 7 个矿化循环后可将尾矿中毒性特征浸出程序 (toxicity characteristic leaching procedure, TCLP) 的可浸出 Cd 浓度降低 80.7%，形成 $(Ca_{0.67},Cd_{0.33})CO_3$ 和方解石共沉淀包裹尾矿颗粒，减少硫暴露抑制 Cd 释放；但该技术仅在实验室土柱体系验证，矿化效率受细菌浓度、矿化液参数等影响显著，原位应用的长期稳定性与大规模适配性仍需进一步研究[78]。土著芽孢八叠球菌属(*Sporosarcina*)尿素分解细菌群落可使酸性矿山废弃物中 Cd、Pb、Zn 浸出浓度降低至少 1 个数量级，同时将流出液 pH 从 1.2−1.3 升至 5.7−8.5，增强重金属固定能力；但存在部分样品 As 浸出浓度升高、Cu 固定效果不稳定的问题，且碳酸钙涂层分布不均，细菌在酸性环境中易失活，制约长期修复效能[79]。尿素驯化土壤微生物群落可将土壤对 $Cd^{2+}$的最大吸附容量提升至 8.92 mg/g，使可交换态 Cd 占比降低 21%、碳酸盐结合态 Cd 占比增加 22%，经济高效且适合原位修复；但当前 $Cd^{2+}$固定效率仍有优化空间，当前尿素分解产物的生态效应尚不明确，且微生物活性易受土壤 pH 等环境因素制约[80]。

总体而言，MICP 技术在重金属污染修复领域已构建单一菌株精准修复和复合体系协同修复 2 类技术体系，其中单一核心模式菌株对 Pb、Cd、Sr 等特定重金属展现出较强的定向转化与固定能力，可精准适配点源污染，具备低成本原位修复的落地潜力；而复合体系通过微生物-材料、土著菌群协同作用适配于较为复杂的污染修复需求，突破了传统单一菌株的应用场景限制，展现出面向工业场地、农田、矿区等多元污染场景的规模化推广潜力。目前，MICP 技术在重金属污染修复领域已从实验室水溶液拓展至土柱模拟及原位修复场景，为污染土壤、尾矿及矿山废弃物的绿色修复提供了一定的解决方案。然而，当前技术仍存在多方面共性局限，原位应用的长期稳定性与场地适配性仍待验证，复杂污染场景下多重金属协同修复机制、矿化产物长期固持效应等基础研究仍需深化。在工程化应用层面，外源菌株与土著微生物竞争、修复效率随时间衰减等问题尚未得到系统解决，同时相应适配规模化推广的低成本工艺与标准体系尚未完全建立，MICP 技术在重金属固定领域距离大范围工程化落地仍有一定距离。

# 4 多维视角下的 MICP 技术评估

## 4.1 MICP 技术的成本与长期性表现

MICP 技术的成本与长期性表现均具有显著的场景依赖性，成本受胶结强度、施工工艺、环境条件控制，长期耐久性则受侵蚀环境、胶结物稳定性主导，二者均需结合具体应用场景进行针对性分析与评估。

在成本水平方面，MICP 技术的总成本由材料、设备、施工、监测维护等共同构成，其中材料成本仅为基础组成部分。针对常规场地工程加固场景，Ivanov 等[81]报道的材料成本仅为 0.5−9.0 USD/$m^3$，而 Dejong 等[82]综述表明，现场综合成本约为 75−500 USD/$m^3$。二者差异主要源于工程尺度下设备投入、人员培训、多次灌注及监测维护等隐性成本占比较高，导致实际

总成本显著高于材料成本。对于一些特定场景，成本可能显著降低或提升。例如，在沙漠环境中，该场景对胶结强度要求较低，仅需形成表层稳定结皮，无需深层灌注，因此成本显著低于常规场地工程加固场景，在最优处理方案下成本为 25–30 USD/$m^2$；该成本仍有进一步优化空间，通过本地材料替代、规模化细菌培养及精准复喷策略的协同优化，可降低 30%–40% 的原材料成本、20% 的菌剂制备成本和 50% 的无效材料消耗[83]；而在海洋环境中，尽管目前尚缺乏公开的直接成本数据，但 Yang 等[84]通过微流控实验研究表明，在高盐度、低氧、低温的海洋综合环境下，MICP 的化学转化效率仅为理想条件的 20.6%，这意味着在海洋工程中应用 MICP 技术时，为达到与陆上相同的加固效果，需大幅增加材料投入与施工周期，综合成本也随之显著上升。综上可见，MICP 技术的经济性评估须紧密结合具体应用场景的技术需求与施工条件，单一的成本数值比较缺乏实际指导意义。

在长期性表现方面，MICP 胶结体的长期耐久性高度依赖环境条件。塔克拉玛干沙漠现场试验显示，30 d 监测期内单次处理形成的土壤结皮表现出显著稳定性，风蚀量减少 95%，Tao 等[83]基于不可逆矿化过程的理论推断，预测该结皮在该干旱地区可维持 3–5 年。在南海珊瑚砂现场试验中，30 d 后 MICP 加固样品无侧限抗压强度保持率仅为 35.19%，且伴随明显性能衰减；在这项研究中还发现，MICP 处理土壤对温度循环抗性最强，其次为干湿循环、温度-干湿耦合循环及盐雾-干燥循环，而对现场海洋环境侵蚀抗性最弱[85]。因此，MICP 技术的长期性能表现高度依赖环境条件，需根据具体应用场景因地制宜地选择适宜的处理策略，以最大化其耐久性。

需要指出的是，目前 MICP 技术在工程尺度下的长期监测数据极为匮乏。现有最大规模现场试验为荷兰 1 000 $m^3$ 天然气管道加固项目，处理深度为 3–20 m，分 2 次注入共 300–600 $m^3$ 含尿素和氯化钙的反应液，成功实现水平定向钻进，但缺乏后续长期性能追踪报道[86]。此外，多数现场试验监测期较短，难以评估数年设计寿命下的性能演化。因此，亟需开展长期现场监测与加速老化试验，以科学评估 MICP 处理在数年乃至数十年时间尺度上的可持续性。

## 4.2 MICP 技术的环境与生态风险

氨氮释放与氮污染风险是脲酶途径的核心环境问题。尿素水解产生的大量铵根离子可能随渗流进入水体，导致水体富营养化，对人体健康和水生生态系统构成威胁。针对氨氮问题，磷酸铵镁(magnesium ammonium phosphate, MAP)沉淀法提供了一种资源化解决方案。通过添加镁离子和磷酸根，可将氨氮转化为难溶性 MAP 晶体($MgNH_4PO_4·6H_2O$)，实现氨氮固定与缓释肥料回收的协同[87]；此外，也可以利用异养硝化菌株实现氨氮与磷酸盐同步去除，有效控制氨氮释放与氮污染风险[88]。

盐分累积与土壤理化性质改变是另一潜在风险。MICP 技术通常以氯化钙作为外源钙源，并伴随尿素等营养盐的大量投加，高浓度 $Cl^-$与未反应离子的残留会显著改变土壤离子组成与渗透压环境，导致土壤次生盐渍化并抑制植物根系发育[89]。针对盐渍化风险，可通过优化钙源种类与投加策略加以缓解，例如采用富钙硅酸盐岩石等天然低氯替代钙源，既可减少 $Cl^-$输入，又能通过矿物风化过程实现大气 $CO_2$ 的协同封存，降低环境碳成本[90]；同时，基于盐渍土改良的 MICP 技术可通过形成微生物矿化阻隔层显著抑制水盐迁移，减少盐分表聚与冻胀变形，为干旱半干旱地区盐渍化土壤的生态修复提供新途径[91]。

MICP 技术中外源菌株的生态影响同样值得关注，通过引入外源菌株可能对本地生态系统产生复杂影响。一方面，外源菌株可能通过竞争性排斥或改变微生物群落结构，影响本地微生物的生物多样性和功能，尤其在营养循环或

共生关系中可能引发失衡[92]；另一方面，外源菌株在极端环境中的适应性不足可能导致定植困难、代谢活性降低及修复效率下降，甚至因副产物积累而加剧环境风险：研究表明腐殖酸会抑制脲酶活性并阻碍方解石结晶，导致无定形沉淀物形成，而外源菌株受此负面影响明显大于土著菌株，进一步凸显其在复杂土壤环境中的生态兼容性缺陷[93]。相比之下，土著菌株因其对本地环境条件的先天适应性而表现出显著优势，例如从石灰石矿分离的红球菌属菌株 L6 和 L8 不仅能在 5 ℃低温下维持脲酶活性和生长，且能有效诱导方解石沉淀并密封岩石裂隙，展现出良好的生态兼容性与生物修复潜力[53]。因此，优先筛选和应用土著功能菌株，可有效规避外源菌株的生态风险，提升 MICP 技术修复中的环境安全性与可持续性。

## 5 MICP 技术应用的问题挑战与解决方案

MICP 技术尽管目前已得到广泛研究，但距离工程化应用仍面临诸多挑战，主要体现在长期性能的不确定性与技术系统集成不足 2 个方面。如 4.1 节所述，工程尺度下长期监测数据的不足与设计寿命验证的缺失，导致工程可靠性评价缺乏科学依据；现有研究多停留于单一环节的局部优化，尚未形成涵盖菌株改造、环境适配、工艺调控及生态安全等多方面的系统性整合方案，难以支撑标准化施工体系的建立与规模化推广。同时 MICP 技术仍存在诸多技术瓶颈，主要涵盖以下 3 个方面。

(1) 菌株多样性匮乏与耐受性不足。MICP 协同应用的核心限制在于功能菌株的生态位窄与功能单一，现有优势菌株如巴氏芽孢八叠球菌、枯草芽孢杆菌等对强酸、高重金属浓度、高盐等极端条件敏感，活性易受抑制，难以适应复杂的场景需求。

(2) 外源菌株引入的矛盾冲突。外源功能菌株难以适应自然环境的复杂性。现场 pH 波动、复合污染毒性、营养匮乏会降低菌株活性；土著菌群的竞争排斥易导致外源菌株定植失败；大量引入外源菌株还可能破坏土著菌群结构，干扰土壤酶活性与养分循环，引发潜在生态风险[92]。

(3) 协同机制代谢路径耦合矛盾。对于同时实现固碳与重金属固定存在一定的代谢路径矛盾，重金属离子可结合脲酶/CA 的 $Ni^{2+}$、$Zn^{2+}$等活性中心进而抑制酶活性，降低固碳效率；而在固碳时，$Ca^{2+}$与重金属离子发生 $CO_3^{2-}$的底物竞争，以及固碳最优 pH 值与重金属固定，如 Cd 共沉淀的最优 pH 值存在一定的差异，以上问题均在一定程度上制约了固碳与重金属固定协同作用，如何对两者进行平衡取舍是实现固碳与重金属协同固定的关键[94-95]。

针对上述技术瓶颈，近期研究人员尝试从不同路径寻求突破。如 Wang 等[65]通过添加镁多肽，在增强了对 Pb 的修复效果的同时，缓解了 MICP 对微生物数量和多样性的损害；而 Chen 等[96]通过引入植物源脲酶中的剑豆脲酶，能够防止脲酶失活并吸附 $Zn^{2+}$；Jia 等[77]则通过施加腐殖酸稳定脲酶的二级结构，不仅保留了酶活性还增强了对镉的固定效果。

然而，这些技术手段并不能完全解决上述难点。整体而言，现有技术多聚焦于单一问题的局部优化，缺乏对上述瓶颈的系统性整合解决方案，这也是未来研究需要突破的核心方向。同时，目前有关 MICP 技术协同实现固碳与重金属固定的研究十分匮乏，多数研究仍停留在单一功能层面的应用。通过对 MICP 技术各应用场景的 pH 适宜区间进行系统梳理(表 4)，可以发现重金属固定的最优 pH 多集中于 6.0−7.2 的弱酸性至中性范围，岩性改良多分布于 7.0−8.0 的中性至弱碱性区间，而高效固碳整体位于 8.0 以上的碱性区间。前两者的 pH 窗口呈现梯度差异，这为协同工艺的时序性构建提供了调控空间。基于此，本文提出一种“重金属固定-矿化

固碳-长期监测”的 3 阶段协同工艺流程假设(图 4)。

第 1 阶段：重金属固定主导期。首先通过驯化改造土著菌株，筛选兼具脲酶活性及重金属耐受性的多功能菌株以适应目标污染环境；随后控制体系 pH 于偏酸性环境中，优先完成重金属的吸附与晶格固定。此阶段以脲酶介导的 MICP 为主，并辅以腐殖酸等外源调控因子，以稳定脲酶二级结构保留酶活性，同时促进亚稳态文石形成以增强重金属吸附能力。

**表4　MICP技术适用pH区间**

Table 4　Applicable pH range for MICP technology

| Classification of MICP technology | Applicable pH | References |
|---|---|---|
| Heavy metal immobilization | Pb aqueous system (pH 6.0) | [73] |
| | Cd soil column system (pH 6.5) | [80] |
| | HA-Cd aqueous system (pH 7.0) | [77] |
| | Sr aqueous system (pH 7.2) | [39] |
| Lithological improvement | Sandstone-$CO_2$ system (pH 7.2) | [67] |
| | Saline silty soil column system (pH 7.58) | [68] |
| | Calcium chloride aqueous system (pH 8.0) | [69] |
| | Mineralized aqueous system (pH 7.5−9.5) | [70] |
| Efficient carbon sequestration | Engineered *Bacillus subtilis* aqueous system (carbonic anhydrase, pH 8.0) | [24] |
| | Peak urea-driven $CO_2$ capture in aqueous solution (pH 8.9) | [56] |
| | Chlorella aqueous system (pH 9.5−9.8) | [71] |

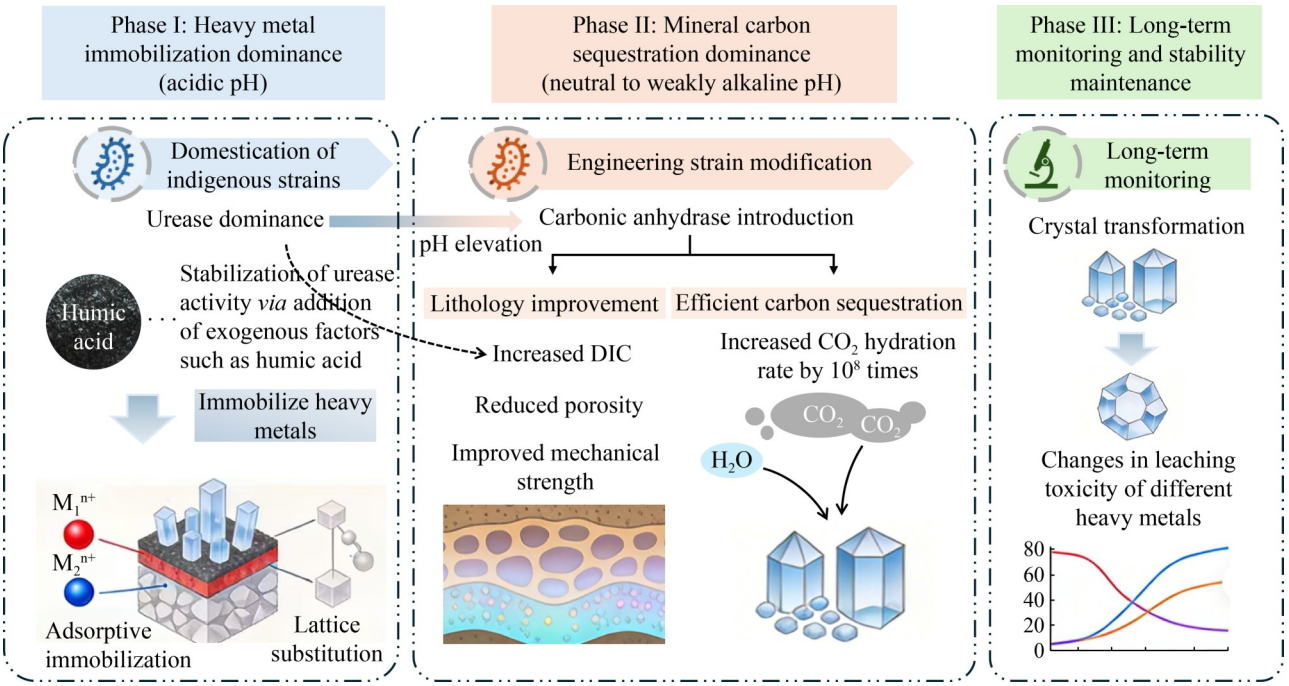



**图4　“重金属固定-矿化固碳-长期监测”的3阶段协同工艺流程**

Figure 4　Three-stage synergistic process flowchart of “heavy metal immobilization-mineralization carbon sequestration-long-term monitoring”.

第 2 阶段：矿化固碳期。第 1 阶段脲酶水解尿素产生的氨持续提升体系 pH，同时释放的 $CO_2$ 为 CA 路径提供了直接底物，形成“产碱-供碳”的协同效应。仿照工程菌改造策略，激活或引入 CA 活性，驱动 $CO_2$ 水合-矿化反应，实现高效固碳；生成的碳酸钙沉淀填充土壤孔隙，同步完成岩性改良与碳封存场地稳定性强化。

第 3 阶段：长期稳定维持期。通过周期性监测矿化产物的晶型转化及重金属浸出毒性，评估固定化产物的长期稳定性。

该流程的核心优势在于利用脲酶路径的“产碱-供碳”协同效应实现 pH 自驱动调控与底物衔接，避免了外源碱的大量投加；通过时序性代谢路径切换，化解重金属固定与矿化固碳的 pH 需求矛盾；土著菌株驯化与功能酶协同则提升了体系的生态兼容性与环境适应性。该设计针对性回应了 MICP 技术规模化应用中的菌株极端环境耐受性不足、外源-土著生态冲突及代谢路径耦合障碍 3 大技术瓶颈。将各技术手段集成，为 MICP 技术的工程化应用提供了新的思路。该假设在现阶段已获得一定的实验验证。Li 等[97]从酸性矿山废水(acid mine drainage, AMD)淤泥中驯化获得耐酸脲酶细菌群落 UBC，该菌群可耐受 pH 为 3.0−4.0 的酸性环境，在合成 AMD 溶液中，反应体系 pH 由初始 3.6 自然升至 8.4，并于 32 h 内实现了超过 99.9% 稀土元素(rare earth elements, REEs)的去除，同时伴随其他金属离子的协同沉淀，该研究证实了脲酶驱动 pH 自调控与重金属同步去除的可行性；Li 等[98]研究表明，胶胨样类芽孢杆菌来源的 CA 在 pH 8.0−11.0 的偏碱性环境中表现出更高催化活性，这与本研究第 2 阶段矿化固碳期的 pH 适配需求相吻合；而 Gilmour 等[24]通过工程改造枯草芽孢杆菌重组表达巨大普里斯特氏菌(*Priestia megaterium*)来源的 CA，在密闭反应体系中实现了 $CO_2$ 浓度从 3 800 mg/L 至 820 mg/L 的显著降低，并生成稳定的方解石型 $CaCO_3$ 沉淀，该研究进一步证实，将脲酶模式菌与 CA 活性联用是可行的；此外，Ali 等[99]采用 X 射线衍射(X-ray diffraction, XRD)等多尺度表征手段对 MICP 产物进行晶型分析，证实微生物可诱导 $Mn^{2+}$、$Sr^{2+}$ 等形成菱锰矿、菱锶矿等稳定碳酸盐晶体，通过晶格固持实现重金属的长期稳定化，为第 3 阶段长期稳定维持期的监测评估提供了直接依据。需要说明的是，尽管目前已有部分实验证据支撑该工艺假设的核心机制，但各阶段转化的精确临界点、微生物群落的动态演替规律及其长期稳定性仍需通过系统性实验进一步验证。

## 6 总结与展望

本文系统梳理了 MICP 技术的代谢路径、影响因素与应用场景，阐明了其作为减污固碳协同治理技术的核心机制：固碳与重金属固定并非 2 个孤立的过程，而是微生物代谢驱动的碳酸盐矿化反应中相互耦合、动态协同的 2 个维度。脲酶水解尿素产生氨和 $CO_2$，既提升 pH 促进碳酸盐沉淀，又为 CA 路径提供底物，这为 MICP 技术的协同治理创造了生化基础。随着反应的进行，pH 的自然升高，则为分阶段调控工艺参数提供了理论依据。

针对当前 MICP 技术面临的菌株极端环境耐受性不足、外源-土著生态冲突及代谢路径耦合障碍 3 大瓶颈，本文基于现有研究进展创新性地提出“重金属固定-矿化固碳-长期监测” 3 阶段协同工艺流程假设。通过时序性切换代谢路径，利用脲酶路径的“产碱-供碳”效应实现 pH 自驱动调控，化解重金属固定与矿化固碳的 pH 需求矛盾，并通过土著菌株驯化与功能酶协同提升体系的生态兼容性与环境适应性。该设计将现有单一技术整合为系统性解决方案，为 MICP 技术的工程化应用提供了新的思路。

未来研究需在生物、方法、应用 3 个层面实现突破。在生物层面，应从单一菌株筛选转向功能菌群设计，构建代谢互补的微生物体系。重点开展功能菌株改造，通过基因工程手段强化土著菌株的脲酶活性、重金属耐受性及 CA 表

达水平等，同时解析外源-土著微生物互作机制，建立菌群调控策略，尽可能避免生态冲突；在方法层面，应从静态条件优化转向动态过程模拟与机理阐释，深入揭示其转化机理，建立含pH、酶活性、$CO_2$固定速率等多参数的动力学模型，以期实现对各阶段转化临界点的精准预测与调控。在应用层面，应从末端治理转向全流程系统调控，开展从实验室微宇宙到现场中试尺度的逐级验证，以实测数据支撑理论设计向工程应用转化，构建涵盖晶型演变监测、重金属浸出毒性评价、碳封存效率量化的长期稳定性评估体系，为实际应用提供可靠依据。

在全球碳中和与土壤安全利用的双重背景下，MICP技术为污染土壤修复提供了一条值得探索的路径。其核心意义在于以同一生物过程同步实现减污与固碳，这种协同治理思路本身较单一技术优化更具指导价值：从分治走向协同，从干预走向顺应。当然，这一目标的实现尚需基础研究与应用实践的长期磨合，也有赖于环境科学与工程技术的深度交叉，未来仍有广阔的研究空间与应用前景。

## 作者贡献声明

罗正宇：研究指导、论文修改；何煌：资料收集及分析、论文撰写和修改、图表绘制；韩洪瑞：图表建议、论文建议和修改；倪文、范翼寻：资料收集、论文修改；郭玉：资料收集；黄昕昳、刘振宇：资料收集；童晋、陈志：研究指导讨论。

## 作者利益冲突公开声明

作者声明不存在任何可能会影响本文所报告工作的已知经济利益或个人关系。

## 参考文献

[1] Intergovernmental Panel on Climate Change. Climate Change 2022—Mitigation of Climate Change: Working Group III Contribution to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change[M]. Cambridge: Cambridge University Press, 2023.

[2] Huang G, Xu ZH, Qu X, Cao JJ, Long SM, Yang K, Hou HY, Wang Y, Ma XF. Critical climate issues toward carbon neutrality targets[J]. Fundamental Research, 2022, 2(3): 396-400.

[3] Saleem M, Pierce D, Wang YQ, Sens DA, Somji S, Garrett SH. Heavy metal(oid)s contamination and potential ecological risk assessment in agricultural soils[J]. Journal of Xenobiotics, 2024, 14(2): 634-650.

[4] Xia F, Zhao ZF, Niu X, Wang ZF. Integrated pollution analysis, pollution area identification and source apportionment of heavy metal contamination in agricultural soil[J]. Journal of Hazardous Materials, 2024, 465: 133215.

[5] Yao C, Yang YD, Li CX, Shen ZJ, Li JQ, Mei N, Luo CZ, Wang YM, Zhang C, Wang DY. Heavy metal pollution in agricultural soils from surrounding industries with low emissions: assessing contamination levels and sources[J]. Science of the Total Environment, 2024, 917: 170610.

[6] Liu JH, Cao L, Dou SZ. Trophic transfer, biomagnification and risk assessments of four common heavy metals in the food web of Laizhou Bay, the Bohai Sea[J]. Science of the Total Environment, 2019, 670: 508-522.

[7] Feng D, Meng L, Wen YH, Uwiragiye Y, AbuQamar SF, Okoth N, Zhu QL, Wu ZP, Wu YZ, Müller C, Zhang JB, Elrys AS. Edaphic and climatic factors control the response of nutrient-cycling enzyme activity to common heavy metals in soils[J]. Journal of Hazardous Materials, 2025, 494: 138475.

[8] 孙养存, 尹紫良, 葛菁萍. 土壤中重金属污染物的来源及治理方式[J]. 中国农学通报, 2022, 38(6): 75-79.
Sun YC, Yin ZL, Ge JP. Accumulation of heavy metal pollutants in soil: sources and treatment methods[J]. Chinese Agricultural Science Bulletin, 2022, 38(6): 75-79 (in Chinese).

[9] 董海钰, 胡慧, 梁在旭, 王明远, 崔继元, 冯滨滨, 盛况, 刘芳, 陈伟. 土壤重金属污染精准修复技术的研究进展[J]. 科学通报, 2025, 70(1): 1-14.
Dong HY, Hu H, Liang ZX, Wang MY, Cui JY, Feng BB, Sheng K, Liu F, Chen W. Research progress on precision remediation technologies for heavy metal pollution in soil[J]. Chinese Science Bulletin, 2025, 70(1): 1-14 (in Chinese).

[10] Karamalidis AK, Torres SG, Hakala JA, Shao HB, Cantrell KJ, Carroll S. Trace metal source terms in carbon sequestration environments[J]. Environmental Science & Technology, 2013, 47(1): 322-329.

[11] Psarras P, He JJ, Pilorgé H, McQueen N, Jensen-Fellows A, Kian K, Wilcox J. Cost analysis of carbon capture and sequestration from U.S. natural gas-fired power plants[J]. Environmental Science & Technology, 2020, 54(10): 6272-6280.

[12] Yao Y. A woody biomass burial[J]. Science, 2024, 385(6716): 1417-1418.

[13] Chang JJ, Yang DY, Lu C, Shu ZT, Deng SJ, Tan LW, Wen SQ, Huang K, Duan PC. Application of microbially induced calcium carbonate precipitation (MICP) process in concrete self-healing and environmental restoration to facilitate carbon neutrality: a critical review[J]. Environmental Science and Pollution Research, 2024, 31(26): 38083-38098.

[14] Kim JJ, Lee SS, Fenter P, Myneni SCB, Nikitin V, Peters CA. Carbonate coprecipitation for Cd and Zn treatment and evaluation of heavy metal stability under acidic conditions[J]. Environmental Science & Technology, 2023, 57(8): 3104-3113.
[15] Sujiritha PB, Vikash VL, Ponesakki G, Ayyadurai N, Kamini NR. Microbially induced carbonate precipitation with *Arthrobacter creatinolyticus*: an eco-friendly strategy for mitigation of chromium contamination[J]. Journal of Environmental Management, 2024, 365: 121300.
[16] Drew GH. The action of some denitrifying bacteria in tropical and temperate seas, and the bacterial precipitation of calcium carbonate in the sea[J]. Journal of the Marine Biological Association of the United Kingdom, 1911, 9(2): 142-155.
[17] Boquet E, Boronat A, Ramos-Cormenzana A. Production of calcite (calcium carbonate) crystals by soil bacteria is a general phenomenon[J]. Nature, 1973, 246(5434): 527-529.
[18] Kantzas A, Stehmeier L, Marentette DF, Ferris FG, Jha KN, Maurits FM. A novel method of sand consolidation through bacteriogenic mineral plugging[C]//Annual Technical Meeting. Calgary, Alberta: PETSOC, 1992: PETSOC-92-46.
[19] Bang SS, Galinat JK, Ramakrishnan V. Calcite precipitation induced by polyurethane-immobilized *Bacillus pasteurii*[J]. Enzyme and Microbial Technology, 2001, 28(4/5): 404-409.
[20] Whiffin VS. Microbial $CaCO_3$ precipitation for the production of biocement[D]. Perth: Murdoch University, 2008.
[21] Mitchell AC, Ferris FG. The influence of *Bacillus pasteurii* on the nucleation and growth of calcium carbonate[J]. Geomicrobiology Journal, 2006, 23(3/4): 213-226.
[22] Jonkers HM, Schlangen E. Crack repair by concrete-immobilized bacteria[J]. Journal of Industrial Microbiology and Biotechnology, 2007, 34(12): 771-777.
[23] Okyay TO, Nguyen HN, Castro SL, Rodrigues DF. $CO_2$ sequestration by ureolytic microbial consortia through microbially-induced calcite precipitation[J]. Science of the Total Environment, 2016, 572: 671-680.
[24] Gilmour KA, Ghimire PS, Wright J, Haystead J, Dade-Robertson M, Zhang M, James P. Microbially induced calcium carbonate precipitation through $CO_2$ sequestration *via* an engineered *Bacillus subtilis*[J]. Microbial Cell Factories, 2024, 23: 168.
[25] Achal V, Pan XL. Characterization of urease and carbonic anhydrase producing bacteria and their role in calcite precipitation[J]. Current Microbiology, 2011, 62(3): 894-902.
[26] Mazzei L, Cianci M, Benini S, Ciurli S. The structure of the elusive urease-urea complex unveils the mechanism of a paradigmatic nickel-dependent enzyme[J]. Angewandte Chemie International Edition, 2019, 58(22): 7415-7419.
[27] 刘鹏, 曹源兴, 程钰, 白云波. 碳酸酐酶增强微生物矿化固土效果的试验研究[J]. 岩土力学, 2024, 45(9): 2554-2564.
Liu P, Cao YX, Cheng Y, Bai YB. Experimental study of enhancing the effects of microbial-induced calcite precipitation treated sand using carbonic anhydrase[J]. Rock and Soil Mechanics, 2024, 45(9): 2554-2564 (in Chinese).
[28] Cowan RM, Ge JJ, Qin YJ, McGregor ML, Trachtenberg MC. $CO_2$ capture by means of an enzyme-based reactor[J]. Annals of the New York Academy of Sciences, 2003, 984(1): 453-469.
[29] Duda D, Tu C, Qian MZ, Laipis P, Agbandje-McKenna M, Silverman DN, McKenna R. Structural and kinetic analysis of the chemical rescue of the proton transfer function of carbonic anhydrase II[J]. Biochemistry, 2001, 40(6): 1741-1748.
[30] Elder I, Han SF, Tu C, Steele H, Laipis PJ, Viola RE, Silverman DN. Activation of carbonic anhydrase II by active-site incorporation of histidine analogs[J]. Archives of Biochemistry and Biophysics, 2004, 421(2): 283-289.
[31] Zhang YT, Zhang L, Chen HL, Zhang HM. Selective separation of low concentration $CO_2$ using hydrogel immobilized CA enzyme based hollow fiber membrane reactors[J]. Chemical Engineering Science, 2010, 65(10): 3199-3207.
[32] Okada S, Kurahashi N, Tanida Y. Terahertz spectroscopy for analysis of vaterite-to-calcite crystal phase transition induced by distilled water[J]. PLoS One, 2025, 20(5): e0323421.
[33] Kim S, Park CB. Dopamine-induced mineralization of calcium carbonate vaterite microspheres[J]. Langmuir, 2010, 26(18): 14730-14736.
[34] Dhami NK, Reddy MS, Mukherjee A. Synergistic role of bacterial urease and carbonic anhydrase in carbonate mineralization[J]. Applied Biochemistry and Biotechnology, 2014, 172(5): 2552-2561.
[35] Violette MJ, Hyland E, Burgener L, Ghosh A, Montoya BM, Kleiner M. Meta-omics reveals role of photosynthesis in microbially induced carbonate precipitation at a $CO_2$-rich geyser[J]. ISME Communications, 2024, 4: ycae139.
[36] Nakano A. Effect of sand minerals on microbially induced carbonate precipitation by denitrification[J]. Chemosphere, 2024, 363: 142890.
[37] Phillips AJ, Cunningham AB, Gerlach R, Hiebert R, Hwang C, Lomans BP, Westrich J, Mantilla C, Kirksey J, Esposito R, Spangler L. Fracture sealing with microbially-induced calcium carbonate precipitation: a field study[J]. Environmental Science & Technology, 2016, 50(7): 4111-4117.
[38] Xia SX, Song W. Controls on microbially-induced carbonate precipitation in geologic porous media[J]. Science of the Total Environment, 2024, 957: 177647.
[39] Su ZM, Chen T, Li FC, Li XL, Zhang F, Yi Y. Microbially induced carbonate precipitation as an innovative technology for achieves efficient $Sr^{2+}$ bioremediation[J]. New Biotechnology, 2025, 90: 281-289.
[40] Weiner S, Levi-Kalisman Y, Raz S, Addadi L.

Biologically formed amorphous calcium carbonate[J]. Connective Tissue Research, 2003, 44(1): 214-218.
[41] Rodriguez-Blanco JD, Shaw S, Benning LG. The kinetics and mechanisms of amorphous calcium carbonate (ACC) crystallization to calcite, viavaterite[J]. Nanoscale, 2011, 3(1): 265-271.
[42] Clark SM, Grigorova V, Colas B, Darwish TA, Wood K, Neuefeind J, Jacob DE. The kinetics of aragonite formation from solution *via* amorphous calcium carbonate[J]. Nanomaterials, 2022, 12(23): 4151.
[43] Wang QH, Huang WY, Wang JL, Long F, Fu ZY, Xie JJ, Zou ZY. Stabilization and crystallization mechanism of amorphous calcium carbonate[J]. Journal of Colloid and Interface Science, 2025, 680: 24-35.
[44] 何小丹. 微生物合成球霰石去除Cu(II)/亚甲基蓝复合污染的模拟研究[D]. 南京: 南京师范大学, 2023.
He XD. Simulated study on removal of Cu(II)/methylene blue composite pollution by microbially synthesized vaterite[D]. Nanjing: Nanjing Normal University, 2023 (in Chinese).
[45] Liu RL, Lian B. Immobilisation of Cd(II) on biogenic and abiotic calcium carbonate[J]. Journal of Hazardous Materials, 2019, 378: 120707.
[46] Sasamoto R, Kanda Y, Yamanaka S. Difference in cadmium chemisorption on calcite and vaterite porous particles[J]. Chemosphere, 2022, 297: 134057.
[47] Okyay TO, Rodrigues DF. Biotic and abiotic effects on $CO_2$ sequestration during microbially-induced calcium carbonate precipitation[J]. FEMS Microbiology Ecology, 2015, 91(3): fiv017.
[48] Ohan JA, Saneiyan S, Lee J, Bartlow AW, Ntarlagiannis D, Burns SE, Colwell FS. Microbial and geochemical dynamics of an aquifer stimulated for microbial induced calcite precipitation (MICP)[J]. Frontiers in Microbiology, 2020, 11: 1327.
[49] Li ZM, Li TX. New insights into microbial induced calcium carbonate precipitation using *Saccharomyces cerevisiae*[J]. Frontiers in Microbiology, 2022, 13: 904095.
[50] Meister P, Wiedling J, Lott C, Bach W, Kuhfuß H, Wegener G, Böttcher ME, Deusner C, Lichtschlag A, Bernasconi SM, Weber M. Anaerobic methane oxidation inducing carbonate precipitation at abiogenic methane seeps in the Tuscan Archipelago (Italy)[J]. PLoS One, 2018, 13(12): e0207305.
[51] Su F, Yang YY. Microbially induced carbonate precipitation *via* methanogenesis pathway by a microbial consortium enriched from activated anaerobic sludge[J]. Journal of Applied Microbiology, 2021, 131(1): 236-256.
[52] Hu L, Wang HY, Xu P, Zhang YY. Biomineralization of hypersaline produced water using microbially induced calcite precipitation[J]. Water Research, 2021, 190: 116753.
[53] Shen XL, Hashiba K, Yakata T, Yoshida K, Kobayashi H. Characterization of indigenous bacteria for microbially induced carbonate precipitation in a limestone mine[J]. Microorganisms, 2025, 13(9): 1985.
[54] Jiang Y, Liu S, Cai CS, Dang XF, Yang C, Xing YH, Ma F, Tan SX, Huang QY, Chen WL. Microbially induced carbonate precipitation balances survival and function: driven heavy metal remediation and plant growth promotion by *Enterobacter ludwigii* N15[J]. Journal of Hazardous Materials, 2025, 498: 139970.
[55] 章求才, 郑逸非, 黄梅钟, 沈修康, 伍玲玲, 张志军. pH调控脲酶构象影响微生物诱导碳酸盐沉淀效能及模拟研究[J]. 中国生物化学与分子生物学报, 2025, 41(6): 879-894.
Zhang QC, Zheng YF, Huang MZ, Shen XK, Wu LL, Zhang ZJ. Experimental and computational investigation of pH-dependent urease conformational dynamics and its impact on MICP efficiency[J]. Chinese Journal of Biochemistry and Molecular Biology, 2025, 41(6): 879-894 (in Chinese).
[56] Chen R, Kavala AM, Clarà Saracho A, Marek EJ. Modeling the effect of microbially induced calcium carbonate precipitation (MICP) on $CO_2$ trapping[J]. Environmental Science & Technology, 2025, 59(40): 21486-21500.
[57] Hsieh YE, Yang SY, Liu SL, Wang SW, Wang WL, Tang SL, Yang SH. Microbial community shifts and nitrogen utilization in peritidal microbialites: the role of salinity and pH in microbially induced carbonate precipitation[J]. Microbial Ecology, 2025, 88: 31.
[58] Peng J, Liu ZM. Influence of temperature on microbially induced calcium carbonate precipitation for soil treatment[J]. PLoS One, 2019, 14(6): e0218396.
[59] Zhu XJ, Wang JY, De Belie N, Boon N. Complementing urea hydrolysis and nitrate reduction for improved microbially induced calcium carbonate precipitation[J]. Applied Microbiology and Biotechnology, 2019, 103(21/22): 8825-8838.
[60] Nikseresht F, Landi A, Sayyad G, Ghezelbash GR, Schulin R. Sugarecane molasse and vinasse added as microbial growth substrates increase calcium carbonate content, surface stability and resistance against wind erosion of desert soils[J]. Journal of Environmental Management, 2020, 268: 110639.
[61] Yu WW, Huang XW, Zhou LM, Zhang LD, Zheng XM, Luo WJ. Effects of trehalose and sodium alginate on microbially induced carbonate precipitation[J]. Environmental Research, 2024, 263: 120145.
[62] Zhang WC, Shen L, Xu RY, Dong X, Luo SR, Gu HJ, Qin FJ, Liu HW. Effect of biopolymer chitosan on manganese immobilization improvement by microbial-induced carbonate precipitation[J]. Ecotoxicology and Environmental Safety, 2024, 279: 116496.
[63] Li Y, Zhang ML, Wang XB, Ai S, Meng XH, Liu ZQ, Yang F, Cheng K. Synergistic enhancement of cadmium immobilization and soil fertility through biochar and artificial humic acid-assisted microbial-induced calcium carbonate precipitation[J]. Journal of Hazardous Materials, 2024, 476: 135140.
[64] Wang Z, Su JF, Ali A, Yang WS, Zhang RJ, Li YF, Zhang LF, Li JW. Chitosan and carboxymethyl chitosan mimic biomineralization and promote microbially induced calcium precipitation[J]. Carbohydrate Polymers, 2022, 287: 119335.

[65] Wang ZW, Zhang ZY, Peng J, Zhang YX, Zhou F, Yu JX, Chi R, Xiao CQ. Magnesium polypeptide combined with microbially induced calcite precipitation for remediation of lead contamination in phosphate mining wasteland soil[J]. Environmental Research, 2024, 262: 119945.
[66] Shen YF, Li Q, Pei XJ, Wei RJ, Li YH, Jin B, Yang YX, Zhang WB, Lei NF, Zhang XC. Facilitating the restoration of disturbed meadow soils using modified organic materials in the Qinghai-Xizang Plateau[J]. Land Degradation & Development, 2025, 36(9): 3105-3117.
[67] Wang BK, Yuan YY, Qian CX. Carbonic anhydrase-producing bacteria-mediated sandstone-$CO_2$-brine interaction and its impact on carbon sequestration[J]. Environmental Research, 2025, 286: 122801.
[68] Jia YY, Xue CH, Zhang W. Influence of calcium concentration and pH value on the efficacy of MICP in self-healing cementitious materials[J]. Case Studies in Construction Materials, 2025, 23: e05507.
[69] Yuan Y, Zhao Y, Yang X, Zhang W, Han LL, Xie DF, Hou FX. Effect of microbially induced carbonate precipitation (MICP) in the highly saline silty soil of the cold plateau area of the Qinghai-Xizang Plateau[J]. KSCE Journal of Civil Engineering, 2022, 26(11): 4407-4418.
[70] Zhou G, Zhao ZY, Zhang GQ, Gao X, Wang MJ, Wei B, Chen X, Li JH, Li L. Enhanced mineralization and dust suppression mechanism of *Bacillus pasteurii* synergized with *Bacillus mucilaginosus* capable of $CO_2$ capture effect[J]. Chemical Engineering Journal, 2025, 520: 166156.
[71] Chen QS, Yuan XY, Wu AX, Liu YK. Enhancing $CO_2$ mitigation potential and mechanical properties of shotcrete in underground mining utilizing microbially induced calcium carbonate precipitation[J]. International Journal of Mining Science and Technology, 2024, 34(12): 1643-1653.
[72] Fazal T, Wang YZ, Zhang YY, Hou SW, Zhang CL. Evaluating microalgal-induced carbonate precipitation for marine carbon sequestration using *Chlorella* species[J]. Journal of Environmental Management, 2025, 394: 127245.
[73] Xue ZF, Cheng WC, Wang L, Xie YX, Qin P, Shi C. Immobilizing lead in aqueous solution and loess soil using microbially induced carbonate/phosphate precipitation (MICP/MIPP) under harsh pH environments[J]. Journal of Hazardous Materials, 2024, 480: 135884.
[74] 江昭明, 陈永贵, 付俊, 周罕, 文子豪. 微生物诱导碳酸盐沉淀修复镉污染尾矿试验研究[J]. 岩土工程学报, 2025, 47(6): 1308-1317.
Jiang ZM, Chen YG, Fu J, Zhou H, Wen ZH. Experimental study on remediation of cadmium-contaminated tailings using microbial-induced carbonate precipitation[J]. Chinese Journal of Geotechnical Engineering, 2025, 47(6): 1308-1317 (in Chinese).
[75] 张浩男. 微生物诱导碳酸钙沉淀技术的砂土固化实验研究[D]. 北京: 中国地质大学(北京), 2019.
Zhang HN. Experimental study on microbial induced calcium carbonate precipitation technology for sand solidification[D]. Beijing: China University of Geosciences, 2019 (in Chinese).
[76] Xue ZF, Cheng WC, Wang L, Hu WL. Effects of bacterial inoculation and calcium source on microbial-induced carbonate precipitation for lead remediation[J]. Journal of Hazardous Materials, 2022, 426: 128090.
[77] Jia CH, Fan YK, Wei QQ, Wang LJ, Zhang WJ. Soil organic matter enhances urease-mediated microbially induced carbonate precipitation: dual benefits for inorganic carbon sequestration and cadmium immobilization[J]. Journal of Hazardous Materials, 2025, 496: 139342.
[78] Yin TT, Lin H, Dong YB, Wei ZS, Li B, Liu CJ, Chen X. Inhibition of cadmium releasing from sulfide tailings into the environment by carbonate-mineralized bacteria[J]. Journal of Hazardous Materials, 2021, 419: 126479.
[79] Proudfoot D, Brooks L, Gammons CH, Barth E, Bless D, Nagisetty RM, Lauchnor EG. Investigating the potential for microbially induced carbonate precipitation to treat mine waste[J]. Journal of Hazardous Materials, 2022, 424: 127490.
[80] Song HW, Kumar A, Zhang YL. Microbial-induced carbonate precipitation prevents $Cd^{2+}$ migration through the soil profile[J]. Science of the Total Environment, 2022, 844: 157167.
[81] Ivanov V, Chu J. Applications of microorganisms to geotechnical engineering for bioclogging and biocementation of soil *in situ*[J]. Reviews in Environmental Science and Bio, 2008, 7(2): 139-153.
[82] Dejong JT, Soga K, Kavazanjian E, Burns S, Van Paassen LA, Al Qabany A, Aydilek A, Bang SS, Burbank M, Caslake LF, Chen CY, Cheng X, Chu J, Ciurli S, Esnault-Filet A, Fauriel S, Hamdan N, Hata T, Inagaki Y, Jefferis S, et al. Biogeochemical processes and geotechnical applications: progress, opportunities and challenges[J]. Géotechnique, 2013, 63(4): 287-301.
[83] Tao H, Jiang PP, Qu JL, Huang YD. An investigation into enhancing sand stability and minimizing dust emissions through bacterial treatment in arid regions[J]. Scientific Reports, 2025, 15: 29862.
[84] Yang JY, Wang YZ. Efficiency and characteristics of MICP in environments with elevated salinity, diminished oxygen, and lowered temperature: a microfluidics investigation[J]. Journal of Geotechnical and Geoenvironmental Engineering, 2025, 151: 04024151.
[85] Li YJ, Li YL, Guo Z, Xu Q. Durability of MICP-reinforced calcareous sand in marine environments: laboratory and field experimental study[J]. Biogeotechnics, 2023, 1(2): 100018.
[86] Van Paassen LA. Bio-mediated ground improvement: from laboratory experiment to pilot applications[C]//Han J, Alzamora DE. Geo-Frontiers 2011. Dallas, Texas, USA. Reston: ASCE, 2012: 4099-4108.
[87] Shi HQ, Ren XY, Yang RL, Wang JS, Xu HH, Liao XQ, Lou YY, Chen SH, Ye X, Wang XJ. $Mg^{2+}$ and magnesium ammonium phosphate (MAP)-induced anammox granulation for comparable nitrogen removal:

Implementation pathways and microbial mechanisms[J]. Water Research, 2025, 272: 122954.
[88] Yang WS, Xu L, Wang Z, Li K, Hu RZ, Su JF, Zhang LF. Synchronous removal of ammonia nitrogen, phosphate, and calcium by heterotrophic nitrifying strain *Pseudomonas* sp. Y1 based on microbial induced calcium precipitation[J]. Bioresource Technology, 2022, 363: 127996.
[89] Imran MA. Development of an eco-friendly approach for coastal erosion protection using bio-mediated technology[D]. Sapporo: Hokkaido University, 2021: xvii, 164.
[90] Liu YZ, Hu KJ, Pan ML, Dong W, Wang XJ, Zhu XY. Research and application of green technology based on microbially induced carbonate precipitation (MICP) in mining: a review[J]. Sustainability, 2025, 17(17): 7587.
[91] Wang XR, Li C, Shi YR, Zhang ZG, Chi QG, Wang PS. Improvements in saline soil and the law of water-salt transport based on salt inhibition using MICP technology[J]. Biogeotechnics, 2024, 2(1): 100055.
[92] Bakkeren E, Piskovsky V, Lee MNY, Jahn MT, Foster KR. Strain displacement in microbiomes *via* ecological competition[J]. Nature Microbiology, 2025, 10(12): 3122-3135.
[93] Chen MQ, Gowthaman S, Nakashima K, Kawasaki S. Influence of humic acid on microbial induced carbonate precipitation for organic soil improvement[J]. Environmental Science and Pollution Research, 2023, 30(6): 15230-15240.
[94] Fisher KA, Yarwood SA, James BR. Soil urease activity and bacterial *ureC* gene copy numbers: effect of pH[J]. Geoderma, 2017, 285: 1-8.
[95] Mugwar AJ, Harbottle MJ. Toxicity effects on metal sequestration by microbially-induced carbonate precipitation[J]. Journal of Hazardous Materials, 2016, 314: 237-248.
[96] Chen YB, Wang QY, Bian Y, Zhan LT, Gao YF, Guo HW, Wang YZ, Gao YQ. Effects of enzyme-induced carbonate precipitation (EICP) with different urease sources on the zinc remediation[J]. Journal of Hazardous Materials, 2024, 480: 136321.
[97] Li SD, Wu SJ, Wang SY, Liu G, Zhan YD, Tong J, Zhou K, Xie HG. High-efficiency removal of rare earth elements from acid mine drainage by microbially induced carbonate precipitation process[J]. Journal of Water Process Engineering, 2025, 71: 107134.
[98] Li ZF, Liu AZ, Sun CH, Li HT, Kong Z, Zhai HR. Biomineralization process of $CaCO_3$ precipitation induced by *Bacillus mucilaginous* and its potential application in microbial self-healing concrete[J]. Applied Biochemistry and Biotechnology, 2024, 196(4): 1896-1920.
[99] Ali MK, Yan HX, Mujahid M, Geng JB, Wang SQ, Yin RX, Zhao YM, Kumar A, Ali Laghari A, Han ZZ, Guo N, Zhao H. Simultaneous removal of $Mn^{2+}$ and $Sr^{2+}$ from wastewater *via* microbially induced carbonate precipitation (MICP) using immobilized *Bacillus licheniformis* K-1: a sustainable bioremediation approach[J]. Journal of Hazardous Materials, 2026, 501: 140645.