REVIEW PAPER

# Microbial bioremediation processes for radioactive waste

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**Abstract**−Microbial processes can affect the environmental behavior of priority radionuclides, and understanding these reactions is essential for the safe management of radioactive wastes and can contribute to the remediation of radionuclide-contaminated land. Underlying mechanisms that can control radionuclide solubility in biogeochemical systems can range from biosorption and biomineralization process, through direct (enzymatic) and indirect redox transformations. The mechanisms of enzyme-mediated reduction of problematic actinides, in principal, uranium (U), but including neptunium (Np), plutonium (Pu) and Americium (Am), are described in this review. In addition, the mechanisms by which the fission products technetium (Tc), cesium (Cs), and strontium (Sr) are removed from a solution by microorganisms are also described. The present review discusses the status of these microbiological processes, and the potential for cost-effective and scalable in situ remediation of radioactive waste.

## INTRODUCTION

The release of radioactive waste from sources including the testing of nuclear weapons, nuclear energy generation programs, nuclear weapon reprocessing, and accidental release is of considerable public concern [1-3]. In addition, it is likely to remain important with the increasing demand for nuclear energy [4]. Several national research programs have been initiated over the last two decades to assess the potential role of microbes in non-invasive in situ bioremediation processes to treat radionuclide-contaminated land [5,6]. In addition to dealing with land contaminated with legacy wastes, research into finding environmentally safe and publicly acceptable methods for the disposal of radioactive waste has also accelerated.

Bioremediation processes that harness the metabolic activity of microorganisms in situ use relatively low-cost, low-technology techniques, which generally have a high level of public acceptance [3,7]. Compared to other more invasive "civil engineering" methods [8], bioremediation may offer a less expensive route for cleaning up radionuclide contaminated soil and water [2]. Owing to the environmental disturbance, high cost input and limitations of current clean-up technologies, bioremediation has been developed as a potential approach that can be implemented in radioactive waste remediation [9,10].

Microorganisms, including intensively studied bacterial genera such as a *Geobacter, Deinococcus, Shewanella, Serratia, Kineococcus radiotolerans* sp. nov., and *Hymenobacter metalli* sp. nov., can act on radionuclides present in a high level of radioactive waste, affecting their solubility, mobility, and bioavailability [1-3,11,12]. The goal of this article is to give an overview of the range of radioactive wastes remediated using such microorganisms, the mechanisms involved, and the possible applications for these processes. Particular emphasis is placed on the bioremediation of actinides and fission products.

### 1. Nuclear Fuel Cycle

When looking at the environmental impact of the nuclear fuel cycle, the mining, fuel manufacture, reprocessing, recycled fuel and waste management are all a part of this cycle (Fig. 1). And the predominant waste-producing operations are defined as mining and
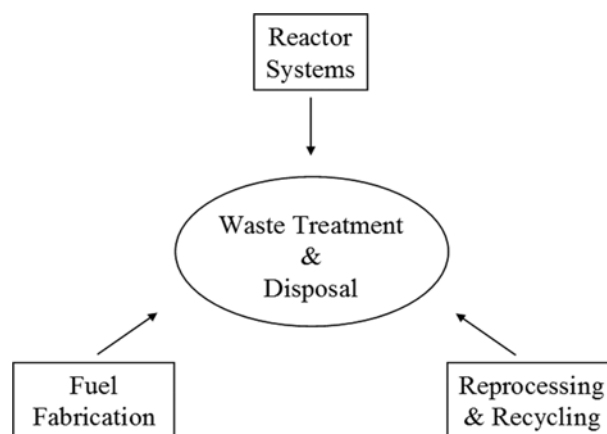
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Fig. 1. Diagram of the nuclear fuel cycle and waste treatment.

**Table 1. Characteristics of isotopes of U, Np, Pu, Am, Tc, Cs, Sr and I: half-life and oxidation state**

| Radionuclide | Isotope | Half Life (years) | Known oxidation states | Stable oxidation state under oxic conditions | Stable oxidation state under reducing conditions |
|---|---|---|---|---|---|
| Uranium | 235 | $7.04 \times 10^8$ | IV, V, VI | $U(VI)O_2^{2+}$ | $U(IV)O_2$ (s) |
|  | 238 | $4.47 \times 10^9$ |  |  |  |
| Neptunium | 237 | $2.14 \times 10^6$ | IV, V, VI | $Np(V)O_2^+$ | $Np(IV)O_2$ |
|  | 238 | $8.77 \times 10^1$ |  |  |  |
| Plutonium | 239 | $2.41 \times 10^4$ | III, IV, V, VI | $Pu(IV)O_2$ | $Pu(IV)O_2$ |
|  | 240 | $6.55 \times 10^3$ |  | $Pu(V)O_2^+$ |  |
|  | 241 | $1.44 \times 10^1$ |  |  |  |
|  | 242 | $3.76 \times 10^5$ |  |  |  |
| Americium | 241 | $4.32 \times 10^3$ | II, III, IV, V, VI, VII | $AmO_2^+$ | Am(III) |
| Technetium | 99 | $2.13 \times 10^5$ | IV, V, VI, VII | $Tc(VII)O_4^-$ | Tc(IV) |
| Cesium | 137 | 30 | - | - | - |
| Strontium | 90 | 29 | - | - | - |

milling of uranium ore, reprocessing of spent nuclear fuel, and the decommissioning of the facilities [13]. The majority of radioactive wastes, especially those containing transuranic elements, have been generated by nuclear power reactors [13,14]. Contamination from fuel reprocessing is much less than from weapons testing [14], but this mode of release can also result in localized high levels of contamination [15]. The scale of the legacy of nuclear activities is perhaps best illustrated by the inventories at the US Department of Energy's 120 sites, which contain 1.7 trillion gallons of contaminated ground water and 40 million cubic meters of contaminated soil and debris [5]. More than 50% of the sites are contaminated with radioactive waste, with the radionuclides being $^{238}U/^{235}U$, $^{237}Np$, $^{239}Pu$, $^{241}Am$, $^{99}Tc$, $^{137}Cs$, and $^{90}Sr$.

## 2. Bioremediation of Actinides and Fission Products

The successful bioremediation of radionuclides relies on a complex interplay of biological, chemical, and physical processes. Underpinning mechanisms include oxidation, reduction, dissolution, precipitation, sorption and leach, and these processes can influence the toxicity and transport of radionuclides in biogeochemical systems. When these mechanisms are performed by organisms, bioprocesses present opportunities for bioremediation of radionuclides in the environment - either to immobilize them in place to accelerate their removal [8,16,17]. The half-lives and most common oxidation states for the key radionuclides of U, Np, Pu, Am, Tc, Cs, and Sr are given in Table 1. As many of these radionuclides such as U, Np, Pu, Am and Tc are both redox-active and less soluble when reduced, bioreduction offers significant promise for controlling the solubility and mobility of target radionuclides in contaminated waste. This and other key microbial processes involved in the mobilization or immobilization of key radionuclides of interest are summarized in Table 2. There is a large and increasing body of information addressing the biogeochemistry of uranium, although comparatively little work has addressed microbial interactions with other important actinides and key fission products. Uranium (as U(VI)) and also technetium (as Tc(VII)) have been shown to be susceptible to enzymatic reduction by microbes. The oxidized forms of U and Tc are highly soluble in aqueous media and are mobile in ground water, whereas the reduced forms are insoluble and often

**Table 2. Summary of microbial processes of radionuclides**

| Process | U | Np | Pu | Am | Tc | Cs | Sr | I |
|---|---|---|---|---|---|---|---|---|
| Oxidation | + | n.d. | n.d. | + | + | n.a. | n.a. | + |
| Reduction | + | + | + | + | + | n.a. | n.a. | + |
| Dissolution | + | ? | + | ? | + | + | + | + |
| Precipitation | + | + | + | + | + | ? | + | + |
| Biosorption | + | + | + | + | ? | + | + | + |

n.d.: Not determined
n.a.: Not applicable

precipitable [18]. A direct enzymatic reduction of soluble U(VI) and Tc(VII) to insoluble species has been characterized and summarized in Fig. 2(a). Furthermore, "indirect" reduction mechanisms may also be important in the immobilization of Tc(VII) in sediments, as biologically reduced Fe(II) can transfer electrons efficiently
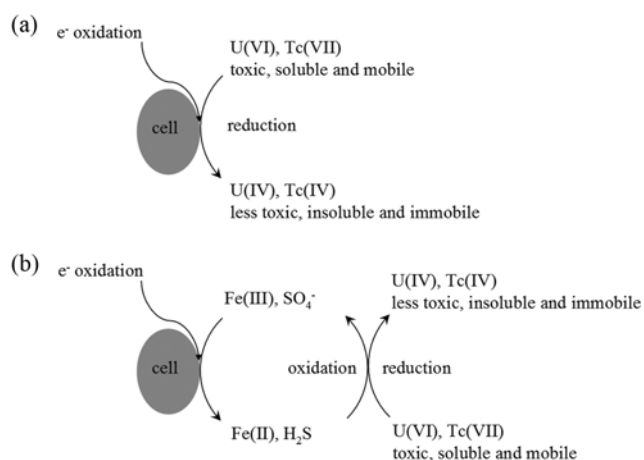


**Fig. 2. Microbial processes of soluble radioactive and less toxic metals into insoluble and less toxic form. (a) Direct reduction of radioactive heavy metals by microorganisms. (b) Indirect electron transfer by metal-reducing microorganisms to radionuclides.**

to Tc(VII) (Fig. 2(b)).

## 3. Characteristics of Uranium and Microbial Interactions

The alpha emitter Uranium-238 ($^{238}$U) is the most common isotope of uranium found in nature, and can be used as a source material for creating plutonium-239, which can in turn be used as nuclear fuel. Around 99% of natural uranium is uranium-238, which has a half-life of $4.47 \times 10^9$ years [4,19,20]. The understanding of the microbial biotransformation of U(VI) to U(IV), producing the insoluble mineral uraninite, has been viewed as a potential mechanism for the sequestration of environmental uranium contamination. Over the past decade, it has been established that a variety of bacteria exhibit this reductive capacity [6]. Various aspects of the microbial process have been explored experimentally, to develop a practical approach for the acceleration of in situ bioremediation using this approach.

Lovley and coworkers first demonstrated that the Fe(III)-reducing bacteria *Geobacter metallireducens* and *Shewanella oneidensis* can conserve energy for anaerobic growth through the reduction of U(VI) [21]. Other organisms, including a *Clostridium* sp. [22] and the sulfate-reducing bacteria *Desulfovibrio desulfuricans* [23] and *Desulfovibrio vulgaris* [24], also reduce uranium but are unable to conserve energy for growth through this process. *D. vulgaris* was the first organism in which the enzyme system responsible for U(VI) reduction was characterized. Purified tetraheme cytochrome *c3* was shown to function as a U(VI) reductase in vitro, in combination with hydrogenase, its physiological electron donor [24]. Payne and coworkers used a cytochrome *c3* mutant of the close relative *D. desulfuricans* strain G20 to confirm a role for this electron transfer protein in hydrogen-dependent U(VI) reduction, but suggested additional pathways from organic electron donors to U(VI) which bypassed the cytochrome [25].

Early studies with *Geobacter sulfurreducens* suggested that outer-surface c-type cytochromes might play a role in U(VI) reduction, but it has recently been suggested that there is a substantial U(VI) reduction at the surface of the electrically conductive *pili*, which have been described as microbial nanowires [26]. Orellana confirmed that wild-type cells did not precipitate uranium along *pili* as previously reported, but that U(IV) was precipitated at the outer cell surface. These findings are consistent with previous studies that have suggested that *G. sulfurreducens* requires outer-surface c-type cytochromes, but not *pili*, for the reduction of soluble extracellular electron acceptors [26-28]. *S. oneidensis* MR-1 contains 42 putative c-type cytochromes [27,28] and mutagenesis studies have shown that some of them are essential for metal reduction. Among these, periplasmic decaheme, cytochrome (MtrA), decaheme cytochromes (MtrC and OmcA) exposed on the outer membrane (OM), and a tetraheme cytochrome (CymA) anchored to the cytoplasmic membrane, have all been known to be required for uranium reduction [28,29]. Furthermore, a trans-OM protein MtrB, part of a membrane-spanning protein complex (MtrABC), is essential for uranium reduction [30,31].

In addition to biochemically pure culture studies, sediment microcosm and in situ studies have focused on the stimulation of indigenous Fe(III)-reducing bacteria through simple amendments to the subsurface such as the addition of an electron donor, e.g., acetate, for U(VI) reduction and precipitation. Under these conditions, *Geobacter* species often predominate, and for this reason Renshaw

et al. elucidated the mechanism of U(VI) reduction by a model *Geobacter sulfurreducens*, using X-ray absorption spectroscopy (XAS) to define the uranium speciation in situ [32]. These studies suggest a two-step process involving an enzymatic reduction of U(VI) to U(V), followed by disproportionation of U(V), to give U(IV) (and U(VI)) for further rounds of reduction.

A successful biostimulation experiment for immobilization was performed at the Uranium Mill Tailings Remedial Action (UMTRA) site in Rifle, Colorado, USA. Laboratory studies using native sediments clearly showed that the addition of acetate as an electron donor resulted in the precipitation of U(IV). Acetate was added successfully to stimulate uranium reducing anaerobes, including both Fe(III)- and sulfate-reducing bacteria, resulting in uranium concentrations dropping to below the US Environmental Protection Agency's drinking water standard [33]. Engineers, hydrologists, geochemists, and microbiologists worked together to design a biostimulation experiment in which over 1,000 gallons of acetate was added to a uranium-contaminated site. Over a period of about two months, concentrations of U(VI) in the ground water decreased as acetate was consumed. The next steps include addressing competition by other microorganisms for electron donors and the long-term stabilization of the precipitated uranium at the UMTRA site. The potential for stimulating microbial U(VI) reduction as an in situ bioremediation strategy for uranium-contaminated-ground water was also evaluated in uranium-contaminated sediment from the Field research center (FRC), Oak Ridge, TN, USA. Over the course of 31 years, millions of gallons of plating wastes containing high concentrations of uranium and nitric acid were generated, resulting in significant subsurface contamination.

## 4. Characteristics of Neptunium and Microbial Interactions

Neptunium, an alpha-emitting transuranic radionuclide produced in ton quantities in nuclear reactors, is of particular importance because of its long half-life ($2.14 \times 10^6$ years), high radiotoxicity, and relatively high solubility as $Np(V)O_2^+$ under oxic conditions. In contrast, under-reducing conditions, Np(IV) species are expected to dominate and Np(IV) can be removed from the solution by hydrolysis and a reaction with the surfaces [34,35]. Fisher and coworkers [36] reported negligible Np uptake by marine algae, and in the only detailed study performed prior to 1993, low uptakes (10 mg/g dry weight) were reported with *Pseudomonas aeruginosa*, *Streptomyces viridochromogenes*, *Scenedesmus obliquus* and *Micrococcus luteus* [37]. Songkasiri and coworkers suggested that *P. fluorescens* can biosorb appreciable quantities of Np, removing 85% of 4.75 mM Np(V) removed from the solution at pH 7 [38]. Neptunyl species ($NpO_2^+$) can also be biologically reduced to insoluble Np(IV) under anaerobic conditions [39-42], although in some studies complexation of Np(IV) by fermentation intermediate products prevented its precipitation [39]. Although Np(IV) is easily oxidized in solution, it is stabilized in the presence of complexing ligands. For example, *Shewenella putrefaciens* reduced Np(V) to Np(IV), which was then precipitated from solution as Np(IV) phosphate in the presence of a *Citrobacter* species (now reclassified as a *Serratia* species) with high phosphatase activity [39]. Reduction of Np(V) to Np(IV) by cell suspension of *S. putrifaciens* MR-1 [41] and by the sulfate-reducing bacteria *Desulfovibrio desulfuricans* [40] have also been reported. Law et al. also elucidated the microbial redox cycle

of neptunium to in sediment systems, with a reduction of Np(V) to Np(IV) mediated by the indigenous microbial community, with biogenic Fe(II) implicated as the reductant [43]. Furthermore, this study suggested that once Np(IV) is formed, it is surprisingly resistant to reoxidation by air or nitrate additions. These observations have significant implications for contaminated land and geological disposal scenarios in which bioreduction could enhance Np retention [44].

## 5. Characteristics of Plutonium and Microbial Interactions

Plutonium is a transuranic radioactive chemical element with various fissible isotopes. Studies on plutonium are far more challenging than for the other actinides above, owing to its high radiotoxicity and complex redox chemistry. Although the most stable oxidation state of Pu is (IV) under most environmental conditions, Pu(III), Pu(V), and Pu(VI) can also be stable. Several studies have attempted to clarify the potential role of microorganisms in controlling Pu solubility. Most studies on Pu(IV)-microbe interactions have used very low concentrations of the radionuclide, but one investigation utilized a stock solution of $^{239}$Pu of 2 mg/mL as the carbonate, introduced to natural sediments [45]. All of the Pu was removed after four months, with only 34% of the removal attributed to biosorption using heat-killed controls, implicating additional biochemical mechanisms. Laboratory studies have also suggested that the reduction of Pu(IV) to Pu(III) can be achieved by Fe(III)-reducing bacteria, although the Pu(III) was reported to reoxidize spontaneously [46]. Although Pu(IV) reduction may lead to solubilization of sediment-bound Pu(IV), it will yield a trivalent actinide that could react with a range of microbially produced ligands [47]. The direct enzymatic reduction of Pu(VI) and Pu(V) to Pu(IV) by bacterial cell suspension of *S. putrefaciens*, *S. oneidensis* or *G. metallireducens* has also been reported [41]. The potential exists for the microbial oxidative dissolution of the more stable and environmentally predominant form of Pu(IV) to the more soluble and bioavailable form of Pu(V) and Pu(VI), with some authors proposing that microbes may indirectly affect the oxidation state and solubility of Pu by changing the Eh and pH of the medium, as well as by producing sequestering agents [41,48,49]. Kimber et al. [50] also investigated the biogeochemistry of Pu in contaminated soil undergoing stimulated "bioreduction" processes, as microbial processes have the potential to mobilize Pu through numerous mechanisms including the reduction of Pu(IV) to the potentially more mobile Pu(III). A broad suite of geochemical markers and terminal electron acceptors were monitored throughout the incubation of the microcosms. and the Pu in solution was quantified, showing that only very low levels of Pu were released from the sediment concomitant with metal (Fe(III)) reduction. This suggests that Pu in contaminated sediments is relatively resistant to reductive mobilization.

## 6. Characteristics of Americium and Microbial Interactions

Americium-241 has a half-life of 432.2 years, by emitting alpha particles it turns into neptunium-237. Americium can exist in oxidation states ranging from II to VII, although II is only found in the solid state, and the environmentally important, oxidation state of Am is trivalent oxidation state. Microorganisms can play a pivotal role in changing $^{241}$Am(III) solubility and efficient biosorption by *E.coli*, a marine bacterium, *Rhizopus arrhizus*, and *Candida utilis* [51-54]. In particular, Marumo et al. reported the new radionuclides-tolerant bacterial strains such as *Flavobacterium* spp., *Pseu-*domonas gladioli, *Chryseobacterium indologenes* and *Ochrobactrum anthropi*, which have an influence on the microbial community responsible for the degradation processes of organic waste [55,56].

## 7. Characteristics of Technetium and Microbial Interactions

Technetium-99 is a long-lived (half-life, $2.13 \times 10^5$ years), beta-emitting radionuclide formed in high yield in nuclear reactors that has been released into the environment in authorized and accidental discharges and is an important component of radioactive wastes. The redox chemistry of technetium is the major control on its environmental solubility. Under oxic conditions, technetium is present as pertechnetate ion (Tc(VII); $TcO_4^-$), which is only weakly sorbed into mineral surfaces and is one of the most mobile radionuclide species in the environment [57]. Tc(VII) has weak ligand-complexing capabilities and is difficult to remove from a solution using conventional 'chemical' approaches. Several reduced forms of the radionuclide are insoluble, however, and metal-reducing microorganisms can reduce Tc(VII) and precipitate the radionuclide as a low-valency oxide [Tc(IV); $TcO_2$]. In an early study on Tc(VII) bioreduction, a novel phosphor imaging technique was used to show a reduction of the radionuclide by *Shewanella putrefaciens* and *Geobacter metallireducens*, with similar activities subsequently detected in laboratory cultures of *Rhodobacter sphaeroides*, *Paracoccus denitrificans*, some *Pseudomonas species* [58], *Escherichia coli* [59] and a range of sulfate-reducing bacteria [58,60]. The biochemical basis of Tc(VII) reduction has been best characterized in *E. coli*. Initial studies demonstrate that anaerobic, but not aerobic, cultures of *E. coli* reduced Tc(VII) with the reduce radionuclide precipitated within the cell via the hydrogenase component of the formate hydrogen lyase (FHL) complex [59]. *D. desulfuricans* [61] and related strains [60] were also able to utilize formate as an efficient electron donor for Tc(VII) reduction. This is consistent with the existence of a rudimentary FHL complex (consisting of a formate dehydrogenase coupled to a hydrogenase via a cytochrome) located in the periplasm of these strains [62]. Accordingly, the site of reduced Tc precipitation was identified as the periplasm in *D. desulfuricans* [61], and more recent studies have confirmed a role for a periplasmic Ni-Fe hydrogenase in Tc(VII) reduction by a relative in the N subclass of the Proteobacteria, the sulfate-reducing bacterium *Desulfovibrio fructosovorans* [63]. Other studies on biostimulated "post reduction" microcosm systems [64,65] suggest that the introduction of air results in Fe(II) oxidation and mobilization of significant quantities of Tc to solution (oxidized from Tc(IV) to (VII)). Nitrate addition, in contrast, resulted in the "bio-oxidation" of Fe(II). However, despite the presence of excess oxidant there was limited mobilization of the reduced, sediment-bound Tc, which remained largely as Tc(IV). Tc(VII) was variably reductively precipitated as Tc(IV) by all the reduced iron phases used in this study, and when associated with the mineral was largely recalcitrant to reoxidation or remobilization, and was retained as Tc(IV). McBeth et al. reported that the recalcitrance of Tc to reoxidation and thus remobilization in these minerals is similar to observations with sediment reoxidation, where, typically, significant fractions of Tc are retained on sediments as short-range-ordered, hydrous Tc(IV)-like phases co-associated with Fe, and are stable over several months of reoxidation [66]. Recent works suggest that Tc associated with Fe(II)-bearing sediments is recalcitrant to reoxidation even though significant

reoxidation of Fe(II) to Fe(III) occurs during both air and nitrate reoxidation [67,68]. This has implications when considering remediation approaches and in predictions of the long-term fate of Tc in the nuclear legacy.

## 8. Characteristics of Cesium and Microbial Interactions

Cesium-137, with a half-life of about 30 years, is a radioactive isotope formed as a fission product. Significant amounts of ^{137}Cs were released into the environment during nearly all nuclear weapon tests and some nuclear accidents. Recent reports have shown that higher than expected levels of ^{137}Cs persisted in a mobile form in the environment following the Chernobyl, Goiânia, and Fukushima accidents, with accumulation of the Cs by microorganisms potentially playing a role in passing the radionuclide to higher trophic levels in the food chain [69]. Pure culture studies have shown that microbial biosorbents are relatively inefficient for Cs uptake [70], but uptake of $Cs^+$ by actively metabolizing microorganisms is more efficient. Owing to the similarity of the $K^+$ and $Cs^+$ cations, both are taken up by the same metabolism-dependent transport systems. Indeed, broad-specificity alkali earth metal uptake transporters have been reported in all microbial groups [69,71]. Most studies on microbial $Cs^+$ accumulation have used relatively simple, well defined laboratory solutions. Tomioka and coworkers studied cesium-accumulating bacteria isolated from soil [72]. Cs uptake is optimal at pH 8.5 and cannot be modeled as simple sorption phenomena. Potassium and rubidium inhibit Cs accumulation supporting the hypothesis that Cs is taken up through the potassium transport system [73].

## 9. Characteristics of Strontium and Microbial Interactions

The fission product Strontium-90 (^{90}Sr) is produced in nuclear explosions or by nuclear reactors. Strontium-90 is not quite as likely to be released as cesium-137 as a part of a nuclear reactor accident because it is much less volatile, but is probably the most dangerous component of radioactive fallout from a nuclear weapon [74,75]. Strontium-90 undergoes beta decay, with a half-life of about 29 years. It is present in significant quantities in spent nuclear fuel and in radioactive waste from nuclear reactors and in nuclear fallout from nuclear tests. Bioaccumulation of Sr has been reported for several microorganisms [76]. Consequently, there is considerable interest in using microorganisms to remove radioactive Sr from waste streams and contaminated sites. Mixed cultures of bacteria, isolated from low-level radioactive waste leachates preferentially accumulated [85]Sr in a mineral salt medium containing a mixture of radionuclides [77]. Sr-binding activity in *Microccous luteus* is localized on the cell envelope and is sensitive to pretreatment. Bound Sr can be displaced by chelating agents, divalent cations or $H^+$ (other monovalent cations are less effective at displacing Sr). Sr binding in *M. luteus* is reversible, though both ion exchange, mediated by acidic cell surface components and intracellular uptake may be involved [76]. Recently, Thorpe et al. investigated that an increase in $OH^-$ and $CO_3^{2-}/HCO_3^-$ during nitrate reduction may lead to increased adsorption of $Sr^{2+}$ to mineral surfaces, and once over-saturation was reached, the precipitation and or incorporation of $Sr^{2+}$ into carbonate phases at a high Sr/Ca ratio [78]. Their results showed that bioreduction approaches may be relevant to a range of problematic radionuclides including redox active U and Tc as well as ^{90}Sr, and thus provide a holistic remediation strategy where co-contamina-

tion of these radionuclides occurs. Radio-strontium incorporation into carbonate phases is desirable in remediation scenarios as they are redox-insensitive phases and are potentially more resistant to remobilization than adsorbed $Sr^{2+}$. Indeed, under constrained conditions, bioreduction may have the potential to co-treat redox active radionuclides and ^{90}Sr increasing the range of applications for this clean-up technology.

## CONCLUSION

The microbial bioremediation of radionuclides is of paramount importance to the development of new strategies and technologies to protect the environment. Microorganisms have the potential to control the solubility of radionuclides through many mechanisms, either directly through interactions with the microbial cell, or through indirect transformations driven by local changes in the chemistry of the environment attributed to microbial activity. Of particular interest are microbial processes that can affect the environmental behavior of redox sensitive radionuclides, and understanding of the underpinning redox reactions is essential for the safe management of radioactive wastes. Over the last decade there have been significant advances in our understanding of the biogeochemical controls on key radionuclides, facilitated by advances of factors including improvements in techniques available in specialist areas as diverse as microbiology, molecular biology, and analytical chemistry. However, there is in many cases still a very significant gap between knowledge of radionuclide-microbe interactions obtained from laboratory studies using pure cultures of microorganisms under well-defined conditions, and detailed understandings of the complex mechanisms of key radionuclides in microbial communities. Therefore, it is necessary to understand the mechanism that enables microbes to eliminate the radionuclides from contaminated sites.

Studies to identify the molecular network in complex environmental systems, using "omics" based system biology approaches will be crucial to support the development of bioremediation programs that can have a positive impact on environmental contamination. Supported by omics-based studies ongoing in many labs worldwide, we can expect this research area to develop further in the near future, delivering more robust technologies for the bioremediation of radioactive contaminants and potential applications in decommissioning and radwaste disposal.

## ACKNOWLEDGEMENTS

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