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Microbial Selenate Detoxification Linked to Elemental Sulfur Oxidation: Independent and Synergic Pathways

Liuliu Li[a,b], Baogang Zhang[a,b*], Lei Li[a,b], Alistair G.L. Borthwick[c,d,e]

[a] *School of Water Resources and Environment, China University of Geosciences Beijing, Beijing 100083, China*

[b] *Key Laboratory of Groundwater Circulation and Evolution (China University of Geosciences Beijing), Ministry of Education, Beijing 100083, China*

[c] *St Edmund Hall, Queen's Lane, Oxford OX1 4AR, UK*

[d] *School of Engineering, The University of Edinburgh, The King's Buildings, Edinburgh EH9 3JL, UK*

[e] *School of Engineering, Computing and Mathematics, University of Plymouth, Drakes Circus, Plymouth PL4 8AA, UK*

[*] Corresponding author. Tel.: +86 10 8232 2281; Fax: +86 10 8232 1081; E-mail: baogangzhang@cugb.edu.cn (B. Zhang)

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ABSTRACT

Elevated selenium levels in the environment, with soluble selenate [Se(VI)] as the common chemical species, pose a severe threat to human health. Anaerobic Se(VI) bioreduction is a promising approach for selenium detoxification, and various organic/inorganic electron donors have proved effective in supporting this bioprocess. Nevertheless, autotrophic Se(VI) bioreduction driven by solid inorganic electron donors is still not fully understood. This work is the first to employ elemental sulfur [S(0)] as electron donor to support Se(VI) bioreduction. A batch trial with mixed culture demonstrated the feasibility of this bioprocess, with Se(VI) removal efficiency of 92.4 ± 0.7% at an initial Se(VI) concentration of 10 mg/L within 36 h. Continuous column tests showed that increased initial concentration, flow rate, and introduction of NO_3^--N depressed Se(VI) removal. Se(VI) was mainly bioreduced to solid elemental Se with trace selenite in the effluent, while S(0) was oxidized to SO_4^{2-}. Enrichment of *Thiobacillus*, *Desulfurivibrio*, and *Sulfuricurvum* combined with upregulation of genes *serA*, *tatC*, and *soxB* indicated Se(VI) bioreduction was coupled to S(0) oxidation. *Thiobacillus* performed S(0) oxidation and Se(VI) reduction independently. Intermediate metabolites as volatile fatty acids, hydrogen, and methane from S(0) oxidation were utilized by heterotrophic Se(VI) reducers for Se(VI) detoxification, indicative of microbial synergy.

Keywords: Elemental sulfur; Selenate; Microbial reduction; Biodetoxification

1. Introduction

Selenium has an unusually narrow gap between its benefits and toxicity to humans (Lai et al., 2016). For a typical adult person, 40 to 55 μg/day of selenium is needed to support their basic nutritional needs. However, adverse impact occurs through selenosis if total selenium intake exceeds 400 μg/day (Navarro-Alarcon and Cabrera-Vique, 2008). The World Health Organization defines the selenium intake value of 40 μg/L as reference, whereas the United States Environmental Protection Agency rules that the maximum concentration of selenium must not exceed 50 μg/L in drinking water (Zhu et al., 2017; Nancharaiah et al., 2018). However, excessive concentrations of selenium are found in the environment. For example, selenium at 6000 μg/L was reported for groundwater in Utah, USA, whereas up to 9000 μg/L selenium was detected in certain areas of Colorado, USA (Ji and Wang, 2017). It is obviously important to reduce such high selenium levels below the safety threshold, especially in regions where drinking water is abstracted.

Selenate [Se(VI)] and selenite [Se(IV)] are the main forms of selenium species found in the natural aqueous environment (Latorre et al., 2013). Both Se(VI) and Se(IV) can be transformed to elemental selenium [Se(0)] enabling selenium removal due to its lower solubility, toxicity, and bioavailability (Lai et al., 2016; Song et al., 2021; Wang et al., 2021). With eco-friendly, cost-effective and in situ application advantages, microbial reduction of Se(VI) to Se(0) is preferred among remediation methods (Lenz et al., 2009; Fu et al., 2014; Qiao et al., 2018), especially with indigenous microorganisms (Li et al., 2021). Bioreduction of Se(VI) can be achieved

through either heterotrophic or autotrophic processes (Ontiveros-Valencia et al., 2016; Zhang et al., 2019). Compared with heterotrophic metabolism, autotrophic reduction is advantageous in that it prevents secondary pollution and produces less biomass, making it a promising option that is also practicable and efficient (Liu et al., 2016; Zhang et al., 2018).

Electron donors are absolutely essential in the process of autotrophic Se(VI) bioreduction. For example, hydrogen (H_2) is used in gas-supported bioreduction of Se(VI) (Zhou et al., 2018), but requires considerable care in its usage and storage from a safety perspective. Solid inorganic electron donors that support autotrophic bioreduction for metal oxyanion detoxification have also been reported (Shi et al., 2019; Lu et al., 2020; He et al., 2021), of which elemental sulfur [S(0)], an insoluble, stable and low-cost industrial product, has gained increasing scrutiny (Wang et al., 2021). S(0) is able to serve as an electron donor for autotrophic bioprocesses (Sahinkaya et al., 2014; Li et al., 2019). Many studies have explored the performance of S(0) in assisting in the bioreduction of contaminants such as nitrate (NO_3^--N), perchlorate (ClO_4^-), vanadate [V(V)], and chromate [Cr(VI)] (Zhang et al., 2018; Ucar et al., 2019). In spite of this, autotrophic microbial Se(VI) reduction supported by S(0) has been little researched, and the mechanism of Se(VI) bioreduction driven by S(0) remains largely unknown.

Herein, bioreduction of Se(VI) supplemented by S(0) is investigated through both batch and column trials, with analysis of product characterization, microbial community dynamics, functional genes, and intermediate metabolites. The objectives

85 of this paper are: (1) to examine the feasibility and performance of S(0)-supported
86 Se(VI) bioreduction; (2) to reveal microbial dynamics at community, gene, and
87 metabolic levels; and (3) to analyze the mechanisms related to S(0)-driven Se(VI)
88 bioreduction. The present study focuses on bioremediation as a potentially practicable
89 technology for Se(VI)-polluted environments.

90 2. Materials and methods

91 *2.1. Batch trial*

92 Three groups of 250 mL volume plexiglass bottles were employed as biotic,
93 sterilized, and abiotic batch reactors. Strict anaerobic conditions were preserved by
94 sealing each bottle with butyl rubber stoppers and aluminum crimp caps. Before being
95 transferred to the bottles, simulated groundwater was flushed with nitrogen gas for 30
96 min to remove oxygen. The biotic group with 50 mL aquifer sediment from a smelting
97 site in China was combined with 5 g S(0) and 200 mL simulated groundwater
98 containing the following basic constituents (g/L): $CaCl_2$ 0.2464; NH_4Cl, 0.1557;
99 $MgCl_2 \cdot 6H_2O$, 1.0572; NaCl, 0.4459; KCl, 0.0283; KH_2PO_4, 0.0299; and $NaHCO_3$,
100 0.5040. The pH, ORP, and electrical conductivity of aquifer sediment were 7.99, -
101 48.2 mV, and 424.9 μS/cm, respectively. Concentrations of organic matter and
102 available sulphur in aquifer sediment were 34.4 g/kg and 21.9 mg/kg, respectively,
103 whereas the cell dry weight was 6.7 mg/g soil. Concentrations of trace elements were
104 as follows: Fe (84.2 mg/kg), Mn (2.22 mg/kg), Zn (0.19 mg/kg), Pb (0.13 mg/kg), V
105 (0.34 mg/kg), Cr (0.33 mg/kg). The medium was supplemented with bicarbonate as

106 the inorganic carbon source with inorganic carbon concentration at 72 mg/L. Na_2SeO_4
107 was used as the source of Se(VI), at 10 mg/L concentration. All regents were
108 purchased from Sinopharm Chemical Reagent Co., Ltd. (Beijing, China) with
109 analytical grade, and utilized directly without further processing. The sterilized group
110 was supplemented with S(0), simulated groundwater, and 50 mL high temperature
111 sterilized aquifer sediment (20 min at 121 ºC). The abiotic group was supplemented
112 with S(0) and simulated groundwater, but without aquifer sediment. All batch reactors
113 were run for two months before data collection. The cycle of 72 h was set to evaluate
114 Se(VI) reduction in different groups. All experiments were conducted at room
115 temperature (22 ± 2 ºC) and in triplicate.

116 *2.2. Column trial*

117 A biological column of 25 cm in length and interior diameter of 5 cm was made
118 of plexiglass. The column was covered by aluminum foil to prevent ingress of
119 sunlight. The column was inoculated with 50 mL aquifer sediment supplemented by
120 200 g S(0). The remaining space was filled with quartz sand. All additives were mixed
121 thoroughly beforehand. The synthetic water component was the same as in the batch
122 experiment. A peristaltic pump (BT100-1L, Longer, UK) drove influent through the
123 column at a prescribed flow rate (Zhong et al., 2011). Se(VI) and other ingredients at
124 specified concentrations were added to the influent. The column experiment took
125 place over 258 consecutive days divided into five phases in order to investigate the
126 effects of different hydrodynamic and hydrochemical factors on S(0)-based Se(VI)
127 removal (Table 1). Data were then obtained on Se(VI) removal performance, reaction

128 products, microbial communities, functional genes and intermediate metabolites.

129 *2.3. Chemical analysis and characterization*

130 Aqueous samples were filtered through 0.22 μm filter membrane, then Se(0) and
131 the suspended microbial cells were removed by centrifuging at 12580 g for 20 min.
132 pH, ORP, and conductivity were monitored by a multifunctional meter
133 (SevenExcellenceS400, Mettler-Toledo, Switzerland) (Zhu et al., 2021). Inductively
134 coupled plasma-mass spectrometry (ICP-MS, X series, Thermo Fisher, Germany) was
135 used to determine total Se concentration. Se(IV) concentration was analyzed using a
136 spectrophotometer (UV2300, Shanghai, China) as previously described (Li et al.,
137 2014; Jain et al., 2015). Se(VI) was determined after being reduced to Se(IV) by
138 hydrochloric acid (Brimmer et al., 1987; Xie et al., 2017). NO_3^--N, nitrite (NO_2^--N)
139 and ammonium (NH_4^+-N) concentrations were also determined using
140 spectrophotometry (Shi et al., 2020a). Ion chromatography monitored sulfate (SO_4^{2-}),
141 sulfite (SO_3^{2-}) and thiosulfate ($S_2O_3^{2-}$) concentrations (Basic IC 792, Metrohm,
142 Switzerland) (Shi et al., 2020b).

143 Analyses of the surface morphology and composition of the biomass samples
144 were conducted using a scanning electron microscope (SEM) (JEOL JAX-840,
145 Hitachi, Japan) equipped with energy dispersive X-ray spectroscopy (EDS). X-ray
146 photoelectron spectroscopy (XPS) measurement was carried out to identify
147 components of the produced deposits, using a Kratos XSAM-800 spectrometer (UK)
148 (Zhang et al., 2012). Surface crystallographic structures of these deposits were

149 determined by X-ray diffraction (XRD) (Rigaku-D/MAX-PC 2500, Rigaku, Japan).

150 *2.4. Microbial and metabolic analyses*

151 Microbial community analysis was conducted for both inoculum and biomass
152 obtained from each stage of the biological column. Samples were collected for DNA
153 extraction, amplification, and purification, then sent to Shanghai Majorbio
154 Technology (Shanghai, China) for high-throughput 16S rRNA gene sequencing using
155 MiSeq (Illumina, USA) (Zhang et al., 2019; Wang et al., 2020). Prior to analysis,
156 centrifugation was used to remove moisture from the samples. Operational taxonomic
157 units (OTUs) were clustered from sequences with a 0.03 distance limit (equivalent to
158 97% similarity). Rarefaction curves and alpha diversity indexes were acquired by
159 Mothur (version v.1.30.1) software. Non-metric multi-dimensional scaling (NMDS)
160 analysis was performed using Bray-Curtis similarities to cluster the microbial
161 community composition.

162 Real-time quantitative PCR (qPCR) was chosen to quantify the functional genes
163 with previously reported primers (Table S1, Supporting Information) during the S(0)-
164 driven Se(VI) bioreduction process (Throbäck et al., 2004; Bru et al., 2007; Ma et al.,
165 2007; Meyer et al., 2007; Wen et al., 2016). Electron transfer compounds, such as
166 cytochrome c (Cyt c) on cell surfaces and intracellular nicotinamide adenine
167 dinucleotide (NADH), were also examined, and their measurements normalized to
168 volatile suspended solids (VSS) according to previously documented procedure
169 (Zhang et al., 2019; Shi et al., 2020a; Zhang et al., 2021). Intermediate metabolites

170 were also monitored. Volatile fatty acids (VFAs) were measured by a gas
171 chromatograph (Agilent 4890, J&W Scientific, USA) equipped with a flame
172 ionization detector (Zhang et al., 2018). A gas chromatograph (GC) (Agilent, 4890,
173 J&W Scientific, USA) was utilized to analyze gas concentrations (including H_2 and
174 CH_4) through a thermal conductivity detector.

175 **3. Results and discussion**

176 *3.1. Feasibility of Se(VI) bioreduction driven by S(0)*

177 Fig. 1a shows that a gradual, progressive decrease in Se(VI) of the biotic group
178 occurred during the batch trial. Removal efficiency of Se(VI) reached 92.4 ± 0.7% by
179 36 h. Hardly any Se(VI) concentration was detected at the end of a 72 h cycle
180 indicating almost complete removal of Se(VI). The mean reduction rate of Se(VI) was
181 1.1 ± 0.08 mg/L·h, corresponding to a pseudo first-order reaction kinetics process
182 with a kinetics rate constant of 0.061 d^{-1} and R^2 of 0.99 (Fig. 1b). Temporal dynamics
183 of Se(VI) in the control groups was also examined comparatively (Fig. S1, Supporting
184 Information). The highest removal efficiency of Se(VI) in the sterilized group was
185 only 18.1 ± 0.2%, much lower than that in the biotic group, suggesting Se(VI)
186 reduction was microbially mediated. A limited Se(VI) removal efficiency of 19.1 ±
187 0.2% was obtained after one cycle in the abiotic group, implying that S(0) made
188 negligible contribution to abiotic reduction of Se(VI). Taken the aforementioned into
189 consideration, it appears quite feasible to use S(0) as an electron donor to bioreduce
190 and biodetoxify Se(VI).

191 *3.2. Long-term Se(VI) removal dynamics*

192 Efficient Se(VI) elimination was observed in the column trial during 258-d
193 operation (Fig. 1c). During Stage 1 (Days 0-94), with 10 mg/L of Se(VI) initial
194 concentration and 0.56 mL/min constant flow rate, Se(VI) was completely removed and
195 the corresponding Se(VI) removal capacity was 41.5 ± 4.1 ($g/m^3 \cdot d$) (Table 1). No
196 accumulation of Se(IV) was observed during this phase. The Se(VI) removal efficiency
197 decreased slightly to 94.1 ± 6.2% when the initial concentration of Se(VI) was raised
198 to 50 mg/L during Stage 2 (Days 95-135), accompanied by an increase in Se(VI)
199 removal capacity to 193.0 ± 11.9 ($g/m^3 \cdot d$). Se(VI) removal efficiency decreased to 84.2
200 ± 5.1% and removal capacity rose to 346.8 ± 30.2 ($g/m^3 \cdot d$) when the initial
201 concentration of Se(VI) was further increased to 100 mg/L during Stage 3 (Days 136-
202 176). This implied that functional microorganisms had gradually adapted to an
203 environment characterized by high Se(VI) loading. Notably, accumulation of Se(IV)
204 occurred for a mean concentration of 9.8 ± 15.2 mg/L, followed by almost sudden
205 disappearance of accumulated Se(IV) at the end of Stage 3 (Fig. 1c). This phenomenon
206 was similar to a previous study where Se(IV) accumulation was observed in Se(VI)
207 reduction using bacterium *Bacillus sp*. SF-1 with Se(IV) removal subsequently
208 occurring (Kashiwa et al., 2000). In Stage 4 (Days 177-217), the flow rate was adjusted
209 to 1.68 ml/min with Se(VI) concentration returned to 10 mg/L. During this stage, Se(VI)
210 removal efficiency increased to 89.7 ± 10.3% but the removal capacity fell to 117.0 ±
211 12.7 ($g/m^3 \cdot d$). In Stage 5 (Days 218-258), during which 10 mg/L NO_3^--N was
212 introduced as co-acceptor to the bioreactor, the Se(VI) removal efficiency dropped to

213 82.8 ± 3.7%, suggesting that NO_3^--N was the preferred electron acceptor used by the
214 microorganisms for metabolism, as also reported by He et al. (He et al., 2021). In a
215 previous study, Lucas and Hollibaugh showed that Se reduction was depressed by NO_3^-
216 -N when both co-existed at equal levels owing to physiological, kinetic, or enzymatic
217 factors involved in the NO_3^--N and Se(VI) reductases (Oremland et al., 1999; Lucas
218 and Hollibaugh, 2001).

219 The evolution of pH and conductivity during long-term study were monitored.
220 Changes in pH were in the range of 7.6 to 8.4 during the Se(VI) bioreduction process
221 (Fig. S2, Supporting Information), with conductivity varying from 3000 to 3700 μS/cm.
222 This ambient condition was suitable for microbial activity to achieve diverse
223 bioprocesses (Zhang et al., 2015).

224 *3.3. Bioprocess description*

225 By the end of the experiment, the biological column had become dark red (Fig.
226 S3, Supporting Information), indicating precipitation of solid Se(0) from the Se(VI)
227 bioreduction process. The speciation percentages of selenium in the effluent were not
228 entirely dominated by Se(VI) and Se(IV) (Fig. 2a), suggesting generation of
229 unaccounted dissolved Se and/or organic selenium (Tan et al., 2018; Xia et al., 2019).
230 Residual Se(VI) and Se(IV) could be further removed by adsorption or co-
231 precipitation (Gezer et al., 2011; Li et al., 2018). To provide further confirmation of
232 the product morphology, characterization analyses were conducted for the precipitate.
233 In Fig. 2b, the SEM image shows spheroids observed on the surface of the biomass.
234 EDS analysis revealed Se(0) was present in the spheroids, with a corresponding

spectral peak (Fig. S4, Supporting Information). The 56.1 eV peak signal obtained from XPS spectra relates to the 3d orbit of Se (Fig. 2c), attributed to Se(0) (Li et al., 2014). XRD demonstrated the presence of elemental Se with various crystal planes (Fig. 2d). These results collectively confirm that Se(0) was the final product in the Se(VI) bioreduction process.

SO_4^{2-} accumulated in the biosystem as a result of S(0) oxidation (Fig. 1c), indicating Se(VI) bioreduction coupled with oxidation of S(0) following Eq. (1). It should be noted that S(0) has previously been reported to act as an electron donor in the reduction of other heavy metals, accompanied by SO_4^{2-} production (Peng et al., 2016; Shi et al., 2019). SO_3^{2-} and $S_2O_3^{2-}$, the intermediates related to S(0) oxidation, were hardly detected in the present system. Moreover, hardly any NO_3^--N, NO_2^--N, and NH_4^+-N were detected at the end of Stage 5 (Fig. S5, Supporting Information), implying complete denitrification according to Eq. (2) (Zhang et al., 2015).

$$2SeO_4^{2-} + 12S^0 + 15HCO_3^- + 3NH_4^+ + H_2O \rightarrow 3C_5H_7NO_2 + 2Se^0 + 12SO_4^{2-} + 8H^+ \quad (1)$$

$$55S^0 + 20CO_2 + 50NO_3^- + 4NH_4^+ + 38H_2O \rightarrow 4C_5H_7NO_2 + 55SO_4^{2-} + 25N_2 + 64H^+ \quad (2)$$

The mass balance is calculated for Stage 4, as an example. During operation, 0.49 ± 0.05 mmol Se(VI) was removed, leading to theoretical production of 2.94 ± 0.29 mmol SO_4^{2-} according to Eq. (1). In fact, 5.15 ± 0.41 mmol SO_4^{2-} was detected, implying over-consumption of S(0), which might be ascribed to other bioactivities, such as biomass growth and the accumulation of metabolites (Shi et al., 2019).

257 *3.4. Microbial community evolution*

258 The evolution process of microbial community structure was examined through
259 16S rRNA gene sequencing for both inoculum and treated samples. OUTs in the
260 bioreactor were lower than inoculum, indicating that the microbial community has a
261 procedure for selection and adaptation (Table S2, Supporting Information). The
262 decreasing total number of species (Sobs indexes) in treated samples reflected
263 reduced community richness, which was also indicated by the Chao1 and Ace
264 indexes. Perfect community diversity was confirmed by the lower Shannon and higher
265 Simpson indexes in Stages 1-5, suggesting highly selected microbial communities
266 were received (Table S2, Supporting Information). The coverage index was invariably
267 close to unity, suggesting that almost all sequences were covered. Smooth rarefaction
268 curves were obtained (Fig. S6, Supporting Information), confirming that the
269 sequencing sample numbers were sufficient.

270 NMDS analysis showed that compared to other stages, biomass in Stage 1 was
271 similar to the inoculum, whereas biomass from Stage 4 and Stage 5 were quite distinct
272 from the inoculum (Fig. 3a). The microbial community composition cultivated in
273 Stage 2 and Stage 3 were similar, which may be attributed to increased Se(VI) loading
274 in Stage 2 that subsequently affected microorganisms in Stage 3. The Venn diagram
275 (Fig. 3b) shows that 906, 188, 31, 188, and 449 of OTUs were found separately in
276 five treated stages, whereas a total of 722 OTUs were common to all cultivated stages.
277 The presence of OTUs shared by all groups implies that such OTUs had adapted
278 simultaneously to the situations of Se(VI) bioreduction, S(0) oxidation, and NO_3^--N

removal.

Significant changes occurred in the microbial community structure of biomass from the bioreactor, in comparison with the inoculum (Fig. 3c). The class related to *Gammaproteobacteria* increased substantially through Stages 1 to 5, and it became the predominant species (75.2%) in Stage 3. Known as NO_3^--N reducing bacteria, *Gammaproteobacteria* was previously found to be the majority class in an S(0)-based Cr(VI) bioreduction bioreactor (Shi et al., 2019). The *Deltaproteobacterian* species increased in Stages 1 (11.1%), 2 (23.2%), 4 (34.9%), and 5 (11.3%) compared to the inoculum (8.9%), but decreased in Stage 3 (7.62%). In Stage 4 especially, *Deltaproteobacterian* accounted for the majority of species present. Recently, Tan et al. reported that *Deltaproteobacterian* was the main community observed in their study of selenium pollution wastewater microbial treatment (Tan et al., 2018). Declines in both *Alphaproteobacteria* and *Actinobacteria* were observed through Stages 1 to 5. *Anaerolineae* were found to be present at all treated stages. This is consistent with a previous finding that bacteria related to the *Anaerolineae* family have the capacity to reduce Se(VI) owing to the presence of Se(VI) reductases in *Anaerolineae* (Fakra et al., 2018).

To develop a more profound understanding of the alteration in microbial community structure, we also analyzed the relative abundance of key genera (Fig. 3d), and found that the relative abundance of *Thiobacillus* increased considerably during the experiment. Compared to its 0.83% abundance in the inoculum, *Thiobacillus* accounted for 4.86% in Stage 1. The relative abundance of *Thiobacillus* rose to 36.8%

301 in Stage 2 and even higher to 72.5% in Stage 3 with elevated Se(VI) loading, and so
302 *Thiobacillus* became the most dominant species. Although its abundance later
303 declined, the proportion of *Thiobacillus* was 13.8% in Stage 4 and 7.31% in Stage 5.
304 As a chemoautotrophic genus, *Thiobacillus* could oxidize S(0) to release electrons and
305 synthesize metabolites, realizing the removal of oxidized contaminants (Yang et al.,
306 2010; He et al., 2021). Therefore, *Thiobacillus* was believed to promote S(0)
307 oxidation and Se(VI) reduction independently. Known as a sulfur-oxidizing
308 bacterium, *Thiobacillus* is widely distributed in natural water, mud and soil
309 (Fjerdingstad, 1969; Zhai et al., 2016), and thus could be utilized for the
310 bioremediation of contaminants in practice, with high microbiological safety. Species
311 related to *Desulfurivibrio* increased from Stages 1 to 5, with 20.8% abundance
312 reached in Stage 4. *Desulfurivibrio* has demonstrable capability of reducing Se(VI)
313 and Se(IV) to Se(0) without growth or respiration, using organic electron donors
314 (Tomei et al., 1995; Lucas and Hollibaugh, 2001). And NO_3^--N can act as the terminal
315 electron accepter of *Desulfurivibrio* in NO_3^--N dissimilatory reduction (Keith and
316 Herbert, 1983). Other SO_4^{2-} reducing bacteria such as *Desulfococcus* and
317 *Desulfocapsa* were detected during operation, which might be related to Se(VI)
318 reduction and sulfur circulation (Qian et al., 2015; He et al., 2021). The relative
319 abundance of *Pseudomonas* first declined from Stages 1 to 3, then increased in Stages
320 4 and 5 with proportions of 4.04% and 19.6%, respectively. Previously, reduction of
321 Se(VI) to Se(0) was achieved using a bacterial strain belonging to *Pseudomonas*,
322 through anaerobic respiration (Kuroda et al., 2011; Subedi et al., 2017). Although

323 almost none was detectable in inoculum, *Sulfuricurvum* was present in Stages 1 to 5
324 with respective percentages of 2.71%, 2.63%, 0.40%, 3.98% and 0.66%. As a sulfur
325 oxidizer, *Sulfuricurvum* is known to extract energy from S(0) oxidation when
326 synthesizing organic metabolites from an inorganic carbon source (Ontiveros-
327 Valencia et al., 2016). *Sulfuricurvum* has also been reported capable of ClO_4^-
328 reduction using S(0) as the electron donor (Sahu et al., 2009). *Clostridium* and
329 *Methanobacterium* genera were also detected, with relatively lower abundances; these
330 genera could produce H_2 and CH_4 through their metabolisms (Kamalaskar et al.,
331 2016; He et al., 2021).

332 *3.5. Metabolic pathway interpretation*

333 Se(VI) was firstly reduced to Se(IV) through specialized selenate reductase
334 (Staicu and Barton, 2021), followed by subsequent reduction of Se(IV) to Se(0) by
335 nitrite reductase, sulfite reductase, or hydrogenase I (Harrison et al., 1984; Yanke et
336 al., 1995; Basaglia et al., 2007). Fig. 4a shows that both *serA* and *tatC* genes were
337 greatly enriched during Stage 1-5 despite of their concentrations were below limit of
338 detection in the inoculum. Located in the periplasmic space, *serA* encoded specific
339 reductase for Se(VI) bioreduction (Schröder et al., 1997; Wen et al., 2016). *tatC* was
340 reported to encode specific enzyme capable of reducing Se(VI) (Ma et al., 2007).
341 About twenty years ago, enzyme encoded by gene *napA* was observed to reduce
342 Se(VI) (Sabaty et al., 2001), whereas it increased in the bioreactor rather than the
343 inoculum in the present study. Herein, the abundance of *nirS* gene increased during
344 Stages 1-5. Se(IV) reduction to Se(0) was attributed to the *nirS* gene because of its

345 ability to reduce Se(IV) (DeMoll-Decker and Macy, 1993); this was consistent with
346 the previously mentioned finding that there was no accumulation of Se(IV) in the
347 bioreactor. Studies also showed that nitrate/nitrite reductase genes were able to
348 accomplish the microbial reduction of vanadate (Zhang et al., 2019; He et al., 2021).
349 Encoding enzymes, gene *soxB*, related to the oxidation of S(0) to SO_4^{2-} exist at all
350 stages during operation, suggesting efficient S(0) oxidation during Se(VI)
351 bioreduction.
352 Electron transfer compounds, such as Cyt c and NADH, were found in the S(0)-
353 driven Se(VI) bioreduction system (Fig. 4b), suggesting that electron transfer
354 occurred during the S(0) oxidation and Se(VI) bioreduction bioactivities. This
355 phenomenon of electron transfer during Se(VI) biodetoxification has also been
356 reported previously (Li et al., 2021). Metabolites, including VFAs, H_2 and CH_4, were
357 also detected in the constructed biosystem (Fig. 4b). It is likely that the autotrophs
358 utilized either carbon dioxide or bicarbonate as the sole carbon source to synthesize
359 VFAs during bioreduction, with energy gained from S(0) oxidation (Estelmann et al.,
360 2011; Zhang et al., 2020). Meanwhile, VFAs have been commonly detected when
361 using H_2 and CH_4 to reduce Se(VI) (Lai et al., 2016; Ontiveros-Valencia et al., 2016).
362 VFAs, generated as the mediate product, can be utilized by heterotrophic
363 microorganisms for reduction of high-valence pollutants (Lu et al., 2020). Notably, H_2
364 and CH_4 were also detected during operation, indicating these two gases were
365 involved in the formation and/or transformation of VFAs during Se(VI) bioreduction.
366 Furthermore, these gaseous intermediate metabolites could also be utilized by Se(VI)

367 reducers either directly or indirectly (Lai et al., 2016; Ontiveros-Valencia et al.,
368 2016).

369 *3.6. Mechanism summary and practical implication*

370 Mechanisms of Se(VI) reduction linked to S(0) oxidation can now be
371 summarized based on the evidence obtained from the present tests (Fig. 5).
372 Independent and synergic pathways existed for Se(VI) reduction. For the former
373 pathway, S(0) oxidation and Se(VI) reduction could be achieved by a single
374 chemoautotrophic genus (e.g., *Thiobacillus*). For the latter pathway, S(0) oxidizers
375 such as *Sulfuricurvum* converted HCO_3^- to intermediate metabolites (e.g., VFAs, H_2,
376 CH_4) as a result of S(0) oxidation, which were utilized by heterotrophic Se(VI)
377 reducers (e.g., *Desulfurivibrio* and *Pseudomonas*). In general, dissimilatory reduction
378 and detoxification of selenium are the main mechanisms behind Se(VI) reduction
379 (Eswayah et al., 2016). Se(VI) bioreduction involved initial reduction of Se(VI) to
380 Se(IV), followed by further reduction of Se(IV) to Se(0). Genes were detected that
381 encoded specialized reductase for Se(VI) reduction to Se(IV) (i.e., *serA* and *tatC*),
382 Se(IV) reduction to Se(0) (i.e., *nirS*), and S(0) oxidation to SO_4^{2-} (i.e., *soxB*). Electron
383 transfer by Cyt c and NADH was verified for the Se(VI) bioreduction process.

384 Selenium pollution is of increasing concern due to adverse environmental impact
385 of Se(VI). Bioremediation has been proposed as a favourable method to biotically
386 reduce Se(VI). S(0) is readily available as a byproduct from oil refining. To our
387 knowledge, this study is the first to report on bioreduction of Se(VI) utilizing S(0) as
388 an electron donor, which might be a promising option for in situ bioremediation of

Se(VI) contaminated aquifers. Our finding elucidated the removal pathway where Se(VI) bioreduction can be coupled to the oxidation of S(0). In practical application, the proposed bioprocesses can be implemented with an S(0)-packed biological permeable reactive barrier installed perpendicular to the migration direction of a groundwater plume (Gibert et al., 2011). Further research can be performed to explore geological influences on the bioreduction process. Optimization of S(0) dosage is required and the effect of coexisting pollutants in aquifers, such as V(V) and SO_4^{2-}, should be considered (Du et al., 2020).

4. Conclusions

This study investigated the chemoautotrophic transformation of Se(VI) driven by S(0) in the simulated aquifer groundwater. Batch trial indicated the feasibility of Se(VI) bioreduction utilizing S(0) as an electron donor under anaerobic conditions, with Se(VI) removal efficiency of 92.4 ± 0.69% in 36 h operation. A 258-d continuous experiment showed that Se(VI) removal performance varied under different hydrodynamic and hydrochemical conditions in a column reactor. Se(VI) was reduced to insoluble Se(0) and S(0) was transformed to SO_4^{2-}. Detailed mechanisms of Se(VI) bioreduction driven by S(0) were proposed through analysis of microbial communities, functional genes, and intermediate metabolites.

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651

652 **Figure Captions.**

653 **Fig. 1.** Se(VI) removal performance in S(0)-driven Se(VI) reduction biosystems. (a)
654 Time histories of Se(VI) removal efficiency during three consecutive operating cycles
655 during batch trial; (b) Pseudo first-order kinetic plots for Se(VI) removal; (c) Time
656 histories of Se(VI), NO_3^--N in influent, Se(VI), Se(IV), NO_3^--N, and SO_4^{2-} in effluent,
657 corresponding Se(VI) and NO_3^--N removal efficiencies, and Se(VI) removal capacities
658 during 258 d operation of the column trial.

659 **Fig. 2.** Characterization of products from Se(VI) bioreduction. (a) Selenium speciation
660 percentages in the effluent. (b) SEM, (c) XPS, and (d) XRD analysis of the precipitates
661 after reaction. SEM: Scanning electron microscope; XPS: X-ray photoelectron
662 spectroscopy; XRD: X-ray diffraction.

663 **Fig. 3.** Microbial community evolution in inoculum and at different stages of column
664 trial. (a) Non-metric multidimensional scaling (NMDS) plot for the distribution of
665 microbial community; (b) Venn plot of microbial richness (b); (c) Class-level relative
666 abundance; (d) Heatmap of functional genera.

667 **Fig. 4.** Metabolic processes of S(0)-driven Se(VI) bioreduction. (a) Abundances of
668 functional genes; (b) Contents of electron transfer compounds and intermediate
669 metabolites. Cyt c: Cytochrome c; NADH: Nicotinamide adenine dinucleotide; VFAs:
670 Volatile fatty acids.

671 **Fig. 5.** Mechanistic pathways of microbial Se(VI) reduction processes linked to S(0)
672 oxidation. VFAs: Volatile fatty acids.

673

Table 1. Ambient conditions and Se(VI) removal performance for each stage during 258-d operation in column trial.

Stage	Period (d)	Flow rate (mL/min)	Initial Se(VI) (mg/L)	Initial NO_3^--N (mg/L)	Se(VI) removal efficiency (%)	Se(VI) removal capacity (g/(m^3/d))	NO_3^--N removal efficiency (%)	NO_3^--N removal capacity (g/(m^3/d))
1	0-94	0.56	10	-	98.1 ± 6.3	41.5 ± 4.1	-	-
2	95-135	0.56	50	-	94.1 ± 6.2	193.0 ± 11.9	-	-
3	136-176	0.56	100	-	84.2 ± 5.1	346.8 ± 30.2	-	-
4	177-217	1.68	10	-	89.7 ± 10.3	117.0 ± 12.7	-	-
5	218-258	0.56	10	10	82.8 ± 3.7	35.2 ± 2.1	97.9 ± 0.8	40.6 ± 3.3

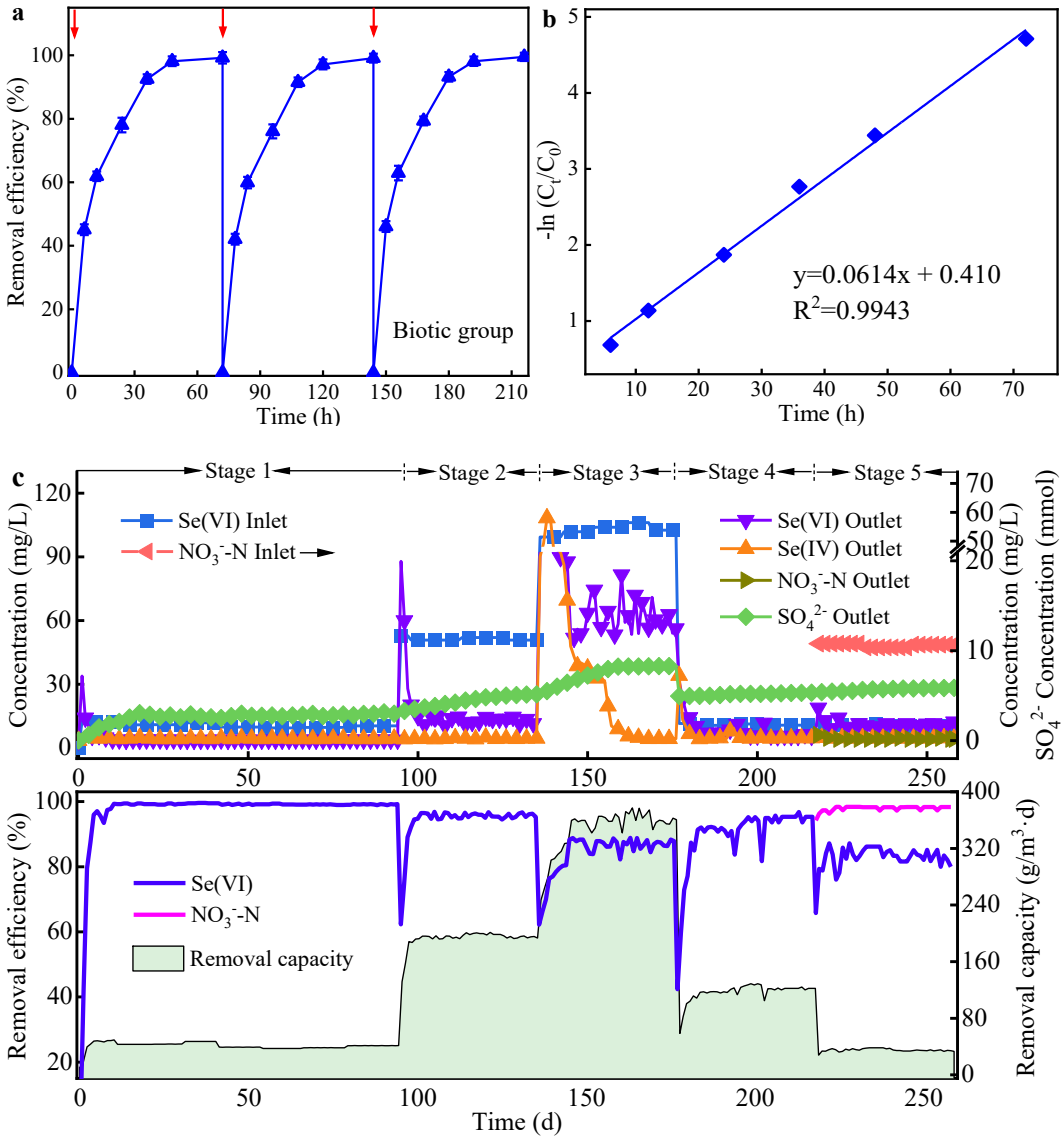

Fig. 1. Se(VI) removal performance in S(0)-driven Se(VI) reduction biosystems. (a) Time histories of Se(VI) removal efficiency during three consecutive operating cycles during batch trial; (b) Pseudo first-order kinetic plots for Se(VI) removal; (c) Time histories of Se(VI), NO_3^--N in influent, Se(VI), Se(IV), NO_3^--N, and SO_4^{2-} in effluent, corresponding Se(VI) and NO_3^--N removal efficiencies, and Se(VI) removal capacities during 258 d operation of the column trial.

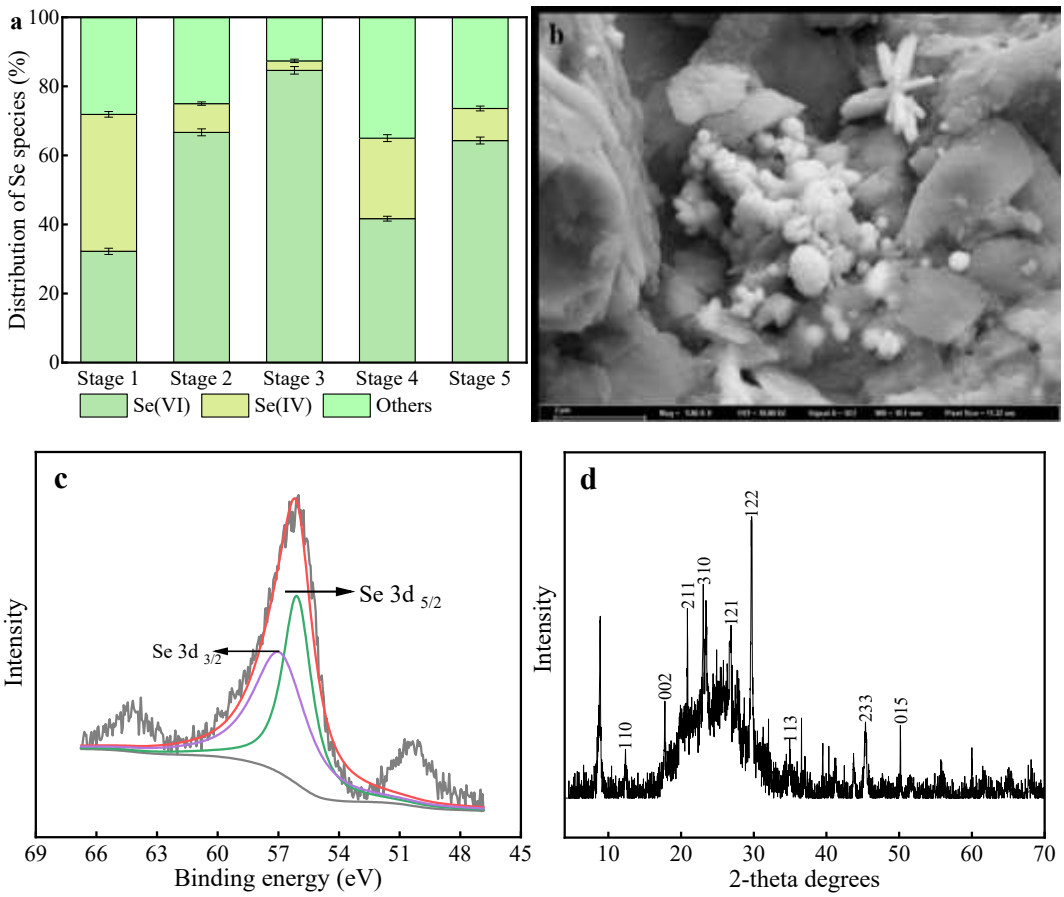

Fig. 2. Characterization of products from Se(VI) bioreduction. (a) Selenium speciation percentages in the effluent. (b) SEM, (c) XPS, and (d) XRD analysis of the precipitates after reaction. SEM: Scanning electron microscope; XPS: X-ray photoelectron spectroscopy; XRD: X-ray diffraction.

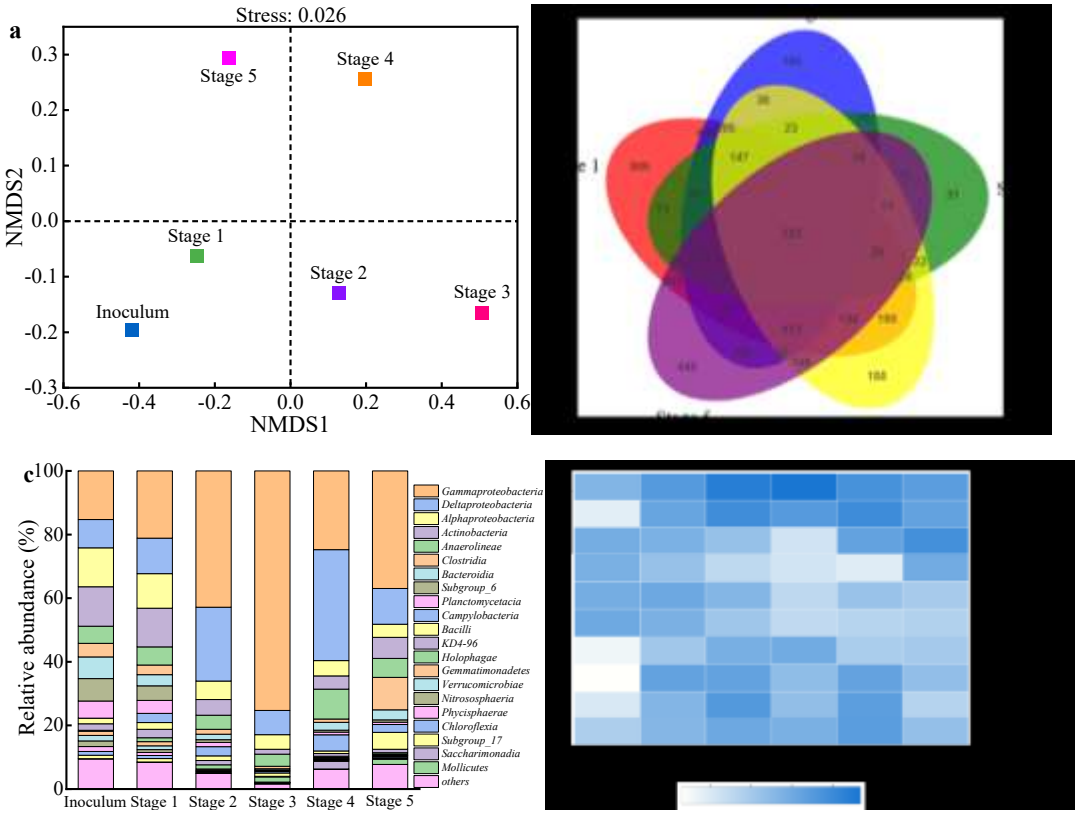

Fig. 3. Microbial community evolution in inoculum and at different stages of column trial. (a) Non-metric multidimensional scaling (NMDS) plot for the distribution of microbial community; (b) Venn plot of microbial richness (b); (c) Class-level relative abundance; (d) Heatmap of functional genera.

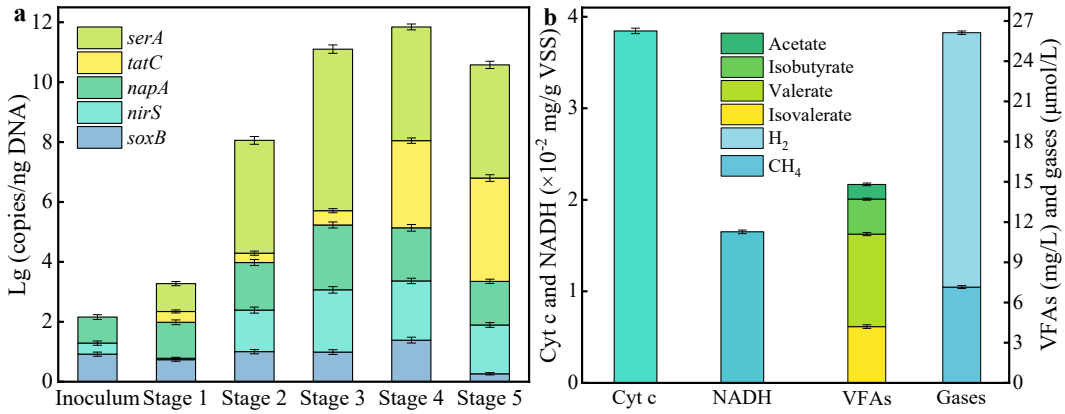

Fig. 4. Metabolic processes of S(0)-driven Se(VI) bioreduction. (a) Abundances of functional genes; (b) Contents of electron transfer compounds and intermediate metabolites. Cyt c: Cytochrome c; NADH: Nicotinamide adenine dinucleotide; VFAs: Volatile fatty acids.

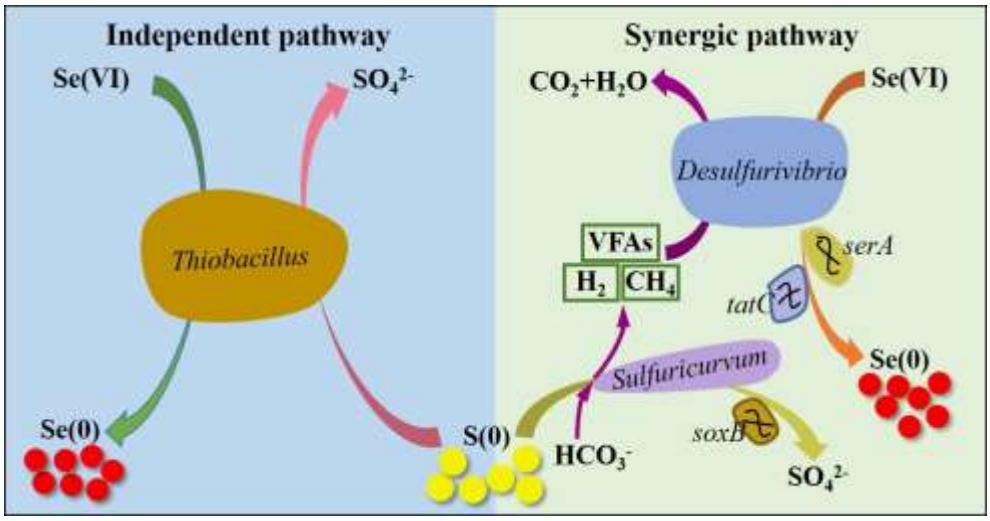

Fig. 5. Mechanistic pathways of microbial Se(VI) reduction processes linked to S(0) oxidation. VFAs: Volatile fatty acids.