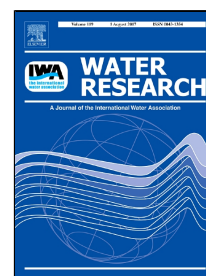


# Accepted Manuscript

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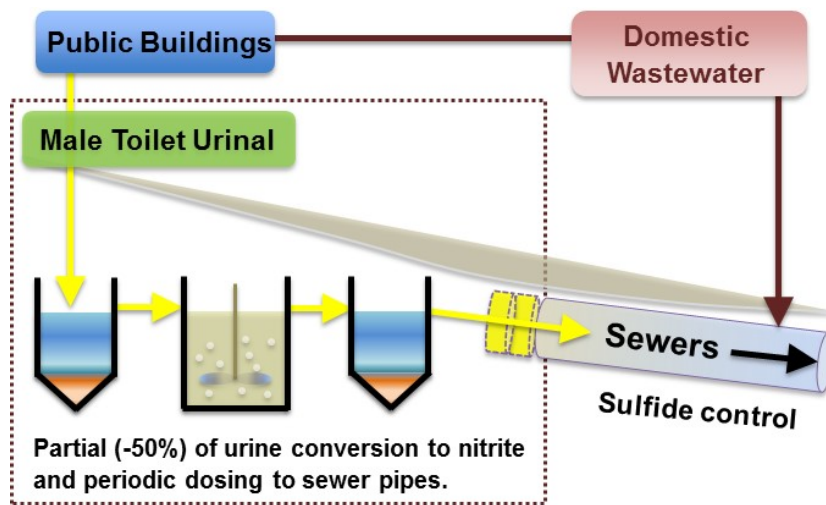


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## Graphic for manuscript



## Abstract graphic

# 1 Nitrite production from urine for sulfide control in sewers

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13

14 **ABSTRACT**

15 Most commonly used methods for sewer sulfide control involves dosing chemical agents to  
16 wastewater, which incurs high operational costs. Here, we propose and demonstrate a cost-  
17 effective and environmentally attractive approach to sewer sulfide control through urine  
18 separation and its subsequent conversion to nitrite prior to intermittent dosage to sewers. Urine  
19 collected from a male toilet urinal was fed to laboratory-scale sequencing batch reactors. The  
20 reactors stably converted roughly 50% of the nitrogen in urine to nitrite, with high abundance  
21 (at 17.46%) of known ammonia-oxidizing bacteria (AOB) of the genus *Nitrosomonas*, and  
22 absence (below detection level) of typical nitrite-oxidizing bacteria of the genus *Nitrospira*,  
23 according to 454 pyrosequencing analysis. The stable nitrite production was achieved at both  
24 relatively high (1.0–2.0 mg/L) and low (0.2–0.3 mg/L) dissolved oxygen concentrations.  
25 Dosing tests in laboratory-scale sewer systems confirmed the sulfide control effectiveness of  
26 free nitrous acid generated from urine. Life cycle assessment indicated that, compared with  
27 commodity chemicals, nitrite/free nitrous acid (FNA) production from urine for sulfide control  
28 in sewers would lower the operational costs by approximately 2/3 and greenhouse gas (GHG)  
29 emissions by 1/3 in 20 years.

30 **Keywords:** Urine separation; Free nitrous acid (FNA); Sulfide control; Life cycle assessment;  
31 Economic analysis

32

## 33 1. Introduction

34 In urban water management, all pollutants, including ammonium, phosphorus, sulfate, and  
35 organics, are collected from households and industries, and then delivered through sewer  
36 networks to a treatment plant for removal. Sulfate is easily reduced anaerobically in sewers,  
37 resulting in production of sulfide (Liu *et al.*, 2014a). The generation of sulfide in sewer systems  
38 is a serious problem because it induces odor, concrete and metal corrosion and health hazards  
39 (Pikaar *et al.*, 2014). Subsequent rehabilitation and replacement of the damaged sewer pipes  
40 are rather expensive. It has been reported that national sewer rehabilitation in the US alone  
41 could cost up to 3.2 billion dollars annually (US Environmental Protection Agency, 2010). To  
42 date, one of the main approaches adopted to mitigate these issues has been the dosage of  
43 chemical agents, such as oxygen, iron salts, nitrate and nitrite salts, and alkalis (Ganigue *et al.*,  
44 2011; Liu *et al.*, 2014b). However, continuous dosing of these commodity chemicals incurs  
45 large operational costs.

46 Recently, the concept of integrated urban water management is receiving more attention  
47 because it can deliver system-wide optimization with tremendous economic and environmental  
48 benefits (Behzadian and Kapelan, 2015). For example, researchers proposed using iron  
49 chloride to replace commonly used alum as a coagulant in drinking water treatment and then  
50 using the same iron for corrosion and odor control in sewers and phosphorus removal in  
51 wastewater treatment plants, thus substantially reducing the use of chemicals in the entire urban  
52 water system (Gutierrez *et al.*, 2010; Ge *et al.*, 2012; Sun *et al.*, 2015a). In this work, we  
53 investigate the feasibility of sulfide control in sewers using urine. This would involve urine

54 separation, its conversion to nitrite and the subsequent, intermittent addition of the nitrite to  
55 sewers. The nitrite addition would normally need to be accompanied by acid addition to  
56 produce free nitrous acid (FNA) at elevated (sub-ppm) levels. FNA was previously discovered  
57 to be a cost-effective agent for sulfide control in sewers (Jiang *et al.*, 2013). It has a biocidal  
58 effect in sewer biofilms and can also lead to biofilm removal (Jiang *et al.*, 2010; Jiang *et al.*,  
59 2011a; Jiang *et al.*, 2011b; Jiang *et al.*, 2013; Jiang and Yuan, 2014; Sun *et al.*, 2015b). The  
60 source separation approach proposed here is essentially to produce nitrite required from  
61 wastewater itself.

62 Source separation of urine from the remaining household wastewater is a promising  
63 approach toward sustainable urban water management (Maurer *et al.*, 2006; Lienert and Larsen,  
64 2010; Udert and Wächter, 2012). Urine contains up to 80% of the total nitrogen in domestic  
65 wastewater but makes up only less than 1% of the total volume (Larsen and Gujer, 1996; Udert  
66 *et al.*, 2006); therefore, source separation of urine can substantially reduce the nitrogen load to  
67 the downstream wastewater treatment plants. Many methods had been proposed for nutrient  
68 recovery from urine by using physical, chemical and biological methods (Maurer *et al.*, 2006;  
69 Zhang *et al.*, 2014; Xu *et al.*, 2016). However, to the best of our knowledge, no information  
70 has yet been reported on sulfide control in sewers by using source-separated urine.

71 In this study, we experimentally demonstrate nitrite production from urine and its  
72 effectiveness in sulfide control in sewers. Urine was collected from a male toilet urinal in a  
73 public building. Nitrite was then produced using laboratory-scale sequencing batch reactors  
74 with urine as the feed. The effectiveness of the urine-generated nitrite in suppressing sulfide

75 production in sewers was then demonstrated using a laboratory-scale sewer reactor.  
76 Additionally, life cycle assessment (LCA) was conducted to assess the economic and  
77 environmental performance of the proposed approach.

## 78 **2. Materials and methods**

### 79 *2.1 Urine collection and characteristics*

80 Urine was collected from a male toilet urinal in the Sino-Italian Environment and Energy-  
81 efficient Building located in Tsinghua University, Beijing. The collected urine, which was  
82 stored in a large container for over three days, was then used as the feed to a nitrifying reactor.  
83 The feed contained approximately 400–900 mg N/L ammonia nitrogen, 450–950 mg N/L total  
84 nitrogen, 300–700 mg/L COD, and 1500–2800 mg/L alkalinity as CaCO<sub>3</sub> with a pH of 8.8±0.1.  
85 The nitrogen concentrations were much lower compared with fresh urine because of the  
86 dilution with flushing water during collection.

### 87 *2.2 Reactor setup and operation*

88 Experiments on the production of FNA were performed in laboratory-scale sequencing batch  
89 reactor. The reactors were made from Plexiglas cylinders with an effective volume of 2.7 L (30  
90 cm in height and 12 cm in inner diameter). Each 8 h cycle consisted of period of feeding (5  
91 min), mixing and aeration (7 h), settling (50 min), and decanting (5 min). The hydraulic  
92 retention time (HRT) and sludge retention time (SRT) of the reactor were 24 h and 40 d,  
93 respectively.

94 The experiments included three phases: I, start-up of a reactor and long-term operation with  
95 ammonia nitrogen concentration of ~550 mg/L as the influent for achievement of stable nitrite

96 production from urine; II, operation of two reactors in parallel under different dissolved oxygen  
97 (DO) conditions with influent ammonia nitrogen concentration fluctuation to investigate the  
98 potential impact of DO on the ammonium to nitrite conversion; and III, operation of a low DO  
99 reactor to investigate the effect of alkalinity supplement in the urine on the reactor performance.

100 The DO concentration was monitored continuously using a DO probe (WTW, Oxi340i), and  
101 controlled at set-points of 1.0–2.0 mg/L and 0.2–0.3 mg/L in the high and low DO reactors,  
102 respectively. Temperature and pH were automatically recorded using a pH meter (WTW,  
103 pH340i). Both reactor was inoculated with nitrification sludge from a full-scale municipal  
104 wastewater treatment plant on the campus of Tsinghua University (Tsinghua Water Reuse,  
105 Beijing) in both Phase I and Phase II. The seed sludge in the Phase III was from a laboratory  
106 nitrification reactor. The ammonium, nitrite and nitrate concentrations in the effluent were  
107 measured 1–4 times every week. The nitrite accumulation ratio in the effluent was calculated  
108 as the percentage of nitrite nitrogen in the total nitrate and nitrite nitrogen, as described  
109 previously (Zheng *et al.*, 2013a). Cycle studies were conducted by measuring the ammonium,  
110 nitrite and nitrate concentrations throughout a cycle. In Phase I, microbial communities in the  
111 inoculated sludge and in the reactor sludge were analyzed using 454 pyrosequencing, as  
112 described in 2.4.

### 113 *2.3 Batch tests to determine sulfide control effectiveness*

114 Batch tests were carried out in laboratory reactors to determine the effectiveness of the  
115 effluent from the above nitrifying reactors in controlling sulfide production by sewer  
116 sediments. Nine reactors were used in the study. Among them, one was used as control. Each



117 reactor had an effective volume of 1 L and was made from a Plexiglas cylinder with a diameter  
118 of 80 mm and a height of 200 mm. Sediments were collected from a mature gravity sanitary  
119 sewer receiving domestic wastewater in Tsinghua University, Beijing, using a shovel. The bulk  
120 density of the sewer sediment was  $1.46 \text{ g/cm}^3$ . The total and volatile solids contents were 0.19  
121 g/mL and 0.14 g/mL, respectively. The collected sediments were exposed for 12 h to different  
122 nitrite concentrations (50, 100, 150 and 200 mg N/L) at a pH of 6.5 (equal to FNA  
123 concentrations of 0.036, 0.072, 0.108, and 1.144 mg  $\text{HNO}_2\text{-N/L}$ , respectively). Nitrite was  
124 prepared from the treated urine in four groups and  $\text{NaNO}_2$  in other four groups. Then, 100 mL  
125 of the exposed sediments and 900 mL of deoxygenated water with organics and sulfate were  
126 added into the reactors with no headspace. A shaker with a rotating speed at 30 r/min was used  
127 to keep a slight disturbance on the sediment. After 12 h, the produced sulfide concentration in  
128 the reactors was measured, and used for calculating the sediment sulfide production rate. The  
129 control test was repeated six times to obtain the baseline sulfide production rate, as  $1.02 \pm 0.07$   
130  $\text{mg S}/(\text{d} \cdot \text{m}^2)$ .

#### 131 *2.4 Analytical methods*

132 Measurements of COD,  $\text{NH}_4^+\text{-N}$ ,  $\text{NO}_2^-\text{-N}$ ,  $\text{NO}_3^-\text{-N}$  and sulfide concentrations in the reactor  
133 liquid phase were performed in accordance with standard methods (Ministry of Environmental  
134 Protection, 2006). Free ammonia (FA) and FNA concentrations were calculated as described  
135 by Anthonisen *et al.* (1976), based on the measured ammonium and nitrite concentrations,  
136 respectively, along with the pH and temperature levels.

137 For DNA extraction and 454 pyrosequencing, the sludge samples were collected and

138 concentrated by centrifugation at 5000 rpm for 4 min. Then, the total DNA was extracted from  
139 each prepared sample using a Fast DNA Spin Kit for Soil (MP Biomedicals, LLC, Solon, OH,  
140 USA). The OD<sub>260</sub>/OD<sub>280</sub> and OD<sub>260</sub>/OD<sub>230</sub> values were determined on a NanoDrop 2000  
141 to evaluate the purity and quantity of each DNA sample. For each DNA sample, the V1-V3  
142 region of the 16S ribosomal RNA gene was amplified using the primers 27F and 533R. The  
143 amplified amplicons were detected by 454 pyrosequencing, and the sequenced data were  
144 processed using Quantitative Insights Into Microbial Ecology (QIIME) pipeline with default  
145 settings. The operational taxonomic units (OTUs) with a 97% sequence density threshold were  
146 clustered using UCLUST software. Sequences that were associated with one genus divided by  
147 total sequences were defined as the relative abundance of that genus in the sludge sample.

#### 148 *2.5 LCA and economic analysis*

149 To determine the practical feasibility of the proposed approach for nitrite production from  
150 urine, the capital cost, operational cost, and environmental impacts such as greenhouse gas  
151 (GHG) emissions and energy inputs must be assessed and compared with the industrial  
152 production of the chemical. In this study, we used the experimental results and then employed  
153 LCA methodology to design and compare life cycle energy inputs, GHG emissions and cost  
154 between the following two sulfide control strategies: 1) a commercial operation of nitrite  
155 production from urine and 2) a conventional chemical dosing method.

156 Sulfide control in the sewer using nitrite produced from urine consists of three steps: urine  
157 collection, on-site treatment and dosing. A hypothetical commercial building with a male  
158 population of approximately 1000 was assumed to have a built-in urine diverting system for

159 urine collection from the toilet urinals in men's lavatories, which, compared with typical urinal  
160 units, only requires additional pipes. The on-site treatment units are assumed to be located in  
161 the basement of the building and to include a urine collection tank, a treatment reactor, and a  
162 storage tank for dosing. The collected fresh urine was thoroughly decomposed in the collection  
163 tank with simultaneous conversion of urea to ammonia in 2 days, driving the pH to increase  
164 from approximately 6 to 9 (Mobley and Hausinger, 1989). Nitrite was then produced in the  
165 treatment reactor, and the nitrite-containing effluent was stored in the third unit and  
166 intermittently dosed into the sewer for sulfide control to nearby branch pipes with a gravity  
167 system. Assuming that 15 liters of diluted urine was produced per male person per day, and  
168 that people worked 5 days a week, a total of 7.14 m<sup>3</sup>/day diluted urine was estimated to be  
169 produced in the building during working hours. The volume of the collection tank, reactor tank  
170 and storage tank was designed as 16 m<sup>3</sup>, 8 m<sup>3</sup> and 8 m<sup>3</sup>, respectively. The life-time of the  
171 system was assumed to be 20 years.

172 The infrastructure inputs included pipes, steel, concrete, and excavation. The operation  
173 inputs included electricity for mixing, aeration, and pumping. The infrastructure input data  
174 were taken from Cashman *et al.* (2014); electricity consumption data for operation were  
175 calculated based on data from Jiang *et al.* (2014); and life cycle inventory data for chemicals,  
176 electricity and construction materials were from the GaBi database (GaBi Software, 2012).  
177 GHG emissions were expressed as CO<sub>2</sub> equivalents (CO<sub>2</sub>-e) using the 100-year global warming  
178 potentials (GWPs) of CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O from the IPCC's fifth report (Edenhofer *et al.*, 2014).  
179 The conversion ratio of nitrous oxide (N<sub>2</sub>O) nitrogen to consumed ammonia nitrogen under

180 aeration conditions was assumed to be 1.7%, as evaluated in a full-scale nitrification reactor for  
181 reject water treatment (Kampschreur *et al.*, 2008). Energy inputs, GHGs and cost were reported  
182 for 1 kg N of nitrite produced from urine (kg nitrite-N). The capital cost of the reactor was  
183 estimated using a unit price per volume of approximately \$60/m<sup>3</sup> (Hao and Hao, 2009). The  
184 price of electricity was set at \$0.12/kWh from the literature (Law *et al.*, 2015). The dosing rate  
185 requirement of NaNO<sub>2</sub> was calculated to achieve an equivalent sulfide control effect as realized  
186 with the FNA application approach. Acid dosage is likely required to lower the sewage pH in  
187 order to achieve an effective FNA level. However, this was neglected in the analysis as the  
188 amount of acid needed in the two cases would be the same or similar. The life cycle inventory  
189 energy and GHG emissions were obtained from the GaBi database (GaBi Software, 2012). The  
190 price of NaNO<sub>2</sub> was assumed to be \$460/tonne (Alibaba website).

### 191 **3. Results**

#### 192 *3.1 Reactor operation to achieve stable nitrite production from urine*

193 Experiments were performed in three phases to investigate the effects of influent  
194 characteristics and operational conditions on nitrite production from urine, as described in  
195 section 2.2. The reactor performance is shown in Figure 1-3, and summarized in Table 1.

##### 196 *3.1.1 Phase I: start-up and stable operation with influent ammonia concentration at ~550* 197 *mg N/L*

198 Initially, the influent ammonia nitrogen concentration was 424.3±81.7 mg N/L, and the  
199 average FA concentration reached up to 140 mg NH<sub>3</sub>-N/L. The high FA concentration  
200 significantly inhibited the activities of both AOB and nitrite oxidizing bacteria (NOB)

201 (Anthonisen *et al.*, 1976). However, the activity of AOB recovered after four days (Figure 1).  
202 The effluent nitrite nitrogen concentration increased sustainably and stabilized at above 250  
203 mg/L on day 10. The effluent nitrate nitrogen concentration slightly increased to 46.1 mg/L in  
204 the initial 6 days, before gradually decreasing to below 10 mg/L on day 10, and stayed at this  
205 low level. On day 10, the effluent pH became stable at approximately 5.7, and stayed at 5.6-  
206 5.8 for the remaining period of the study. The typical reactor operational cycle studies showed  
207 that the pH gradually decreased as ammonia was converted to nitrite because of the nitrification  
208 process (data not shown). On day 15, the influent ammonia nitrogen concentration was  
209 increased and maintained at 550 mg/L. Stable nitrite production with average nitrogen  
210 concentration at  $280.3 \pm 16.6$  mg/L was achieved during 100 days' operation (Figure 1). The  
211 average nitrite accumulation ratio in the effluent was high as  $99.9 \pm 0.3\%$ . The average ammonia  
212 removal efficiency was  $48.4 \pm 2.0\%$ , which indicates that nearly half of the ammonium in the  
213 feed was converted to nitrite.

### 214 3.1.2 Phase II: Reactor operation with influent ammonia nitrogen concentration fluctuation 215 under low and high DO conditions

216 Two reactors were operated in parallel in Phase II. The reactor operated at a lower DO level  
217 (0.2–0.3 mg/L) performed similarly to the reactor at a higher DO level (1.0–2.0 mg/L) in the  
218 initial 40 days (Figure 2). The nitrite accumulation ratios in the effluent were both very high,  
219 and reached  $89.4 \pm 6.2\%$  and  $87.9 \pm 6.3\%$  in the low and high DO reactors, respectively, with a  
220 high influent ammonia concentration at 850 mg/L. When the influent ammonia concentration  
221 decreased to a lower level down to 375 mg/L, the effluent nitrite accumulation ratio

222 considerably decreased approximately 50% in the high DO reactor, while the ratio remained  
223 basically unchanged in the low DO reactor. The results indicate that DO indeed played an  
224 important role for the achievement of nitrite production with relatively low influent ammonia  
225 concentration but did not have a significant impact on the reactor performance fed with  
226 wastewater containing a high-level ammonia like urine collected in most cases. A lower DO is  
227 also preferred due to lower energy consumption.

### 228 3.1.3 Phase III: Reactor operation with alkalinity supplement in the urine

229 Figure 3 shows the reactor performance with and without alkalinity supplement in the urine.  
230 The operational DO was set as 0.2-0.3 mg/L. Initially, the reactor was operated to stably  
231 produce nitrite as described above. On day 31, additional  $\text{NaHCO}_3$  was added to the influent  
232 as an alkalinity supplement, which facilitated higher ammonia removal efficiencies of up to  
233  $94.8 \pm 13.5\%$ . The average nitrite accumulation ratio was as high as  $90.3 \pm 13.3\%$ . The ammonia  
234 removal efficiency returned to approximately 50% on day 84, when alkalinity supply was  
235 terminated. These results indicate that the technology is flexible and can produce nitrite at  
236 appropriate levels when properly controlled.

### 237 3.1.4 Community analysis

238 The microbial communities in the inoculated sludge (sample T0) and reactor sludge when  
239 stable nitrite production was achieved in Phase I (sample T1 take on Day 85, see Figure 1) are  
240 compared in Figure 4. The 454-pyrosequencing analysis generated 19352 DNA gene sequences  
241 from the two samples and separated them into 373 OTUs. The OTUs calculated from each  
242 sample using Good's coverage estimator showed that this test captured 99% of the species in

243 the samples. The relative abundances of the *Nitrosomonas* and *Nitrospira* genera in the  
244 inoculated sludge were 3.08% and 4.79%, respectively. The relative abundance of the  
245 *Nitrosomonas* genus reached up to 17.46%, while that of the *Nitrospira* genus was as low as  
246 0.02% during the stable production of FNA. In the biomass, the AOB in the *Nitrosomonas*  
247 genus became predominant, whereas the NOB in the *Nitrospira* genus were selectively  
248 eliminated.

### 249 3.1.5 Organics consumption in the nitrifying reactor

250 The aeration process also resulted in a decrease in the organics concentration of the urine.  
251 The average COD concentration decreased from  $464.6 \pm 56.9$  mg/L in the influent to  $94.5 \pm 28.4$   
252 mg/L in the effluent, but the loss of total organics in domestic wastewater could be ignored  
253 because most of those organics were from feces (Larsen and Gujer, 1996) and other sources.

### 254 3.2 Dosing tests for sulfide control in the laboratory-scale sewer system

255 Previous studies demonstrated that dosing of FNA prepared from hydrochloric acid and  
256 sodium nitrite is effective for sulfide control in sewers (Jiang *et al.*, 2011a; Jiang *et al.*, 2013).  
257 Sulfide production was reduced by more than 80% in 10 days following each dosing of FNA  
258 at a concentration of 0.26 mg  $\text{HNO}_2\text{-N/L}$  and with a dosing duration of 8 h in real rising main  
259 sewers in Australia (Jiang *et al.*, 2013). Thus, dosing tests for sulfide control comparing the  
260 FNA agent produced from urine with the chemical FNA agent were experimentally conducted  
261 in the laboratory-scale sewer systems. The sediment sulfide production rate gradually  
262 decreased with the nitrite/FNA concentration increase in the groups of reactors (Figure 5).  
263 After a 12 h exposure to FNA at 0.14 mg  $\text{HNO}_2\text{-N/L}$ , achieved through the addition of either

264  $\text{NaNO}_2$  or nitrite produced from urine (a similar amount of HCl was added in both cases to  
265 achieve a pH level of 6.5), the sediments sulfide production rate decreased by approximately  
266 50%, and was down to 0.48-0.52 mg S/(d·m<sup>2</sup>) in both cases. This means that nitrite produced  
267 from urine is as effective as a commodity nitrite source when applied to sulfide control in  
268 sewers.

### 269 *3.3 Life cycle GHG, energy and cost analysis*

270 According to the results reported above, nitrite produced from urine has the same sulfide  
271 control effect as  $\text{NaNO}_2$ . For the testing case designed in 2.5, the annual nitrite production is  
272 estimated to be 781 kg N, equivalent to 3850 kg  $\text{NaNO}_2$  as a commodity product for achieving  
273 the same sulfide control effectiveness. The production cost, energy inputs and GHG emission  
274 results of our approach are compared to those of the commercial supply in Table 2. The  
275 production of FNA from urine showed significant advantages compared with the commercial  
276  $\text{NO}_2^-$  supply in all aspects. The life cycle energy and GHGs for the commercial production of  
277 pure  $\text{NaNO}_2$  are 3514 MJ/kg-N and 13.75 kg  $\text{CO}_2\text{e}$ /kg-N, respectively. In comparison, the  
278 energy requirement and GHG emissions for urine conversion to nitrite are 44.74 MJ/kg-N  
279 (98.7% reduction) and 9.76 kg  $\text{CO}_2\text{e}$ /kg-N (29% reduction), respectively. Additionally, the  
280 production cost is also reduced by 69.1%. For the GHG emissions,  $\text{N}_2\text{O}$  emission during the  
281 biological conversion of ammonia to nitrite represents the most significant source due to the  
282 high global warming potential of  $\text{N}_2\text{O}$ . Uncertainty analysis indicated that the savings in GHG  
283 emissions would be negative when the  $\text{N}_2\text{O}$  conversion factor is increased to above 2.7% in the  
284 urine nitrification reactor. Therefore, efforts are needed to minimize  $\text{N}_2\text{O}$  emissions during



285 urine treatment.

## 286 **4. Discussion**

### 287 *4.1 The proposed integrated urban water management strategy*

288 This study demonstrates that source-separated urine can be used for sulfide control in sewers.  
289 With aerobic treatment, approximately 50% of the ammonia can be converted to nitrite,  
290 generating FNA in sewage for sewer biofilm control when discharged to sewers particularly  
291 when acid is also dosed. This decentralized treatment also reduces the total nitrogen load to the  
292 downstream centralized wastewater treatment plants. The experimentally obtained results also  
293 showed that the sulfide control effectiveness of nitrite thus produced is essentially the same as  
294  $\text{NaNO}_2$  as a commodity product. According to Jiang *et al.* (2010) and Jiang *et al.* (2011a), FNA  
295 is effective for sewer biofilm control when added to sewage at 100 mg N/L for 12 hours every  
296 1-2 weeks. As an ammonium to nitrite conversion ratio of 50% in the urine treatment reactor,  
297 this means a total nitrogen addition of 200 mg N/L during each dosing period. This means that  
298 roughly 15-30% of the urine should be collected in a catchment in order to produce an adequate  
299 amount of nitrite for sewer biofilm control. The strategy is evaluated as being easy to  
300 implement and providing economic and energy savings.

301 Conventionally, ammonium in domestic wastewater is transported through sewers and then  
302 removed by nitrification and denitrification processes in downstream treatment plants. Strict  
303 effluent standards for total nitrogen incur not only high aeration costs but also loss of carbon  
304 sources for denitrification (Zheng *et al.*, 2013b; Zheng *et al.*, 2014). Resources can be  
305 recovered more effectively if wastewater streams are not mixed but treated separately.

306 Fundamentally, urine source separation has been reported as a promising innovation in  
307 sustainable urban water management (Lienert and Larsen, 2010). In these publications, the  
308 main purposes of separate urine treatment were hygienization, volume reduction, stabilization,  
309 nutrient recovery, and micropollutant handling (Maurer and Larsen, 2006; Udert and Wächter,  
310 2012; Zhang *et al.*, 2014). However, developing low cost resource recovery technologies is  
311 still critical. In this study, we converted ammonia to nitrite for sulfide control in sewers, a  
312 ‘niche’ application. As previously reported, after being treated by FNA at ppm levels, only 2–  
313 3% of microorganisms in an anaerobic sewer biofilm remained viable (Jiang *et al.*, 2011b), a  
314 result that is a good indication of the feasibility of using the FNA product from urine to control  
315 the ubiquitous sulfide problem in sewers.

316 Commonly used methods for sulfide control in sewers involved dosing chemicals agents  
317 such as oxygen, iron salts, nitrates, nitrites, sodium hydroxide, alkalis and microbial inhibitors  
318 have been reported (Ganigue *et al.*, 2011; Liu *et al.*, 2014b). Among them, FNA dosing had  
319 the lowest cost and had been evaluated in field trials by Jiang *et al.* (2013). Using the urine-  
320 generated FNA agent should make this approach even more competitive. Differences in the  
321 purity of the FNA agent from urine and industrial chemicals should be noted. Before  
322 environmental and human health risk assessments, expanding the scope of using the urine-  
323 generated FNA beyond the sewers is unexpected.

324 In general, dosing urine-generated FNA in nearby upstream pipes should have an inherent  
325 advantage because these pipes have a high area-to-volume ratio and are important sources of  
326 sulfide leading to corrosion of main pipes (Sharma *et al.*, 2008; Guisasola *et al.*, 2009).

327 Previous studies showed that through the optimization of intermittent and simultaneous dosage  
328 in pressure pipe systems, FNA treatment at a concentration of 0.26 mg HNO<sub>2</sub>-N/L with a 12 h  
329 exposure was able to effectively suppress sulfide production (Jiang *et al.*, 2011a; Jiang *et al.*,  
330 2013). However, in commonly used gravity pipe systems, maintaining the same exposure time  
331 requires substantially more chemicals and thus has additional negative effects on downstream  
332 wastewater treatment plants. Therefore, the dosing strategy significantly depends on the  
333 successive exposure time of FNA in pipes, and the effect of dilution on the dosage needs to be  
334 considered further when using the urine-generated FNA. In addition, the wastewater volume  
335 in the sewer system can exceed the capacity of the sewer system during periods of heavy  
336 rainfall or snowmelt. In this case, the dosing of the urine-generated FNA should be terminated  
337 to avoid nitrite emissions.

#### 338 *4.2 Achieving nitrite production and its relationship with the pH limit for AOB metabolism.*

339 Through urine wastewater treatment, nitrite production was achieved in the reactor. Half of  
340 the influent ammonium was converted to nitrite at a pH of approximately 5.7. The results are  
341 similar to nitrite production from anaerobic digester liquor in a treatment plant (Law *et al.*,  
342 2015). Many experimental results presented before showed a strong dependency of the nitrite  
343 pathway on DO concentration (Ma *et al.*, 2009; Wang *et al.*, 2016), while this study  
344 demonstrated that achieving the nitrite pathway was not significantly affected by DO.  
345 Community analysis confirmed that stable maintenance of the nitrification process was due to  
346 AOB predominance and NOB elimination in the biomass, which has been identified in the  
347 literature as a key factor for achieving the nitrite pathway (Wang *et al.*, 2014; Zheng *et al.*,

2016). However, to achieve the nitrite pathway, this study further revealed that in addition to the above community response, obtaining a final pH as low as 5.7 is essential. Normally, pH drops with proton production and with alkalinity consumption coupled with ammonia oxidation. The rate of the ammonia oxidation declines as pH decreases and very often stops at a pH slightly below 6. Therefore, in the case of nitrification, cessation of ammonia oxidation is very important to obtain a low final pH and to achieve the nitrite pathway. Conventional kinetic studies assumed that FNA inactivates NOB more significantly than it does on AOB (Wang *et al.*, 2014) and relatively high concentrations of FNA would completely inhibit the activity of AOB (Park *et al.*, 2010). These assumptions indicate that the oxidation of ammonia to nitrite, together with a decrease in pH by AOB, would increase FNA inhibition and result in a cessation of ammonia oxidation. However, the pH could also drop from 9 to approximately 6 in the case of nitrification without FNA inhibition using a moving bed biofilm reactor (MBBR) with biofilm carriers (Udert *et al.*, 2003). Fumasoli *et al.* (2015) proposed a mathematical model to simulate the pH limit of a known AOB, *Nitrosomonas eutropha*, and indicated that energy limitation would be responsible for the cessation of ammonia oxidation. Thus, cessation of ammonia oxidation might be mainly influenced by the FA substrate rather than by external FNA inhibition. In addition, the lowest possible ammonium concentration for achieving the FNA pathway requires further research.

## 5. Conclusions

This study proposed a promising approach to treating urine to produce nitrite for sulfide control in sewers. The methods for achieving the production of nitrite and then using the urine-

369 generated nitrite (forming FNA along with acid dosage) for sulfide control were experimentally  
370 investigated. Economic and environmental analysis was also conducted. The main conclusions  
371 are as follows:

- 372 ● Nitrite produced from urine is effective for sulfide control in sewers.
- 373 ● Based on urine collection from a public building, the proposed strategy of dosing urine-  
374 generated nitrite/FNA for sulfide control in sewers would significantly reduce the cost and  
375 GHG emissions compared with conventional chemical agent approach.
- 376 ● A nitrifying reactor can easily convert 50% of ammonium in urine to nitrite suitable for  
377 sewer discharge for sulfide control.

### 378 **Acknowledgments**

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### 383 **References**

- 384 Anthonisen, A.C., Loehr, R.C., Prakasam, T.B.S., Srinath, E.G, 1976. Inhibition of nitrification  
385 by ammonia and nitrous-acid. *J. Water. Pollut. Con. F.* 48 (5), 835–852.
- 386 Behzadian, K., Kapelan, Z., 2015. Advantages of integrated and sustainability based  
387 assessment for metabolism based strategic planning of urban water systems. *Sci. Total.*  
388 *Environ.* 527–528, 220–231.

- 389 Cashman, S., Gaglione, A., Mosley, J., Weiss, L., Hawkins, T.R., Ashbolt, N.J., Cashdollar, J.,  
390 Xue, X., Ma, C., Arden, S., 2014. Environmental and Cost Life Cycle Assessment of  
391 Disinfection Options for Municipal Wastewater Treatment. EPA 600/R-14/377.
- 392 Edenhofer, O., Pichs-Madruga, R., Sokona, Y., 2014. Climate Change 2014: Mitigation of  
393 Climate Change: Working Group III Contribution to the Fifth Assessment Report of the  
394 Intergovernmental Panel on Climate Change, 511-597.
- 395 Fumasoli, A., Morgenroth, E., Udert, K.M., 2015. Modeling the low pH limit of *Nitrosomonas*  
396 *eutropha* in high-strength nitrogen wastewaters. Water Res. 83, 161–170.
- 397 GaBi Software: Life Cycle Assessment LCA Software. Professional US Extension. Leinfelden-  
398 Echterdingen. 2012 (<http://www.gabi-software.com>).
- 399 Ganigue, R., Gutierrez, O., Rootsey, R., Yuan, Z., 2011. Chemical dosing for sulfide control  
400 in Australia: an industry survey. Water Res. 45 (19), 6564–6574.
- 401 Ge, H., Zhang, L., Batstone, D., Keller, J., Yuan, Z., 2012. Impact of iron salt dosage to sewers  
402 on downstream anaerobic sludge digesters: Sulfide control and methane production. J.  
403 Environ. Eng 139 (4), 594–601.
- 404 Guisasola, A., Sharma, K.R., Keller, J., Yuan, Z., 2009. Development of a model for assessing  
405 methane formation in rising main sewers. Water Res. 43 (11), 2874–2884.
- 406 Gutierrez, O., Park, D., Sharma, K.R., Yuan, Z., 2010. Iron salts dosage for sulfide control in  
407 sewers induces chemical phosphorus removal during wastewater treatment. Water Res.  
408 44 (11), 3467–3475.

- 409 Hao, Y., Hao, Z., 2009. Analysis of the influences of wastewater treatment plant water quality  
410 parameters on the project cost. *Water & Wastewater Engineering in Chinese*. 35 (7), 39–  
411 41.
- 412 Jiang, G., Gutierrez, O., Sharma, K. R., Yuan, Z., 2010. Effects of nitrite concentration and  
413 exposure time on sulfide and methane production in sewer systems. *Water Res.* 44 (14),  
414 4241–4251.
- 415 Jiang, G., Gutierrez, O., Sharma, K.R., Keller, J., Yuan, Z., 2011a. Optimization of intermittent,  
416 simultaneous dosage of nitrite and hydrochloric acid to control sulfide and methane  
417 productions in sewers. *Water Res.* 45 (18), 6163–6172.
- 418 Jiang, G., Gutierrez, O., Yuan, Z., 2011b. The strong biocidal effect of free nitrous acid on  
419 anaerobic sewer biofilms. *Water Res.* 45 (12), 3735–3743.
- 420 Jiang, G., Keating, A., Corrie, S., O'halloran, K., Nguyen, L., Yuan, Z., 2013. Dosing free  
421 nitrous acid for sulfide control in sewers: Results of field trials in Australia. *Water Res.*  
422 47 (13), 4331–4339.
- 423 Jiang, G., Yuan, Z., 2014. Inactivation kinetics of anaerobic wastewater biofilms by free nitrous  
424 acid. *Appl. Microbiol. Biotechnol.* 98 (3), 1367–1376.
- 425 Kampschreur, M. J., van der Star, W. R., Wielders, H. A., Mulder, J. W., Jetten, M. S., van  
426 Loosdrecht, M. C. M., 2008. Dynamics of nitric oxide and nitrous oxide emission during  
427 full-scale reject water treatment. *Water Res.* 42 (3), 812–826.
- 428 Larsen, T.A., Gujer, W., 1996. Separate management of anthropogenic nutrient solutions  
429 (human urine). *Water Sci. Technol.* 34 (3–4), 87–94.

- 430 Law, Y., Ye, L., Wang, Q., Hu, S., Pijuan, M., Yuan, Z., 2015. Producing free nitrous acid - A  
431 green and renewable biocidal agent - From anaerobic digester liquor. *Chem. Eng. J.* 259,  
432 62–69.
- 433 Lienert, J., Larsen, T.A., 2010. High acceptance of urine source separation in seven european  
434 countries: A review. *Environ. Sci. Technol.* 44 (2), 556–566.
- 435 Liu, Y., Wu, C., Zhou, X., Zhang, T., Mu, L., Shi, H., 2014a. Effect of variation of liquid  
436 condition on transformation of sulfur and carbon in the sediment of sanitary sewer. *J.*  
437 *Environ. Manag.* 154, 65–69.
- 438 Liu, Y., Wu, C., Zhou, X., Shi, H., Zhu, D.Z., 2014b. Study on the sulfide elimination by  
439 intermittent nitrate dosing in sediments of sewer system. *J. Environ. Manag.* 27, 259–265.
- 440 Ma, Y., Peng, Y., Wang, S., Yuan, Z., Wang, X., 2009. Achieving nitrogen removal via nitrite  
441 in a pilot-scale continuous pre-denitrification plant. *Water Res.* 43 (3), 563–572.
- 442 Maurer, M. Pronk, W., Larsen, T.A., 2006. Treatment processes for source-separated urine.  
443 *Water Res.* 40 (17), 3151–3166.
- 444 Ministry of Environmental Protection, P. R. C. Monitoring and analytical methods of water  
445 and wastewater. 4th ed.; China Environmental Science Press: Beijing, 2006.
- 446 Mobley, H.L.T., Hausinger, R.P., 1989. Microbial ureases: significance, regulation, and  
447 molecular characterization. *Microbiol Rev.* 53 (1), 85–108.
- 448 Park, S., Bae, W., Rittmann, B.E., 2010. Operational boundaries for nitrite accumulation in  
449 nitrification based on minimum/maximum substrate concentrations that include effects of



- 450 oxygen limitation, pH, and free ammonia and free nitrous acid inhibition. *Environ. Sci.*  
451 *Technol.* 44 (1), 335–342.
- 452 Pikaar, I., Sharma, K.R., Hu, S., Gernjak, W., Keller, J., Yuan, Z., 2014. Reducing sewer  
453 corrosion through integrated urban water management. *Science*. 345 (6198), 812–814.
- 454 Sharma, K.R., Yuan, Z., de Haas, D., Hamilton, G., Corrie, S., Keller, J., 2008. Dynamics and  
455 dynamic modelling of H<sub>2</sub>S production in sewer systems. *Water Res.* 42 (10–11), 2527–  
456 2538.
- 457 Sun, J., Pikaar, I., Sharma, K.R., Keller, J., Yuan, Z., 2015a. Feasibility of sulfide control in  
458 sewers by reuse of iron rich drinking water treatment sludge. *Water Res.* 71, 150–159.
- 459 Sun, X., Jiang, G., Bond, P. L., Keller, J., Yuan, Z., 2015b. A novel and simple treatment for  
460 control of sulfide induced sewer concrete corrosion using free nitrous acid. *Water Res.*  
461 70, 279–287.
- 462 Udert, K.M., Fux, C., Münster, M., Larsen, T.A., Siegrist, H., Gujer, W., 2003. Nitrification  
463 and autotrophic denitrification of source-separated urine. *Water Sci. Technol.* 48 (1),  
464 119–130.
- 465 Udert, K.M., Larsen, T.A., Gujer, W., 2006. Fate of major compounds in source-separated  
466 urine. *Water Sci. Technol.* 54, 413–420.
- 467 Udert, K.M., Wächter, M., 2012. Complete nutrient recovery from source-separated urine by  
468 nitrification and distillation. *Water Res.* 46 (2), 453–464.

- 469 US Environmental Protection Agency, July 2010. State of Technology for Rehabilitation of  
470 Wastewater Collection Systems. US Environmental Protection Agency, Office of  
471 Research and Development. EPA/600/R-10/078.
- 472 Wang, D., Wang, Q., Laloo, A., Xu, Y., Bond, P.L., Yuan, Z., 2016. Achieving stable nitrification  
473 for mainstream deammonification by combining free nitrous acid-based sludge treatment  
474 and oxygen limitation. *Sci. Rep.* 6, 25547.
- 475 Wang, Q., Ye, L., Jiang, G., Hu, S., Yuan, Z., 2014. Side-stream sludge treatment using free  
476 nitrous acid selectively eliminates nitrite oxidizing bacteria and achieves the nitrite  
477 pathway. *Water Res.* 55, 245–255.
- 478 Xu, K., Zhang, C., Li, J., Cheng X., Wang, C., 2017. Removal and recovery of N, P and K from  
479 urine via ammonia stripping and precipitations of struvite and struvite-K. *Water Sci.*  
480 *Technol.* 75 (1), 155–164.
- 481 Zhang, J., She, Q., Chang, V.W.C., Tang, C.Y., Webster, R.D., 2014. Mining nutrients (N, K,  
482 P) from urban source-separated urine by forward osmosis dewatering. *Environ. Sci.*  
483 *Technol.*, 48 (6), 3386–3394.
- 484 Zheng, M., Liu, Y., Xu, K., Wang, C., He, H., Zhu, W., Dong, Q., 2013a. Use of low frequency  
485 and density ultrasound to stimulate partial nitrification and simultaneous nitrification and  
486 denitrification. *Bioresour. Technol.* 146, 537–542.
- 487 Zheng, M., Liu, Y., Wang, C., Xu, K., 2013b. Study on enhanced denitrification using  
488 particulate organic matter in membrane bioreactor by mechanism modeling.  
489 *Chemosphere.* 93: 2669–2674.

- 490 Zheng, M., Liu, Y., Wang, C., 2014. Modeling of enhanced denitrification capacity with  
491 microbial storage product in MBR systems. *Sep. Purif. Technol.* 126: 1–6.
- 492 Zheng, M., Liu, Y., Xin, J., Zuo, H., Wang, C., Wu, W., 2016. Ultrasonic treatment enhanced  
493 ammonia-oxidizing bacterial (AOB) activity for nitrification process. *Environ. Sci.*  
494 *Technol.* 50 (2), 864–871.

495 **Table 1.** Reactor performance for the production of nitrite.

Experiments	Time (days)	DO	Influent	Effluent			Ammonia removal	
		control	Ammonia (mg N/L)	Ammonia (mg N/L)	Nitrite (mg N/L)	Nitrate (mg N/L)	Nitrite accumulation ratio (%)	efficiency (%)
I	1-14	-	424.3±81.7	239.6±53.3	140.4±116.3	16.3±12.3	73.0±29.6	41.9±17.0
	15-98	-	546.5±8.8	282.0±9.6	280.3±16.6	0.2±0.7	99.9±0.3	48.4±2.0
II	1-25	Low*	615.9±34.8	323.5±130.8	200.7±144.0	20.1±22.4	80.9±31.4	47.9±19.7
		High**		352.4±130.9	139.4±162.5	23.6±21.8	71.7±30.2	42.9±20.6
	25-37	Low	850.0±0.0	394.2±58.9	389.8±49.3	47.1±26.1	89.4±6.2	53.6±6.9
		High		440.8±44.8	442.2±46.1	61.9±35.4	87.9±6.3	52.8±5.3
	38-55	Low	565.0±0.0	185.4±29.4	251.5±26.1	35.9±9.8	87.6±2.5	67.2±5.2
		High		204.0±34.1	155.1±25.3	166.2±53.2	49.5±13.2	63.9±6.0
	56-70	Low	375.0±0.0	156.8±47.3	211.1±42.6	0.0±0.0	100.0±0.0	72.2±8.4
		High		153.0±45.4	90.8±9.9	87.8±31.9	52.1±10.9	72.9±8.0
III	1-30	-	803.7±115.4	353.0±63.6	330.7±55.9	0.0±0.0	100.0±0.0	56.1±3.9
	31-83***	Low	457.6±6.4	23.9±62.7	358.2±40.3	21.1±14.8	90.3±13.3	94.8±13.5
	84-126	-	559.3±72.4	288.7±14.1	217.0±91.8	16.4±17.4	87.6±19.4	58.6±5.0

496 \*Reactor was performed with low DO of 0.2–0.3 mg/L.

497 \*\*Reactor was performed with high DO in the range of 1.0–2.0 mg/L.

498 \*\*\* An alkalinity supplement of NaHCO<sub>3</sub> was added in the influent to increase ammonia removal efficiency.

499 **Table 2.** LCA and cost analysis of the production of nitrite from urine and the commodity nitrite  
 500 (functional unit in 1 kg nitrite-N produced).

System Assumptions		Urine nitrite	NaNO <sub>2</sub>
Nitrite Production from Urine	Lifetime (Period over which capital costs are annualized)	20	
	Male population in the building	1,000	
	Operation time (days/year)	365	
	Flow rate of the urine (m <sup>3</sup> /day)	7.14	
	NH <sub>4</sub> <sup>+</sup> -N concentration in urine (mg N/L)	600	
	NH <sub>4</sub> <sup>+</sup> -N flow (kg N/year)	1564	
	Volume of the urine collection tank (m <sup>3</sup> )	16	
	Volume of the FNA production reactor (m <sup>3</sup> )	8	
	Volume of the FNA storage tank (m <sup>3</sup> )	8	
	<b>NO<sub>2</sub>-N production (kg N/year)</b>	<b>781</b>	<b>781</b>
Construction Materials	PVC pipe length (m)	500	
	PVC pipe mass (kg)	1,520	
	Steel (kg/m <sup>3</sup> )	0.15	
	Steel (kg)	4.80	
	Concrete (m <sup>3</sup> /m <sup>3</sup> )	0.0009	
	Concrete (kg)	69	
	Excavation (m <sup>3</sup> )	32	
Operation Inputs	O <sub>2</sub> /NH <sub>4</sub> <sup>+</sup> -N (g/g)	3.43	
	O <sub>2</sub> /COD (g/g)	1	
	Reactive NH <sub>4</sub> <sup>+</sup> -N (kg N/y)	781	
	COD (kg COD/y)	781	
	O <sub>2</sub> requirement (kg/y)	3460	
	Electricity for NH <sub>4</sub> <sup>+</sup> -N oxidation (kWh/kg O <sub>2</sub> )	0.66	
	Electricity for pumping (kWh/(m <sup>3</sup> .d))	0.078	
Electricity for reactor mixing (kWh/(m <sup>3</sup> . d))	0.12		
Life Cycle Energy Consumption	Life cycle energy for construction (MJ/ 20 years)	110,894	
	Life cycle energy for operation electricity (MJ/ 20 years)	587,976	
	<b>Total life cycle energy (MJ/ kg NO<sub>2</sub>-N)</b>	<b>44.74</b>	<b>3,514</b>
	Life cycle energy saving		<b>98.7%</b>
Life Cycle GWP	On-site N <sub>2</sub> O emission (kg/year)	20.88	
	GHGs from N <sub>2</sub> O emission (kg CO <sub>2</sub> e/ 20 years)	110695	
	GHGs from construction (kg CO <sub>2</sub> e/ 20 years)	5,074	
	GHGs from operation electricity (kg CO <sub>2</sub> e/ 20 years)	36829	
	<b>Total GHGs (kg CO<sub>2</sub>e/ kg NO<sub>2</sub>-N)</b>	<b>9.76</b>	<b>13.75</b>
	GHG Emissions Saving		<b>29.0%</b>
Life Cycle Cost	Cost of tanks (\$/m <sup>3</sup> )	60	
	Capital cost of three tanks and major equipment (\$)	2160	
	Interest applied for initial capital expenditure	8.50%	
	Annualized capital cost (\$/y)	228	
	Power price (\$/kWh)	0.12	
	Cost of O <sub>2</sub> supply (\$/y)	247	
	Cost of mixing (\$/y)	61	
	Total cost (\$/y)	536	
	<b>Cost (\$/ kg NO<sub>2</sub>-N)</b>	<b>0.69</b>	<b>2.22</b>
		Cost saving	

501 **Figure Captures**

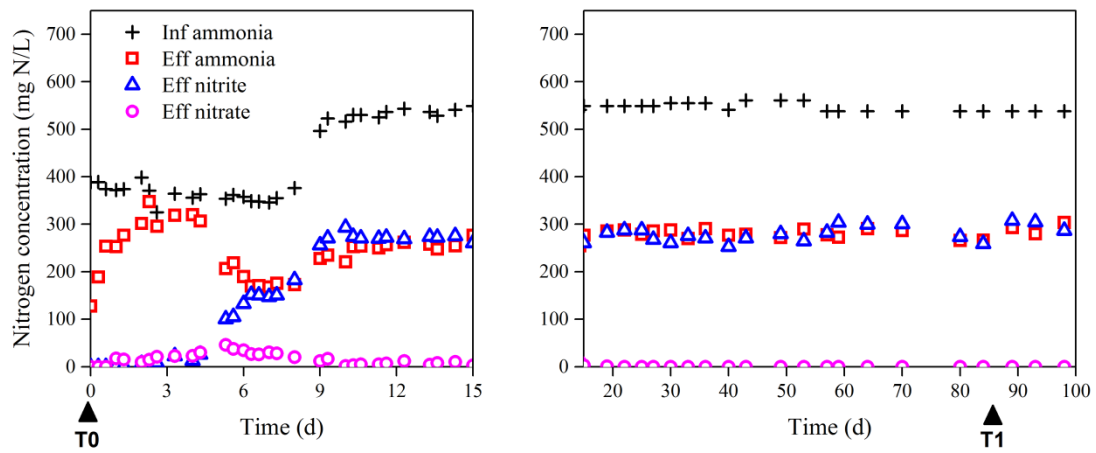
502 **Figure 1.** Phase I results. Reactor start-up (left) and stable (right) performance with the  
503 treatment of urine from a male toilet urinal with influent ammonia concentration around 550  
504 mg/L. ▲: Sludge sample collection for community analysis.

505 **Figure 2.** Phase II results. Reactor performance with fluctuation in the influent ammonia  
506 nitrogen concentration under low DO of 0.2-0.3 mg/L (A) and high DO in the range of 1.0-2.0  
507 mg/L (B) operational conditions.

508 **Figure 3.** Phase III results. Reactor performance with and without alkalinity supplement in the  
509 influent (DO = 0.2-0.3 mg/L).

510 **Figure 4.** The relative abundance of the predominant groups at genus level according 454  
511 pyrosequencing. T0: The inoculated nitrification sludge. T1: The sludge collected from the  
512 reactor stably producing nitrite.

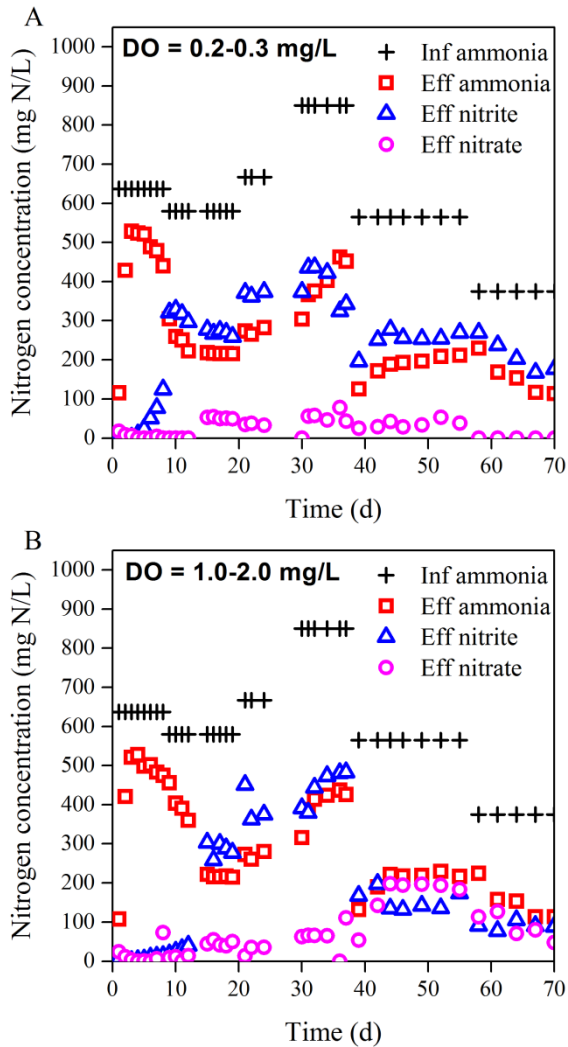
513 **Figure 5.** Measured sulfide production rates under different nitrite/FNA concentrations (pH =  
514 6.5).



515

516 **Figure 1.** Phase I results. Reactor start-up (left) and stable (right) performance with the  
 517 treatment of urine from a male toilet urinal. The influent ammonia concentration was around  
 518 550 mg/L during stable operation. ▲: Sludge sample collection for community analysis.

519

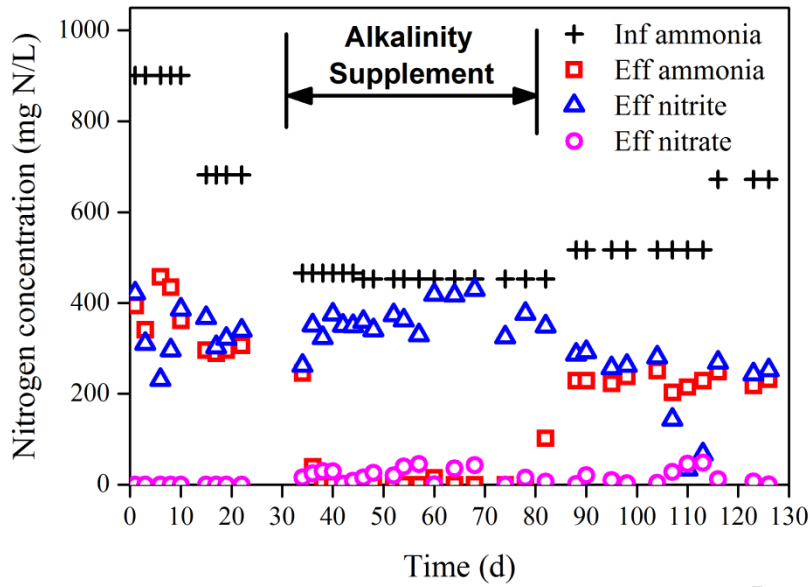


520

521 **Figure 2.** Phase II results. Reactor performance with fluctuation in the influent ammonia  
 522 nitrogen concentration under low DO of 0.2-0.3 mg/L (A) and high DO in the range of 1.0-2.0  
 523 mg/L (B) conditions.

524

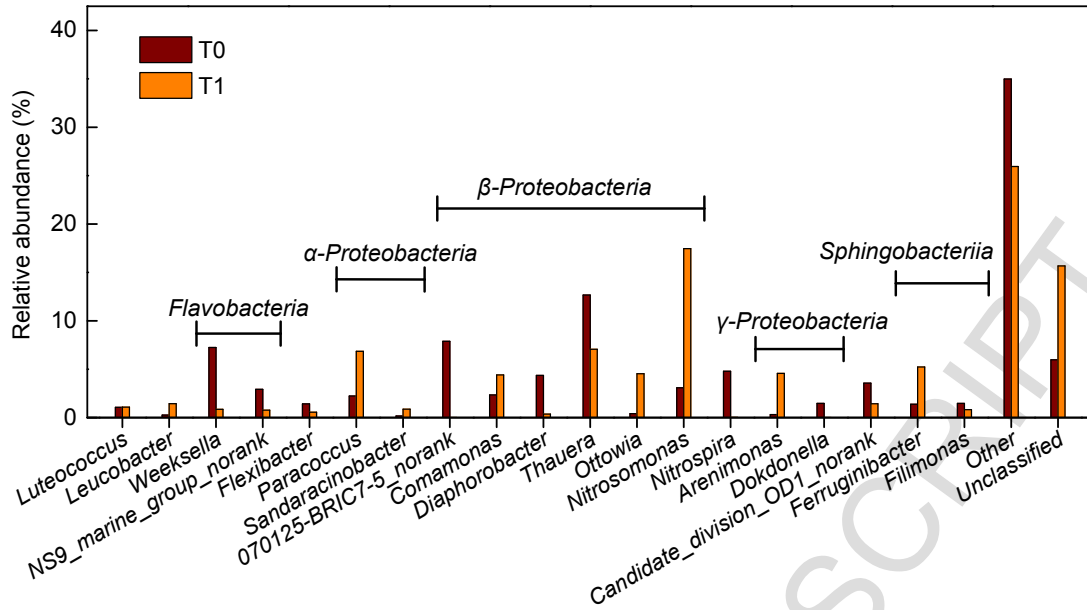




525

526 **Figure 3.** Phase III results. Reactor performance with and without alkalinity supplement in the  
 527 influent (DO = 0.2-0.3 mg/L).

528



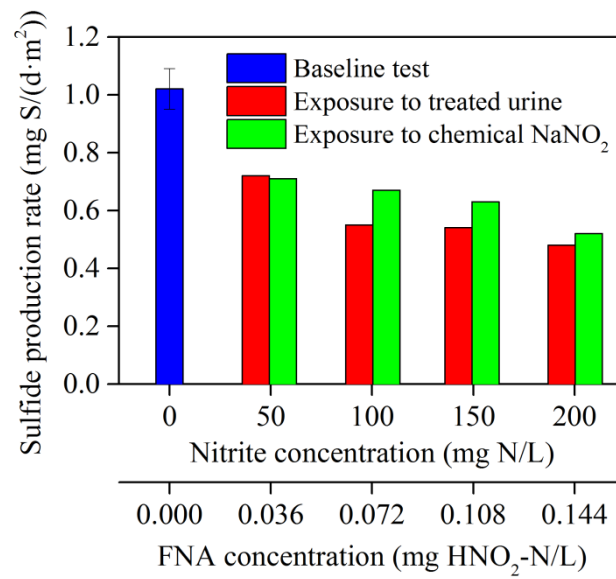
529

530 **Figure 4.** The relative abundance of the predominant groups at genus level according to 454

531 pyrosequencing. T0: The inoculated nitrification sludge. T1: The sludge collected from the

532 reactor stably producing nitrite.

533



534

535 **Figure 5.** Measured sulfide production rates after 12 h exposure to different nitrite/FNA

536 concentrations (pH = 6.5).

**Highlights**

- A green chemical produced from urine for sulfide control in sewer
- Urine can be partially (-50%) converted to nitrite
- The approach is cost-effective with a lower carbon footprint