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Microbial Electrolysis Followed by Chemical Precipitation for Effective Nutrients Recovery from Digested Sludge Centrate in WWTPs

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Abstract

A two-step sidestream process was investigated for nitrogen (N) and phosphorous (P) recovery from digested sludge centrate. In the first step, a dual-chamber microbial electrolysis cell (MEC) was used for N recovery on the cathode. In the second step, P was recovered as solid precipitates by the addition of Ca^{2+} or Mg^{2+} salts in the anodic effluent. The operation of MEC with centrate indicate that N transport from the anode to the cathode chamber is primarily driven by anodic electron transport rather than diffusional transport. Low concentration of readily biodegradable organics in centrate significantly hindered current density (<0.15 A/m^2) and led to trivial N recovery on the cathode chamber. The addition of primary sludge fermentation liquor (25 vol%) with centrate as an exogenous source of readily biodegradable organics substantially increased current density up to 6.4 A/m^2, along with high TAN removal efficiency of $53\pm5\%$. The energy requirement was calculated at 5.8 ± 0.1 kWh/kg-TAN; however, the recovered H_2 gas from the cathode was adequate to offset this energy input completely. The addition of Ca^{2+} salt at a Ca: P molar ratio of 3:1 was optimum for P recovery from the anodic effluent; Mg: P molar ratio of 2:1 was found to be optimum for Mg^{2+} salt addition. However, optimum doses of both salts resulted in maximum P recovery efficiency of ~85%, while Mg^{2+} addition provided an additional 38% TAN removal. These results demonstrate that microbial electrolysis followed by chemical precipitation can promote sustainable nutrients recovery from centrate at municipal wastewater treatment plants where sludge fermentation has already been adopted to provide readily biodegradable carbon source in the biological nutrient removal process.

1. Introduction

Anaerobic digestion for bio-methane production is one of the most popular approaches for

sewage sludge management in centralized wastewater treatment facilities. During the anaerobic

digestion process, a substantial amount of inorganic nutrients (primarily nitrogen and

phosphorous) is released due to the hydrolysis of particulate organics [1–4]. Anaerobic

methanogenic microbiome usually lacks adequate autotrophic metabolism of inorganic nutrients;

hence, the liquid effluents from digesters contain a high concentration of these nutrients [1–4]. In

many municipal wastewater treatment plants (WWTPs), digested sludge centrate is

recycled back to the aerobic biological treatment process (i.e., activated sludge process), which

leads to a significant increase in aeration cost due to 15-20% additional nitrogen (N) load

depending on the influent wastewater characteristics [4,5]. Moreover, the extra load imposed by

centrate may cause the effluent quality to exceed effluent regulatory limits of nutrient

constituents [4].

The energy required for N removal in conventional two-stage nitrification-denitrification

process is estimated to be 12.5 kWh/kg-N, while emerging anaerobic ammonia oxidation process

(Anammox®) requires about 4.2 kWh/kg-N [6,7]. These biological treatment processes primarily

remove N, while it would be more sustainable to recover N from concentrated wastewater stream

like digested sludge centrate. There is a recognized market-value of nutrients available in

digested sludge centrate. Currently, the Haber-Bosch process is widely used for

ammonia synthesis for various industrial applications [8–10], which require high energy input of

10-12.5 kWh/kg-N. Additionally, a significant portion of the phosphorus (P) used for agriculture

and other industrial application is obtained by mining, while the global reserve of minable

phosphate is expected to be depleted by the end of the twenty-first century [8]. Hence, there has

73   been a growing interest in developing effective processes for N and P recovery from centrate,

74   which can eventually reduce the combined load of nutrients to the mainstream treatment process

75   [4,5]. Different sidestream processes studied for N and P recovery from digested sludge centrate

76   includes direct stripping, ion exchange, electrodialysis, chemical precipitation and microbial

77   electrochemical technologies [1,3–5,10–15]. Most of these processes focus on recovery of one

78   constituent (either N or P). Ammonia stripping with acid absorption for ammonium sulfate

79   production has been proven to be an efficient method for N recovery, while high energy demand

80   (~28 kWh/kg-N) is one of the major barriers to its widescale adoption [10]. Recovery of

81   ammonia nitrogen using ion exchange resins has often been ruled out due to high regeneration

82   cost as well as deterioration of their ion exchange capacities after multiple regeneration cycles

83   [11]. Electrodialysis (ED), an emerging process which deploys a series of ion exchange

84   membranes with applied electric potential, can efficiently recover N from various wastewater

85   streams [4,10]. A recent study by Ward et al.[10] reported that the electrodialysis system could

86   attain concentrated ammonium-N solution from digester sludge centrate with a relatively lower

87   energy input of 4.9±1.5 kWh/kg-N. However, frequent cleaning of multiple membrane modules

88   remained a concern, as membrane fouling can lead to higher energy inputs.

89   Chemical precipitation of nutrients as struvite (magnesium ammonium phosphate) from

90   various wastewater stream has been predominantly investigated for P recovery. Recovered

91   struvite can be directly utilized as a slow-release fertilizer [4,14,16,17]. Along with the addition

92   of magnesium (Mg^{2+}) salt, struvite process may consume a significant amount of base chemicals

93   due to the requirement of high pH conditions for struvite precipitation [13]. Struvite process can

94   provide up to 90% P recovery from wastewater with minimal N removal: theoretically, per gram

95   of P removal as struvite removes 0.45 gram of N [4]. In comparison with struvite, only a few

96 studies reported efficient P recovery as calcium phosphate (CaP) from digested sludge centrate

97 [15,18]. To the best of our knowledge, to date, no studies have compared P recovery efficiencies

98 as struvite and calcium phosphate.

99   Microbial electrochemical systems, such as microbial electrolysis cells (MECs) and

100 microbial fuel cells (MFCs), have recently received much attention for energy neutral wastewater

101 treatment and value-added resources recovery [19,20]. Studies have also demonstrated that

102 microbial electrochemical systems can simultaneously recover nutrients from wastewater, which

103 makes them more attractive over other nutrients recovery technologies [21–26]. In dual-chamber

104 microbial electrochemical systems having cation exchange membrane (CEM), efficient cathodic

105 nitrogen recovery could be attained due to ammonium transport from the anode to the cathode

106 chamber for meeting charge neutrality [23,24]. Thus, a concentrated ammonia nitrogen solution

107 can be recovered from the cathode. Additionally, a few studies demonstrated that cathodic proton

108 consumption for H_2 production in MEC might lead to highly alkaline local pH near the cathode

109 which facilitates struvite precipitation on the cathode surface [22,27]. Interestingly, most of these

110 studies primarily focused on nutrients recovery from source-separated urine [22–24,27–31]. In

111 contrast, microbial electrochemical systems for nutrients recovery from digested sludge centrate

112 has been rarely investigated. A recent study by Yuan and Kim [13] reported that a small

113 concentration of readily biodegradable organics in digested sludge centrate resulted in low current

114 density and inferior cathodic nutrients recovery in a MEC. Thus, inadequate readily biodegradable

115 organics could be a possible reason that made digested sludge centrate a relatively unattractive

116 stream for nutrients recovery with microbial electrochemical systems. Therefore, the development

117 of an efficient engineering strategy to improve current density and nutrients recovery from centrate

118   would be essential to promote the microbial electrochemical system as a sidestream process for

119   centrate, and thereby reduce nutrients load to mainstream biological process.

120   It is apparent that none of the abovementioned technologies can be used as a standalone

121   process for simultaneous N and P recovery from digested sludge centrate. More efforts in

122   developing integrated processes are required to promote sustainable nutrients recovery from

123   centrate. Herein, a dual-chamber MEC having CEM followed by chemical precipitation was

124   investigated as an integrated process for N and P recovery from digested sludge centrate

125   collected from a full-scale municipal WWTP. First, primary sludge fermentation liquor was

126   assessed as a supplementary source of readily biodegradable organics for enhancing

127   electrochemical performance of MEC and ensuring efficient recovery of ammonium nitrogen on

128   the cathode chamber. Second, the effluent from MEC was further assessed for P recovery

129   through chemical precipitation with the addition of Ca^{2+} and Mg^{2+} salts. The overall goal of this

130   current study was to determine if this integrated process of microbial electrolysis followed by

131   chemical precipitation with Ca^{2+} or Mg^{2+} could lead to a technically viable option for nutrients

132   recovery from digested sludge centrate in municipal WWTPs.

133   **2. Materials and methods**

134   **2.1 MEC operation for nitrogen removal and recovery**

135   A bench-scale dual-chamber MEC constructed from plexiglass was used in this study. The

136   design of MEC is similar to the one previously described by Zakaria et al. [32]. Briefly

137   describing the design, the working volumes of anode and cathode chambers were 400 mL and

138   140 mL, respectively (see Fig. 1a). A photograph of the MEC is provided in the Supplementary

139   Information.   Bundles of carbon fibers (2293-A, 24A Carbon Fiber, Fibre Glass Development

140   Corp., Ohio, USA) integrated with a stainless-steel frame was used as the anode electrode.

6

141   Carbon fibers were pretreated according to a method previously described by Dhar et al. [33]. A

142   stainless-steel mesh (Type 304, McMaster Carr, OH, USA) was used as the cathode electrode.

143   In this study, the anode and cathode chambers were separated by a CEM (CMI-7000,

144   Membranes International Inc, USA). Ag/AgCl reference electrode (MF-2052, Bioanalytical

145   System Inc., WestLafayette, IN, USA) was placed within ~1 cm of the anode electrode. The

146   anode potential was set at -0.4 V vs. Ag/AgCl using a multi-channel potentiostat system

147   (Squidstat Prime, Admiral Instruments, Phoenix, AZ, USA) throughout the experiment. Effluent

148   from a mother MEC was used as the inoculum for the enrichment of anode biofilms as

149   previously reported by Zakaria et al. [32] and 25 mM of sodium acetate medium supplemented

150   mineral nutrients, and 50 mM phosphate buffer served as the substrate for the enrichment of

151   anode biofilms. The detailed composition of the mineral nutrients can be found in the literature

152   [33]. The MEC was operated at room temperature ($22\pm1°C$), and both the anolyte and catholyte

153   were continuously mixed at 130 ± 10 rpm with magnetic stirrers.

154   During the enrichment process of anodic biofilms, MEC was operated in semi-continuous

155   mode (i.e., 180 mL of fresh substrate medium was fed to the anode chamber using a syringe on a

156   daily basis) until a repeatable peak current of 10.4 ± 0.9 A/m^2 was observed. Then, nitrogen

157   recovery experiments were conducted with digested sludge centrate, centrate supplemented with

158   exogenous readily biodegradable organics (acetate and ethanol), and centrate mixed with primary

159   sludge fermentation liquor (see Fig. 1b). During Cycle-1 and 2, MEC was operated with real

160   digested centrate. Anaerobic digester sludge was collected from a full-scale mesophilic

161   anaerobic digester at the Gold Bar Wastewater Treatment Plant, Edmonton, Alberta, Canada.

162   Anaerobic digester sludge was centrifuged at 18,670xg for 30 min, and the centrate was collected

163   for experiments. During Cycle-3, 4, and 5, MEC was operated with centrate supplemented with

7

164 three different concentrations of exogenous readily biodegradable organics (435, 860, and 1355

165 mg COD/L, respectively). A mixture of acetate and ethanol (COD ratio of 1:1) was used as

166 exogenous readily biodegradable organics.   During Cycle-6 and 7, MEC was operated with a

167 mixture of digested sludge centrate (75 vol%) and liquor from primary sludge fermentation (25

168 vol%). For sludge fermentation experiment, primary sludge was collected from Gold Bar

169 Wastewater Treatment Plant, Edmonton, Alberta, Canada. The fermentation experiment was

170 performed in a bench-scale glass anaerobic bioreactor (working volume of 2 L) equipped with a

171 stirring shaft. The sludge was sparged with ultra-pure nitrogen gas (99.999%) for three minutes.

172 The fermentation process was operated for 7 days at room temperature ($22\pm1°C$), and the reactor

173 media was stirred with a mixer at 300 rpm. After 7 days of operation, fermented sludge was

174 centrifuged at 18,670xg for 30 min, and the supernatant (i.e., fermentation liquor) was collected

175 and used for experiments (Cycle-6 and 7). Table 1 shows the average characteristics of centrate,

176 primary sludge, and fermentation liquor. All experiments were conducted in fed-batch mode.

177 Before each cycle, the cathode chamber was evacuated completely and cleaned multiple times

178 with tap water to remove any residual ammonium that can influence the results of the following

179 cycle.

180 **2.2 Chemical precipitation experiments for P recovery**

181 We performed nutrients precipitation experiments with anodic effluent collected during Cycle-5.

182 The average soluble reactive phosphorous (SRP) and total ammonia-nitrogen (TAN)

183 concentrations in the effluent were 1387 ± 51 mg/L and 451 ± 15 mg/L, respectively. For

184 precipitation experiments, specific amounts of $CaCl_2.2H_2O$ or $MgCl_2.6H_2O$ salts are added into

185 centrifuge vials having 50 mL of anodic effluent samples. The corresponding molar ratios of Ca

186 to P and Mg to P are summarized in Table 2. These ratios were selected based on the possible

187 precipitation of $Ca_3(PO_4)_2$ or $Ca_5(OH)(PO_4)_3$ and struvite ($MgNH_4PO_4 \cdot 6H_2O$), respectively, as

188 previously suggested in the literature [34,35]. Phosphorous precipitation tests with Ca^{2+} salt was

189 performed without any pH adjustment; pH of the anode effluent was 6.8. In contrast,

190 precipitation tests with Mg^{2+} salt were carried out at two different pH (6.8 and 8.6); pH was

191 adjusted to 8.6 using 3M NaOH. After addition of Ca^{2+} or Mg^{2+} salts, vials were tightly capped,

192 and liquids were mixed with a vortex mixer for 2 minutes. After 4 hours of settling period, vials

193 were centrifuged at 18,670xg for 30 min, and solid precipitates were collected. The solid

194 precipitates were examined for elemental composition, and supernatant samples were analyzed

195 for COD, TAN, and SRP concentrations. All experiments were conducted at room temperature

196 $(22\pm1°C)$.

197 **2.3 Analytical methods**

198 Applied voltage and current were recorded every five minutes in a computer using Squidstat

199 User Interface software (Squidstat Prime, Admiral Instruments, USA). COD, TAN, and SRP

200 concentrations were measured using HACH reagent kits (HACH, Loveland, Colorado, USA).

201 For analysis of soluble parameters, samples were filtered with 0.45 μm nylon syringe filter. pH

202 was measured using a bench-top pH meter (Accumet AR15, Fisher Scientific, Pittsburgh, PA,

203 USA), and solution conductivities were measured using a portable conductivity meter (Extech

204 EC100, EXTECH Instruments, USA). The concentrations of VFAs (acetate, propionate, and

205 butyrate) were measured using an ion chromatograph (Dionex ICS-2100, Dionex, Sunnyvale,

206 CA) equipped with an electrochemical detector (ECD) and microbore AS19 column. The

207 elemental composition of the solid precipitates was examined with scanning electron microscopy

208 (SEM) with energy dispersive X-ray spectroscopy (EDX). For analysis of the elemental

209 composition, solid precipitates were air dried and mounted on SEM stubs. Then, the samples

210 were sputter coated with gold and examined using SEM (Zeiss Sigma 300 VP-FESEM, Carl

211 Zeiss, Cambridge, UK).

2.4 Calculations and economic assessment

213 Current density was calculated based on the projected area of membrane (0.0038 m^2), as

214 previously described in the literature [9,33,36]. The anodic TAN removal efficiency was

215 calculated from the difference in initial and final TAN concentration of the anodic liquid during

216 a fed-batch cycle. The cathodic TAN recovery efficiency was calculated from the amount of

217 TAN accumulated in the cathode chamber and the amount of TAN removed from the anode

218 chamber during a fed-batch cycle. A preliminary economic assessment for MEC was performed

219 based on per kg TAN removal from centrate. Average experimental results from Cycle-6 and 7

220 was used for these calculations. Electrical energy input (kWh) to MEC for an entire fed-batch

221 cycle was computed from the current and applied voltage data recorded with time, as previously

222 described in the literature [37]. The cost of electrical energy input and monetary value of H$_2$

223 produced from MEC were estimated at \$0.07/kWh and \$8/kg-H$_2$, respectively [38,39].

3. Results and discussion

3.1 N recovery from centrate

3.1.1 N recovery and current density from centrate

227 Fig. 2a shows current density with time during MEC operated with digested sludge

228 centrate (Cycle-1 and 2). The peak current densities from MEC in both cycles (Cycle-1: 0.11

229 A/m^2; Cycle 2: 0.13 A/m^2) were substantially lower than the peak current density of 10.4±0.9

230 A/m^2 observed with 25 mM sodium acetate medium during enrichment of anode biofilms,

231 indicating that current density was limited due to the lack of readily biodegradable organics

10

232   in centrate. As expected, a major portion of SCOD in centrate was unutilized by anodic

233   microbial communities (Fig. 2b); SCOD removal efficiencies were ranged from 14% (Cycle-1)

234   to 26% (Cycle-2). After the operation of both cycles, TAN concentrations in the anode chamber

235   remained almost unchanged (Fig. 2c), while TAN concentration slightly increased in the cathode

236   chamber (Fig. 2d). Thus, it is possible that additional ammonia nitrogen has been released

237   through hydrolysis of particulate organics in the anode chamber. In MECs, ammonium transport

238   through CEM comprises of migration (driven by electric field) and diffusion (driven by

239   concentration gradient) mechanisms [23,24]. A few studies previously suggested that alkaline

240   pH (~8.8) and high TAN levels in source-separated urine (>3 g/L) might trigger diffusional

241   ammonium transport from the anode to the cathode chamber [23,24]. In contrast, the results of

242   this current study demonstrated that migration would be the primary mechanism for ammonium

243   transport from centrate possibly due to relatively lower pH (~7.9) and TAN level (~1 g/L).

244   Therefore, increasing current density in MEC would be crucial to enhance ammonium migration

245   to the cathode chamber.

246   **3.1.2 Enhanced N recovery from centrate supplemented with exogenous electron donors**

247   In response to the addition of exogenous readily biodegradable organics at different

248   concentrations, current densities in MEC noticeably increased as compared to centrate alone

249   (Fig. 3a). The peak current densities in Cycle-3, 4, and 5 were 2.4, 8.4, and 9.4 A/m^2,

250   respectively. Anodic TAN concentrations were observed to decrease as current densities

251   increased (Fig. 3b). The addition of readily biodegradable organics at high concentrations

252   (Cycle-4: 860 mg SCOD/L; Cycle-5: 1355 mg SCOD/L) showed a considerable decrease in

253   anodic TAN concentrations. During Cycle 4 and Cycle-5, TAN concentrations in the anode

254   chamber decreased from 1165 and 1178 mg/L to 594 and 456 mg/L, respectively; and TAN

255 concentrations in the cathode chamber increased up to 1152 mg/L (Cycle-4) and 1537 mg/L

256 (Cycle-5), respectively (Fig. 3c). Thus, anodic TAN removal efficiencies were 49% (Cycle-4)

257 and 61% (Cycle-5). Correspondingly, TAN fluxes through the membrane during these cycles

258 were 13 g-TAN/m^2-d (Cycle-4) and 18 g-TAN/m^2-d (Cycle-5). However, transported ammonium

259 recovery efficiencies on the cathode were 70% (Cycle-4) and 73% (Cycle-5). A portion of the

260 transported ammonium (27-30%) were possibly lost through stripping as ammonia gas due to the

261 highly alkaline cathodic pH (~12) as a result of proton reduction to H_2 gas on the cathode

262 ($2H^+ + 2e^- \rightarrow H_2$). The detailed pH profiles are provided in the Supplementary Information. In

263 comparison, low concentration of exogenous readily biodegradable organics (Cycle-3: 435 mg

264 SCOD/L) exhibited relatively lower N recovery efficiency. As shown in Fig. 3d, a substantial

265 decrease in SCOD concentrations were observed in comparison with MEC operation with

266 digested sludge centrate alone, indicating the lower biodegradability of residual organics in

267 digested centrate. As expected, there was no change in SRP concentrations due to the use of

268 CEM (Fig. 3e). Overall, these results support that the addition of exogenous readily

269 biodegradable organics favored current density and nitrogen recovery from digested centrate.

270 **3.1.3 Efficient N recovery from centrate supplemented with sludge fermentation liquor**

271 Because the addition of exogenous readily biodegradable organics improved current density and

272 N recovery, primary sludge fermentation liquor was assessed as a potential source of readily

273 biodegradable organics (Cycle-6 and 7) (Fig. 4). The TAN and SRP concentrations in digested

274 sludge centrate were not substantially diluted by the addition of fermentation liquor (25% by

275 volume). The operation of MEC with a mixture of centrate and fermentation liquor exhibited

276 peak current density of 6.4 A/m^2 (Cycle-6) and 6.2 A/m^2 (Cycle-7) (Fig. 4a). Fig. 4b and 4c

277 show the changes in TAN concentrations in anode and cathode chambers over the course of

12

278 these two cycles. On average, TAN concentrations in the anode chamber decreased from 957±35

279 mg/L to 455±45 mg/L, and TAN concentrations in the cathode chamber increased to 1194±117

280 mg/L. Average anodic TAN removal efficiency and corresponding flux through the membrane

281 were 53±5% and 26±4 g-TAN/m^2-d. On average, 85±14% of the transported TAN was

282 recovered on the cathode. This performance was comparable with the operation of MEC with

283 centrate supplemented with a high concentration of ethanol and acetate (i.e., Cycle-5). Average

284 SCOD removal efficiency was 21±2% (Fig. 4d). SRP concentrations in the anode chambers

285 were almost unchanged in both cycles, which is also consistent with previous cycles (Fig. 4e).

286 To the best of our knowledge, the results of this study first evidently demonstrated that the

287 addition of fermentation liquor could provide an adequate quantity of readily biodegradable

288 organics in centrate to promote nitrogen recovery with microbial electrochemical

289 systems. Several studies previously demonstrated that primary sludge fermentation liquor could

290 serve as a promising exogenous carbon source for biological nutrients removal (BNR) process in

291 WWTPs [40,41]. In fact, many WWTPs across North America already adopted sludge

292 fermentation to produce short-chain volatile fatty acids for providing exogenous electron donors

293 to the BNR process. We envision that a portion of sludge fermentation liquor can be diverted to

294 MEC for enhancing N recovery from centrate.

295   Based on the results obtained from Cycle-6 and 7, the average electrical energy

296 consumption per kg of TAN transport from the anode chamber to the cathode

297 chamber was estimated at 5.8±0.1 kWh. Thus, electricity cost was calculated at $0.40±0.01/kg-

298 TAN. Simultaneously with N recovery, H_2 gas can be recovered on the cathode. An average

299 H_2 production per cycle was 189±33 mL, which corresponds to H_2 productivity of 0.08±0 kg-

300 H_2 per kg of TAN removal. The monetary value of H_2 that can be simultaneously

301   produced during MEC operation was estimated at $0.64±0.1/kg-TAN, resulting in a net financial

302   gain of $0.23±0.01/kg-TAN, which does not include the economic value of concentrated

303   ammonium solution. This preliminary economic evaluation based on the bench-scale

304   experimental results indicates the potential of energy positive or energy neutral operation of

305   MEC for N recovery from digested sludge centrate. It must be noted that further comprehensive

306   economic assessment should be done at pilot-scale. Furthermore, to become a competitive

307   technology for N recovery, anodic TAN removal efficiency (53±5%) and rate (26±4 g TAN/m^2-

308   d) should be significantly increased. However, these values are within the range previously

309   reported for MECs operated with real or synthetic urine. Haddadi et al. [23] reported N removal

310   rate of 8.5 g-N/m^2-d in a MEC operated with synthetic urine. In contrast, Kuntke et al. [31]

311   reported high ammonium removal rate of 173.4 g-N/m^2-d in a MEC operated with urine;

312   however, ammonium removal efficiency was only ~30%. As nitrogen recovery from digested

313   centrate is primarily driven by migration or charge neutrality, improving current density would

314   be critical to enhance N recovery. Notably, the current density can be significantly enhanced

315   with multiple anode modules [42,43]. Additionally, more concentrated ammonium solution can

316   be achieved by increasing the ratio between dilute to concentrate volumes ($V_{D/C}$) until a certain

317   limit [28]. In this study, a concentration factor of ~1.25 with 53±5% TAN removal was attained

318   at $V_{D/C}$ of 2.86. This result is comparable with a concentration factor of ~1.75 (63% ammonium

319   removal from synthetic urine) at $V_{D/C}$ of 2 reported by a previous study [28].

320   **3.2 P recovery from MEC effluent**

321   Fig. 5 shows the changes in SRP and TAN concentrations in the anodic effluent after chemical

322   precipitation experiments at different conditions. The addition of Ca^{2+} salt induced a significant

323   reduction in SRP concentrations, while TAN concentration remained unchanged. At Ca:P molar

324 ratio of 1.5:1 ($Ca^{2+}_{pH=6.8}$ (1)), SRP concentration decreased by 75±1%. In contrast, comparable

325 SRP removals ($Ca^{2+}_{pH=6.8}$ (2): 85±1%; $Ca^{2+}_{pH=6.8}$ (3): 87±3%) were observed with higher doses of

326 Ca^{2+} salt. The SEM-EDX analysis suggested that the solid precipitate collected from $Ca^{2+}_{pH=6.8}$

327 (2) test was primarily composed of two major elements, namely, P and Ca (Fig. 6a). Thus,

328 P was efficiently recovered without any pH adjustment of MEC effluent. In contrast, SRP and

329 TAN concentrations did not change after addition of various doses of Mg^{2+} salt at pH 6.8 (i.e., no

330 pH adjustment). While pH of the MEC effluent increased to 8.6, both SRP and TAN

331 concentrations remarkably decreased with the addition of Mg^{2+} salt, which is consistent with

332 previous studies reported that alkaline pH would be essential for struvite precipitation [4,12,16].

333 In $Mg^{2+}_{pH=8.6}$ (1) test, the average removals of SRP and TAN were 77±1% and 38±3%,

334 respectively. The addition of higher doses of Mg^{2+} salt did not result in further improvement in

335 TAN removal efficiencies, while SRP removal efficiencies slightly improved ($Mg^{2+}_{pH=8.6}$ (2):

336 84±2%; $Mg^{2+}_{pH=8.6}$ (3): 84±1%). As shown in Fig. 6b, the EDX spectrum confirmed the major

337 components of the solid precipitates to be Mg (17.7%) and P (13.7%), which is comparable to

338 that of the elemental composition of standard struvite (Mg: 9.79%; P: 13.49%) previously

339 reported in the literature [44]. The absence of N peak was also consistent with the previous

340 literature, which could be due to its light weight [44]. In this study, optimum doses of Ca^{2+} (C:P

341 of 3:1 at pH=6.8; 4.8 g Ca^{2+} salt/g SRP) or Mg^{2+} (Mg:P of 2:1 at pH=8.6; 4.5 g Mg^{2+} salt/g SRP)

342 salts exhibited comparable maximum P recovery efficiencies from MEC effluent. A comparison

343 of the results indicated that Mg^{2+} addition at pH 8.6 could provide some additional benefit due to

344 further removal of TAN, while the pH adjustment would induce some additional chemical costs

345 [45,46].

346   To satisfy effluent discharge standard, the effluent (supernatant) from chemical

347   precipitation process can be recycled to a mainstream treatment process for removal of

348   unrecovered nutrients. The low pH of this stream can influence the performance of mainstream

349   biological treatment process. It is evident that Mg^{2+}-based process can provide an advantage

350   over Ca^{2+} addition due to relatively higher pH (≥ 7) in the final effluent (See Supplementary

351   Information). For Ca^{2+}-based process, post-adjustment of pH may be required before recycling

352   supernatant liquid stream. This would be particularly important for activated sludge systems

353   already having alkalinity limitations due to complete nitrification. However, a comprehensive

354   techno-economic assessment would be required for the selection of an optimum process.

355   Additionally, the recovered precipitates from both processes should be assessed for potential

356   environmental risks, including pathogens, antibiotic resistance genes, heavy metals, and

357   micropollutants, as suggested by recent studies [14,47,48]. To the best of our knowledge, this

358   study first provides a systematic comparison between the phosphorous recovery efficiencies with

359   Ca^{2+} and Mg^{2+} salts. It must be asserted that P precipitation with Mg^{2+} salt has been extensively

360   explored, while limited information is available in the literature on the P recovery with Ca^{2+} salt

361   [15,18]. Therefore, our results substantiate the importance of further comprehensive assessment

362   of various options for P precipitation from centrate.

363   **4. Conclusions**

364   The use of primary sludge fermentation liquor as a supplementary source of readily

365   biodegradable organics to digested sludge centrate appears to be a promising solution to boost

366   current density and subsequent N recovery in a dual-chamber MEC. The addition of

367   fermentation liquor (25% by volume) with centrate exhibited average anodic TAN removal

368   efficiency and rate of 53±5% and 26±4 g-TAN/m^2-d, respectively. This performance was almost

369   comparable with the addition of acetate and ethanol as exogenous electron donors for anodic

370   microbial communities. Additionally, recovered H_2 gas from the cathode was adequate to offset

371   the cost of electrical energy input to the process. Under optimum precipitation conditions,

372   maximum P recovery efficiencies from MEC effluents that could be achieved by either Ca^{2+} or

373   Mg^{2+} salts were about 85%. Thus, the results of this study demonstrated that MEC followed by

374   chemical precipitation can act as an efficient sidestream process for N and P recovery from

375   centrate. The process should be investigated in pilot-scale for further optimization and

376   application at municipal wastewater treatment plants.

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References

[1]   M.S. Romero-Güiza, S. Astals, J. Mata-Alvarez, J.M. Chimenos, Feasibility of coupling anaerobic digestion and struvite precipitation in the same reactor: Evaluation of different magnesium sources, Chem. Eng. J. (2015). doi:10.1016/j.cej.2015.02.057.

[2]   N. Marti, A. Bouzas, A. Seco, J. Ferrer, Struvite precipitation assessment in anaerobic digestion processes, Chem. Eng. J. (2008). doi:10.1016/j.cej.2007.10.023.

[3]   F. Zeng, Q. Zhao, W. Jin, Y. Liu, K. Wang, D.J. Lee, Struvite precipitation from anaerobic sludge supernatant and mixed fresh/stale human urine, Chem. Eng. J. (2018). doi:10.1016/j.cej.2018.03.088.

[4]   C. Eskicioglu, G. Galvagno, C. Cimon, Approaches and processes for ammonia removal from side-streams of municipal effluent treatment plants., Bioresour. Technol. 268 (2018) 797–810. doi:10.1016/j.biortech.2018.07.020.

[5]   T.A. Constantine, North american experience with centrate treatment technologies for ammonia and nitrogen removal, Water Environ. Fed. 11 (2006) 5271–5281. doi:10.2175/193864706783763291.

[6]   S. Lackner, E.M. Gilbert, S.E. Vlaeminck, A. Joss, H. Horn, M.C.M. van Loosdrecht, Full-scale partial nitration/anammox experiences – An application survey, Water Res. 55 (2014) 292–303. doi:10.1016/j.watres.2014.02.032.

[7]   P. Kuntke, T.H.J.A. Sleutels, M. Rodríguez Arredondo, S. Georg, S.G. Barbosa, A. ter Heijne, H.V.M. Hamelers, C.J.N. Buisman, (Bio)electrochemical ammonia recovery: progress and perspectives, Appl. Microbiol. Biotechnol. 102 (2018) 3865–3878. doi:10.1007/s00253-018-8888-6.

412   [8]   M. Xie, L.D. Nghiem, W.E. Price, M. Elimelech, Toward resource recovery from

413   wastewater: extraction of phosphorus from digested sludge using a hybrid forward

414   osmosis–membrane distillation process, Environ. Sci. Technol. Lett. 1 (2014) 191–195.

415   doi:10.1021/ez400189z.

416   [9]   M.E.R. Christiaens, S. Gildemyn, S. Matassa, T. Ysebaert, J. De Vrieze, K. Rabaey,

417   Electrochemical ammonia recovery from source-separated urine for microbial protein

418   production, Environ. Sci. Technol. 51 (2017) 13143–13150. doi:10.1021/acs.est.7b02819.

419   [10]   A.J. Ward, K. Arola, E. Thompson Brewster, C.M. Mehta, D.J. Batstone, Nutrient

420   recovery from wastewater through pilot scale electrodialysis, Water Res. 135 (2018) 57–

421   65. doi:10.1016/j.watres.2018.02.021.

422   [11]   Q. Deng, B.R. Dhar, E. Elbeshbishy, H.-S. Lee, Ammonium nitrogen removal from the

423   permeates of anaerobic membrane bioreactors: economic regeneration of exhausted

424   zeolite, Environ. Technol. 35 (2014) 2008–2017. doi:10.1080/09593330.2014.889759.

425   [12]   Y.V. Nancharaiah, S. Venkata Mohan, P.N.L. Lens, Recent advances in nutrient removal

426   and recovery in biological and bioelectrochemical systems, Bioresour. Technol. 215

427   (2016) 173–185. doi:10.1016/j.biortech.2016.03.129.

428   [13]   P. Yuan, Y. Kim, Increasing phosphorus recovery from dewatering centrate in microbial

429   electrolysis cells, Biotechnol. Biofuels. 10 (2017) 70. doi:10.1186/s13068-017-0754-8.

430   [14]   M. Abel-Denee, T. Abbott, C. Eskicioglu, Using mass struvite precipitation to remove

431   recalcitrant nutrients and micropollutants from anaerobic digestion dewatering centrate,

432   Water Res. 132 (2018) 292–300. doi:10.1016/j.watres.2018.01.004.

433   [15]   A.J. Ansari, F.I. Hai, W.E. Price, L.D. Nghiem, Phosphorus recovery from digested sludge

434   centrate using seawater-driven forward osmosis, Sep. Purif. Technol. 163 (2016) 1–7.

435   doi:10.1016/j.seppur.2016.02.031.

436   [16] S.G. Barbosa, L. Peixoto, B. Meulman, M.M. Alves, M.A. Pereira, A design of

437   experiments to assess phosphorous removal and crystal properties in struvite precipitation

438   of source separated urine using different Mg sources, Chem. Eng. J. (2016).

439   doi:10.1016/j.cej.2016.03.148.

440   [17] M. Quintana, M.F. Colmenarejo, J. Barrera, E. Sánchez, G. García, L. Travieso, R. Borja,

441   Removal of phosphorus through struvite precipitation using a by-product of magnesium

442   oxide production (BMP): Effect of the mode of BMP preparation, Chem. Eng. J. (2008).

443   doi:10.1016/j.cej.2007.04.002.

444   [18] J.L. Soler-Cabezas, J.A. Mendoza-Roca, M.C. Vincent-Vela, M.J. Luján-Facundo, L.

445   Pastor-Alcañiz, Simultaneous concentration of nutrients from anaerobically digested

446   sludge centrate and pre-treatment of industrial effluents by forward osmosis, Sep. Purif.

447   Technol. 193 (2018) 289–296. doi:10.1016/j.seppur.2017.10.058.

448   [19] B.R. Dhar, H.-S. Lee, Evaluation of limiting factors for current density in microbial

449   electrochemical cells (MXCs) treating domestic wastewater, Biotechnol. Reports. 4

450   (2014). doi:10.1016/j.btre.2014.09.005.

451   [20] A. Al-Mamun, M.S. Baawain, B.R. Dhar, I.S. Kim, Improved recovery of bioenergy and

452   osmotic water in an osmotic microbial fuel cell using micro-diffuser assisted marine

453   aerobic biofilm on cathode, Biochem. Eng. J. 128 (2017). doi:10.1016/j.bej.2017.09.020.

454   [21] P. Zamora, T. Georgieva, A. Ter Heijne, T.H.J.A. Sleutels, A.W. Jeremiasse, M. Saakes,

455   C.J.N. Buisman, P. Kuntke, Ammonia recovery from urine in a scaled-up microbial

456   electrolysis cell, J. Power Sources. 356 (2017) 491–499.

457   doi:10.1016/j.jpowsour.2017.02.089.

458 [22] P.T. Kelly, Z. He, Nutrients removal and recovery in bioelectrochemical systems: A

459   review, Bioresour. Technol. 153 (2014) 351–360. doi:10.1016/j.biortech.2013.12.046.

460 [23] S. Haddadi, E. Elbeshbishy, H.-S. Lee, Implication of diffusion and significance of anodic

461   pH in nitrogen-recovering microbial electrochemical cells., Bioresour. Technol. 142

462   (2013) 562–569. doi:10.1016/j.biortech.2013.05.075.

463 [24] P. Kuntke, K.M. Śmiech, H. Bruning, G. Zeeman, M. Saakes, T.H.J.A. Sleutels, H.V.M.

464   Hamelers, C.J.N. Buisman, Ammonium recovery and energy production from urine by a

465   microbial fuel cell, Water Res. 46 (2012) 2627–2636. doi:10.1016/j.watres.2012.02.025.

466 [25] M. Qin, Y. Liu, S. Luo, R. Qiao, Z. He, Integrated experimental and modeling evaluation

467   of energy consumption for ammonia recovery in bioelectrochemical systems, Chem. Eng.

468   J. (2017). doi:10.1016/j.cej.2017.06.182.

469 [26] M. Qin, C. White, S. Zou, Z. He, Passive separation of recovered ammonia from catholyte

470   for reduced energy consumption in microbial electrolysis cells, Chem. Eng. J. (2018).

471   doi:10.1016/j.cej.2017.11.190.

472 [27] R.D. Cusick, B.E. Logan, Phosphate recovery as struvite within a single chamber

473   microbial electrolysis cell, Bioresour. Technol. 107 (2012) 110–115.

474   doi:10.1016/j.biortech.2011.12.038.

475 [28] R.C. Tice, Y. Kim, Energy efficient reconcentration of diluted human urine using ion

476   exchange membranes in bioelectrochemical systems, Water Res. 64 (2014) 61–72.

477   doi:10.1016/j.watres.2014.06.037.

478   [29]   P. Ledezma, P. Kuntke, C.J.N. Buisman, J. Keller, S. Freguia, Source-separated urine

479   opens golden opportunities for microbial electrochemical technologies., Trends

480   Biotechnol. 33 (2015) 214–20. doi:10.1016/j.tibtech.2015.01.007.

481   [30]   J. You, J. Greenman, C. Melhuish, I. Ieropoulos, Electricity generation and struvite

482   recovery from human urine using microbial fuel cells, J. Chem. Technol. Biotechnol. 91

483   (2016) 647–654. doi:10.1002/jctb.4617.

484   [31]   P. Kuntke, T.H.J.A. Sleutels, M. Saakes, C.J.N. Buisman, Hydrogen production and

485   ammonium recovery from urine by a microbial electrolysis cell, Int. J. Hydrogen Energy.

486   39 (2014) 4771–4778. doi:10.1016/j.ijhydene.2013.10.089.

487   [32]   B.S. Zakaria, S. Barua, A. Sharaf, Y. Liu, B.R. Dhar, Impact of antimicrobial silver

488   nanoparticles on anode respiring bacteria in a microbial electrolysis cell, Chemosphere.

489   213 (2018) 259–267. doi:10.1016/J.CHEMOSPHERE.2018.09.060.

490   [33]   B.R. Dhar, Y. Gao, H. Yeo, H.-S. Lee, Separation of competitive microorganisms using

491   anaerobic membrane bioreactors as pretreatment to microbial electrochemical cells,

492   Bioresour. Technol. 148 (2013) 208–214. doi:10.1016/j.biortech.2013.08.138.

493   [34]   I. Joko, Phosphorus Removal from Wastewater by the Crystallization Method, Water Sci.

494   Technol. 17 (1985) 121–132. doi:10.2166/wst.1985.0124.

495   [35]   L.E. de-Bashan, Y. Bashan, Recent advances in removing phosphorus from wastewater

496   and its future use as fertilizer (1997–2003), Water Res. 38 (2004) 4222–4246.

497   doi:10.1016/j.watres.2004.07.014.

498   [36]   J. An, H.-S. Lee, Implication of endogenous decay current and quantification of soluble

499   microbial products (SMP) in microbial electrolysis cells, RSC Adv. 3 (2013) 14021–

500   14028. doi:10.1039/c3ra41116h.

501 [37] B.E. Logan, Microbial fuel cells, Wiley-Interscience, 2008.

502 [38] B.R. Dhar, G. Nakhla, M.B. Ray, Techno-economic evaluation of ultrasound and thermal

503   pretreatments for enhanced anaerobic digestion of municipal waste activated sludge,

504   Waste Manag. 32 (2012) 542–549. doi:10.1016/j.wasman.2011.10.007.

505 [39] U. Mukherjee, A. Maroufmashat, J. Ranisau, M. Barbouti, A. Trainor, N. Juthani, H. El-

506   Shayeb, M. Fowler, Techno-economic, environmental, and safety assessment of hydrogen

507   powered community microgrids; case study in Canada, Int. J. Hydrogen Energy. 42 (2017)

508   14333–14349. doi:10.1016/j.ijhydene.2017.03.083.

509 [40] C. Zhang, Y. Chen, Simultaneous nitrogen and phosphorus recovery from sludge-

510   fermentation liquid mixture and application of the fermentation liquid to enhance

511   municipal wastewater biological nutrient removal, Environ. Sci. Technol. (2009).

512   doi:10.1021/es9005948.

513 [41] H. Yuan, Y. Chen, H. Zhang, S. Jiang, Q. Zhou, G. Gu, Improved bioproduction of short-

514   chain fatty acids (SCFAs) from excess sludge under alkaline conditions, Environ. Sci.

515   Technol. (2006). doi:10.1021/es052252b.

516 [42] L. Ren, Y. Ahn, H. Hou, F. Zhang, B.E. Logan, Electrochemical study of multi-electrode

517   microbial fuel cells under fed-batch and continuous flow conditions, J. Power Sources.

518   257 (2014) 454–460. doi:10.1016/j.jpowsour.2013.11.085.

519 [43] B.R. Dhar, H. Ryu, J.W. Santo Domingo, H.-S. Lee, Ohmic resistance affects microbial

520   community and electrochemical kinetics in a multi-anode microbial electrochemical cell,

521   J. Power Sources. 331 (2016) 315–321. doi:10.1016/j.jpowsour.2016.09.055.

23

522 [44] S. Zou, M. Qin, Y. Moreau, Z. He, Nutrient-energy-water recovery from synthetic

523 sidestream centrate using a microbial electrolysis cell - forward osmosis hybrid system, J.

524 Clean. Prod. 154 (2017) 16–25. doi:10.1016/j.jclepro.2017.03.199.

525 [45] Y. Luo, H. Li, Y.R. Huang, T.L. Zhao, Q.Z. Yao, S.Q. Fu, G.T. Zhou, Bacterial

526 mineralization of struvite using MgO as magnesium source and its potential for nutrient

527 recovery, Chem. Eng. J. (2018). doi:10.1016/j.cej.2018.06.106.

528 [46] T.L. Zhao, H. Li, Y.R. Huang, Q.Z. Yao, Y. Huang, G.T. Zhou, Microbial mineralization

529 of struvite: Salinity effect and its implication for phosphorus removal and recovery, Chem.

530 Eng. J. (2019). doi:10.1016/j.cej.2018.10.139.

531 [47] Q.-L. Chen, X.-L. An, Y.-G. Zhu, J.-Q. Su, M.R. Gillings, Z.-L. Ye, L. Cui, Application

532 of struvite alters the antibiotic resistome in soil, rhizosphere, and phyllosphere, Environ.

533 Sci. Technol. 51 (2017) 8149–8157. doi:10.1021/acs.est.7b01420.

534 [48] J.R. Cunha, C. Schott, R.D. van der Weijden, L.H. Leal, G. Zeeman, C. Buisman, Calcium

535 addition to increase the production of phosphate granules in anaerobic treatment of black

536 water, Water Res. 130 (2018) 333–342. doi:10.1016/j.watres.2017.12.012.

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543 **List of Figures**

544 **Fig. 1.** (a) Schematic diagram of MEC, and (b) various experimental schemes with MEC.

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547 centrate in Cycle-1 and Cycle-2.

548 **Fig. 3.** (a) Current density profiles, (b) changes in anodic TAN concentrations, (c) changes in

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550 MEC operation with centrate supplemented with various concentrations of exogenous readily

551 biodegradable organics (Cycle-3, 4, and 5).

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563 phosphorous recovery from anodic effluents.

Highlights

- MEC followed by chemical precipitation was studied for N&P recovery from centrate.
- Inadequate biodegradable COD in centrate led to ineffective N recovery in MEC.
- Centrate was mixed with sludge fermentation liquor to enhance N recovery in MEC.
- Recovered H_2 gas was adequate to offset the electricity cost of MEC operation.
- P was efficiently recovered from MEC anodic effluent using Ca^{2+} or Mg^{2+} salts.

Graphical Abstract
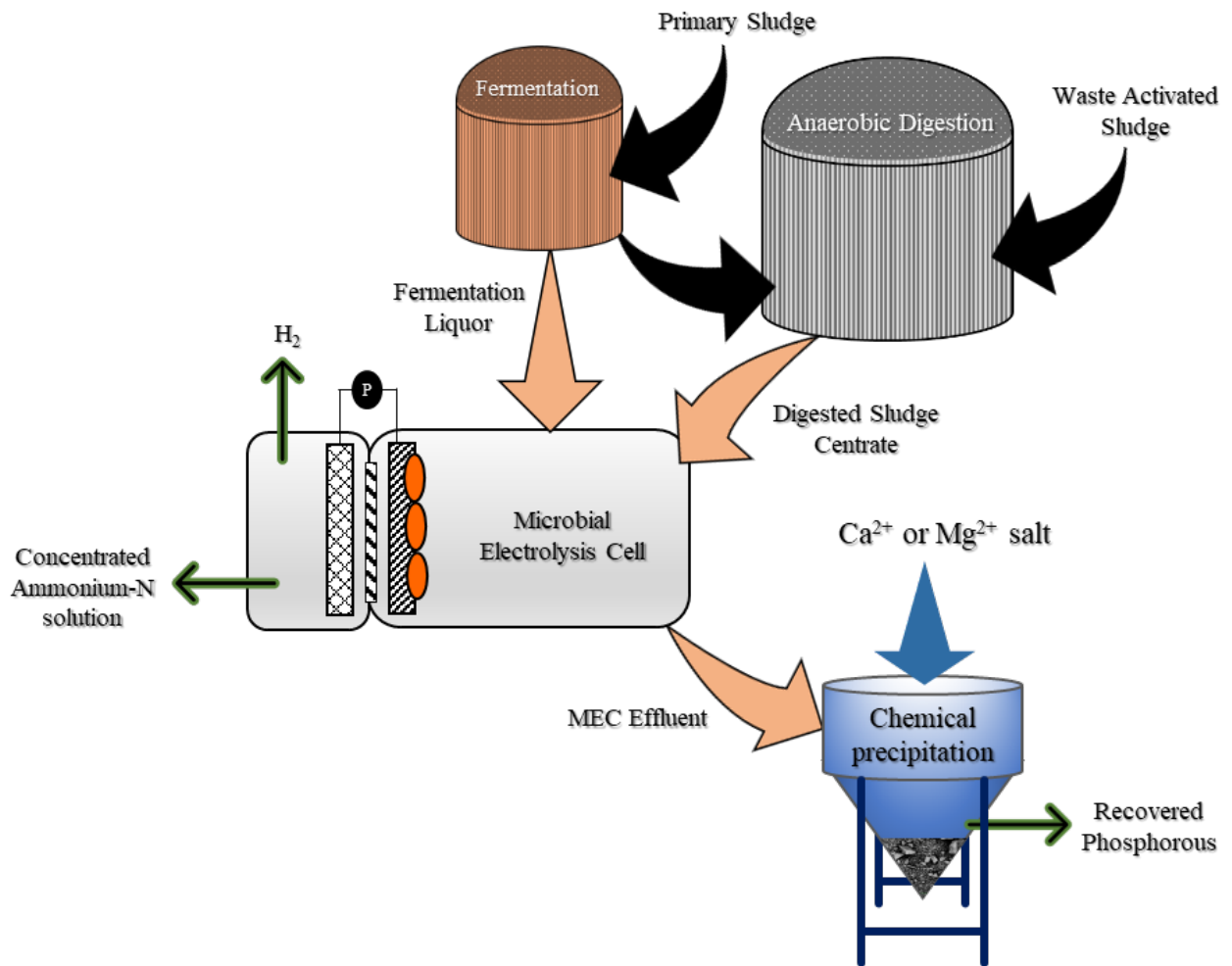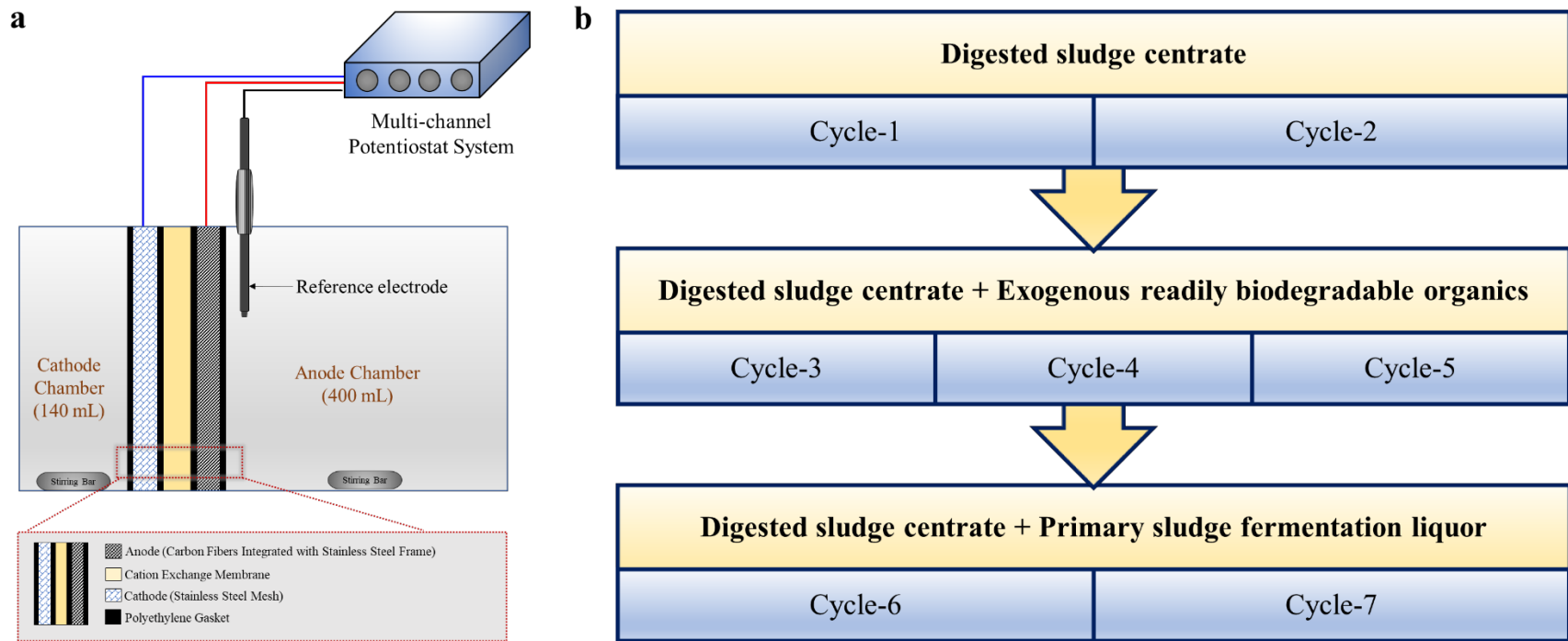
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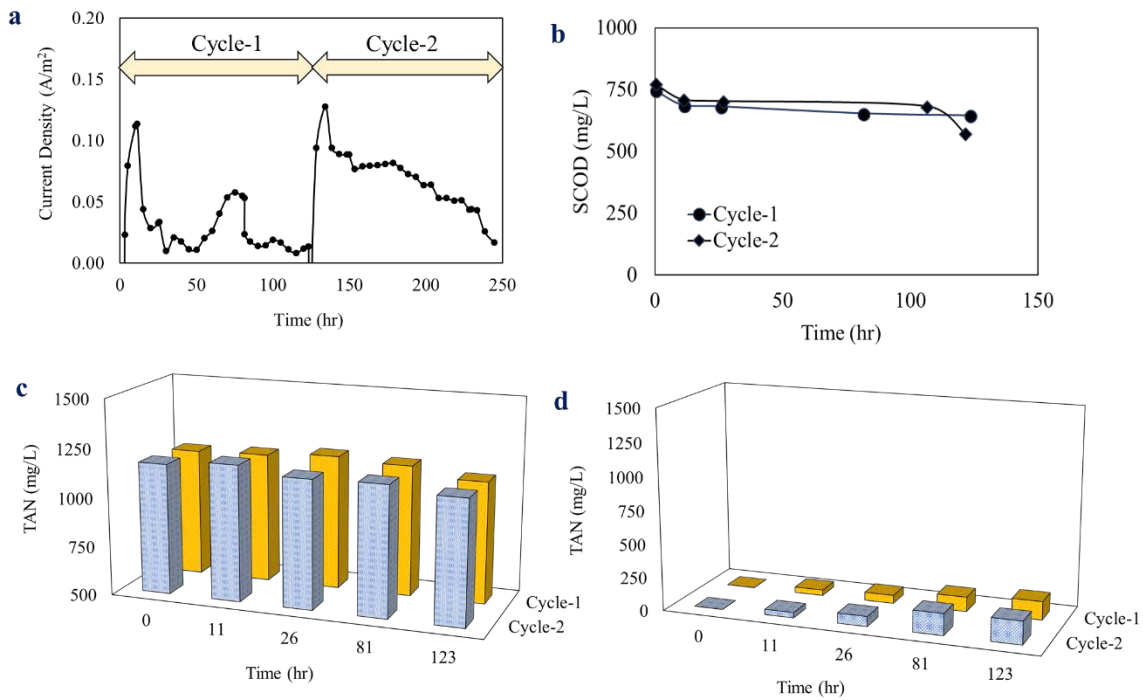
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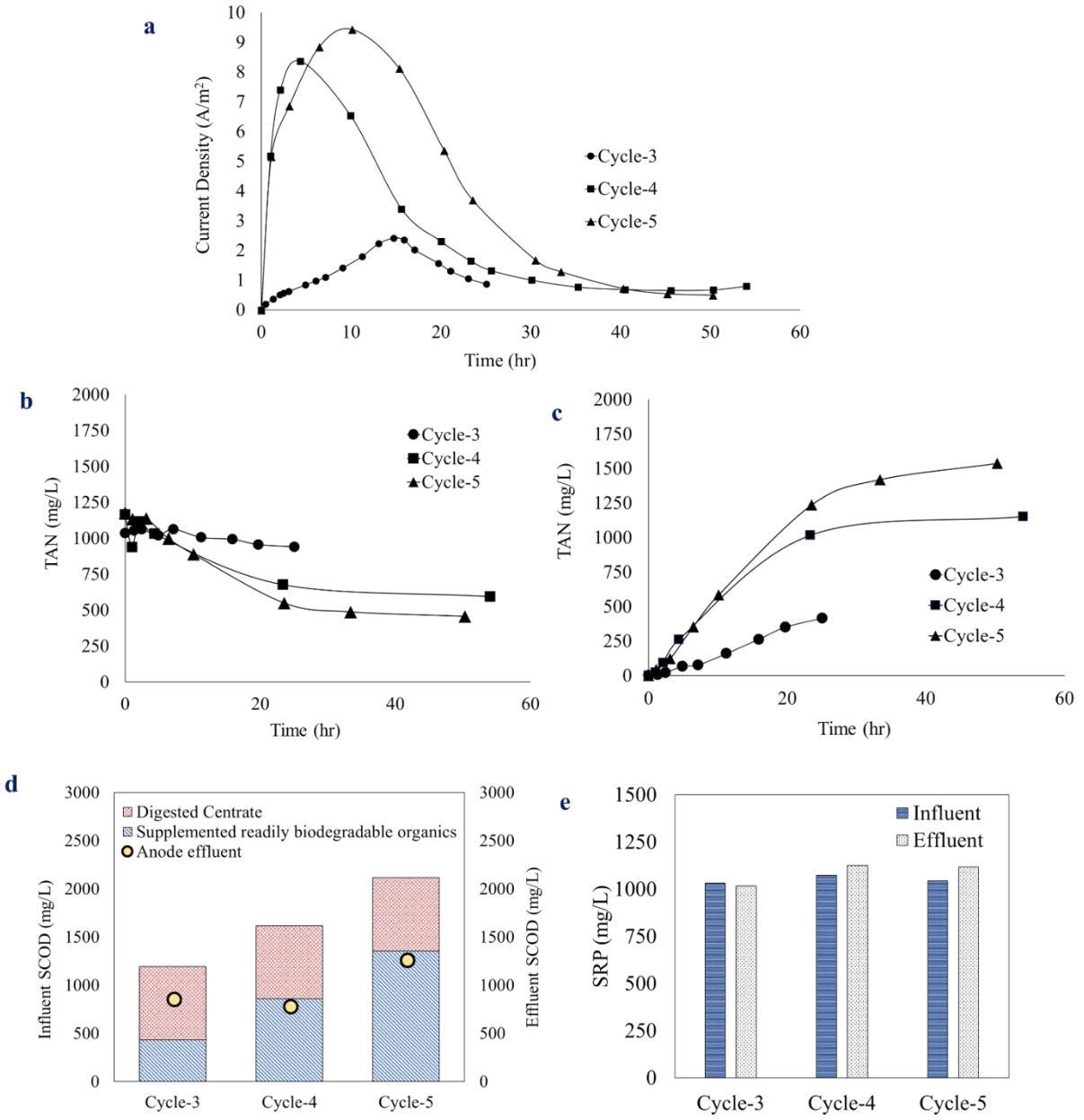
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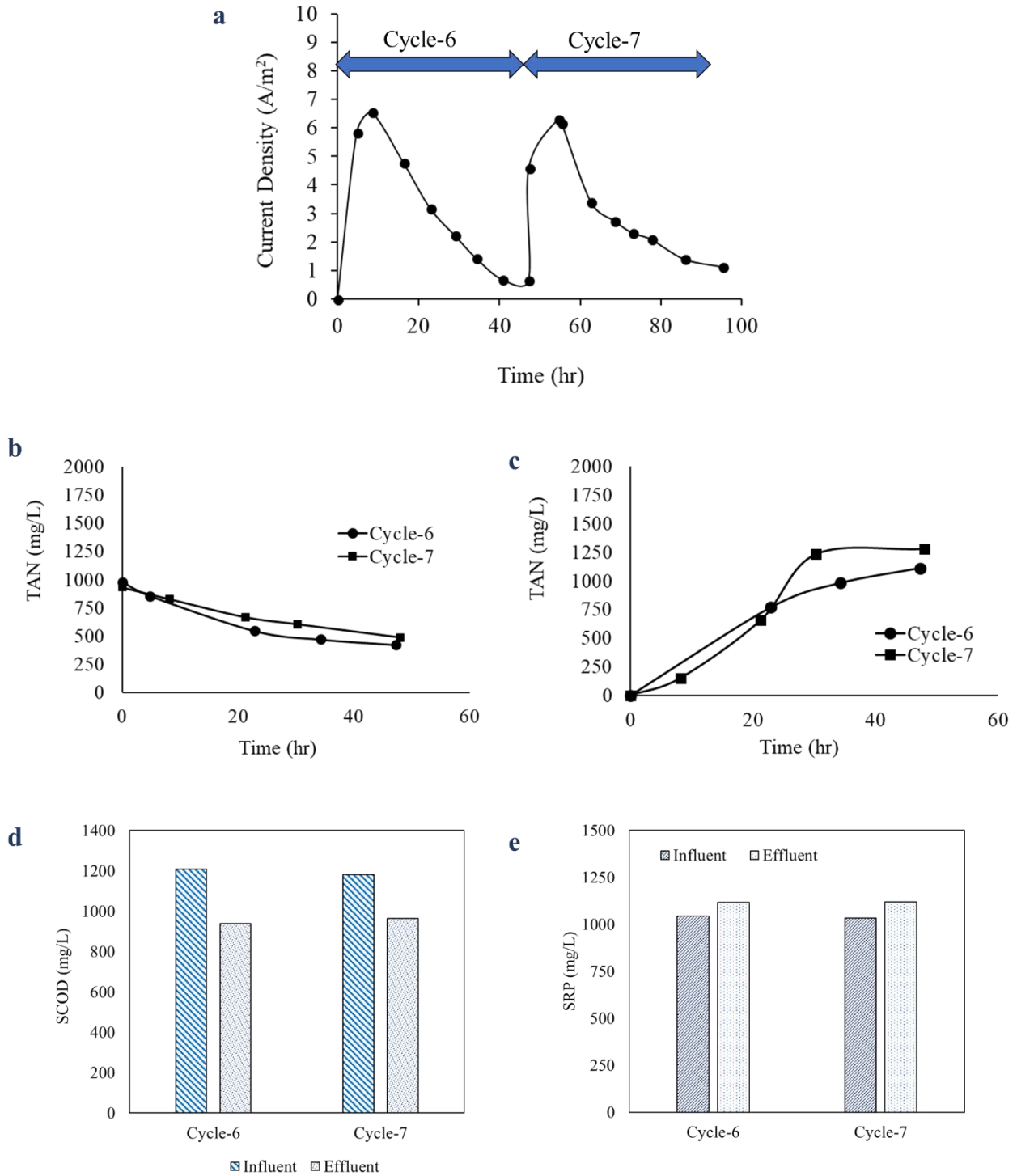
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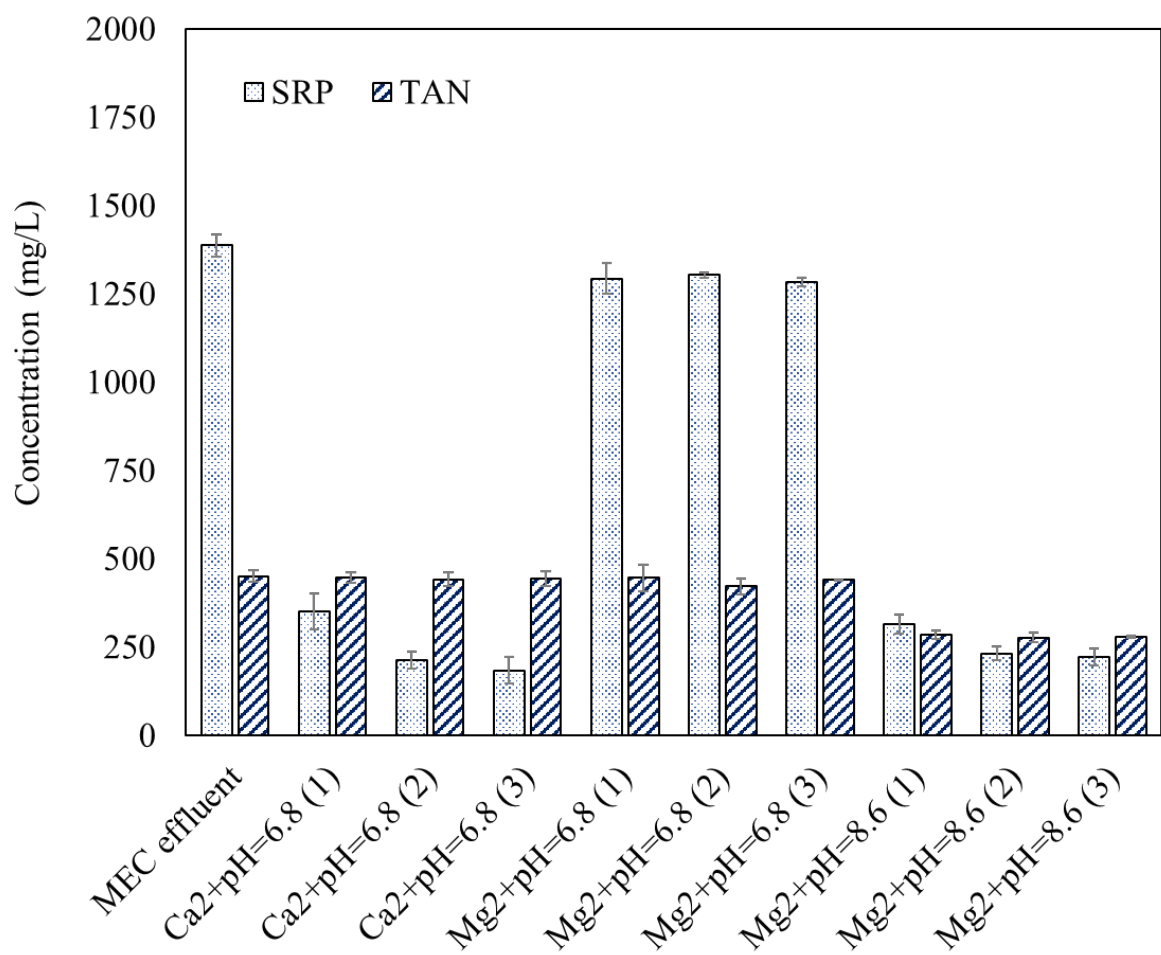
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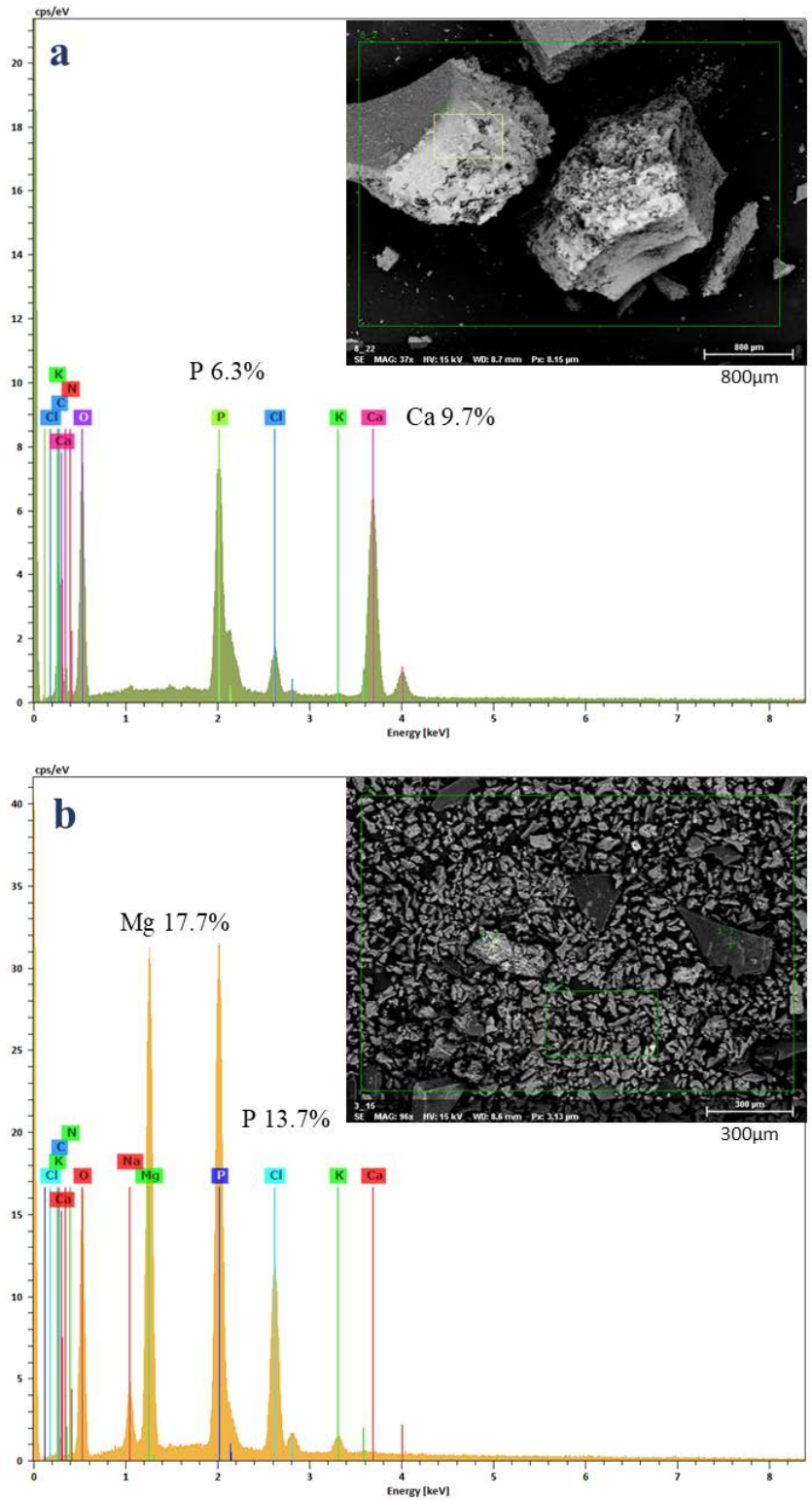
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Table 1. Characteristics of centrate, primary sludge, and fermentation liquor.

Parameter	Digested sludge centrate	Primary sludge	Fermentation liquor
Total chemical oxygen demand, TCOD (mg/L)	2705±130	59130±580	10320±208
Soluble chemical oxygen demand, SCOD (mg/L)	740±6	9770±160	9292±4
Suspended solids, SS (mg/L)	1098±333	41950±1626	1038±265
Total ammonia-nitrogen, TAN (mg/L)	1177±23	530±13	547±5
Total volatile fatty acids, TVFA (mg COD/L)	31±3	3015±465	4605±785
Soluble reactive phosphorus, SRP (mg/L)	1387±51	668±6	447±3
pH	7.9	6.4	5.5

Table 2. Summary of chemical precipitation experiments with Ca^{2+} and Mg^{2+} salts for phosphorous recovery from anodic effluents.

Set	C: P[c]	$CaCl_2.2H_2O$ (g/L)	Mg: P[c]	$MgCl_2.6H_2O$ (g/L)
Ca^{2+} pH=6.8 (1)[a]	1.5:1	3.3	-	-
Ca^{2+} pH=6.8 (2)[a]	3:1	6.6	-	-
Ca^{2+} pH=6.8 (3)[a]	4.5:1	9.9	-	-
Mg^{2+} pH=6.8 (1)[a]	-	-	1:1	3.1
Mg^{2+} pH=6.8 (2)[a]	-	-	2:1	6.2
Mg^{2+} pH=6.8 (3)[a]	-	-	3:1	9.3
Mg^{2+} pH=8.6 (1)[b]	-	-	1:1	3.1
Mg^{2+} pH=8.6 (2)[b]	-	-	2:1	6.2
Mg^{2+} pH=8.6 (3)[b]	-	-	3:1	9.3

[a]Without pH adjustment; [b]pH adjusted to 8.6; [c]moler ratio

Supplementary Information

Microbial Electrolysis Followed by Chemical Precipitation for Effective Nutrients Recovery from Digested Sludge Centrate in WWTPs

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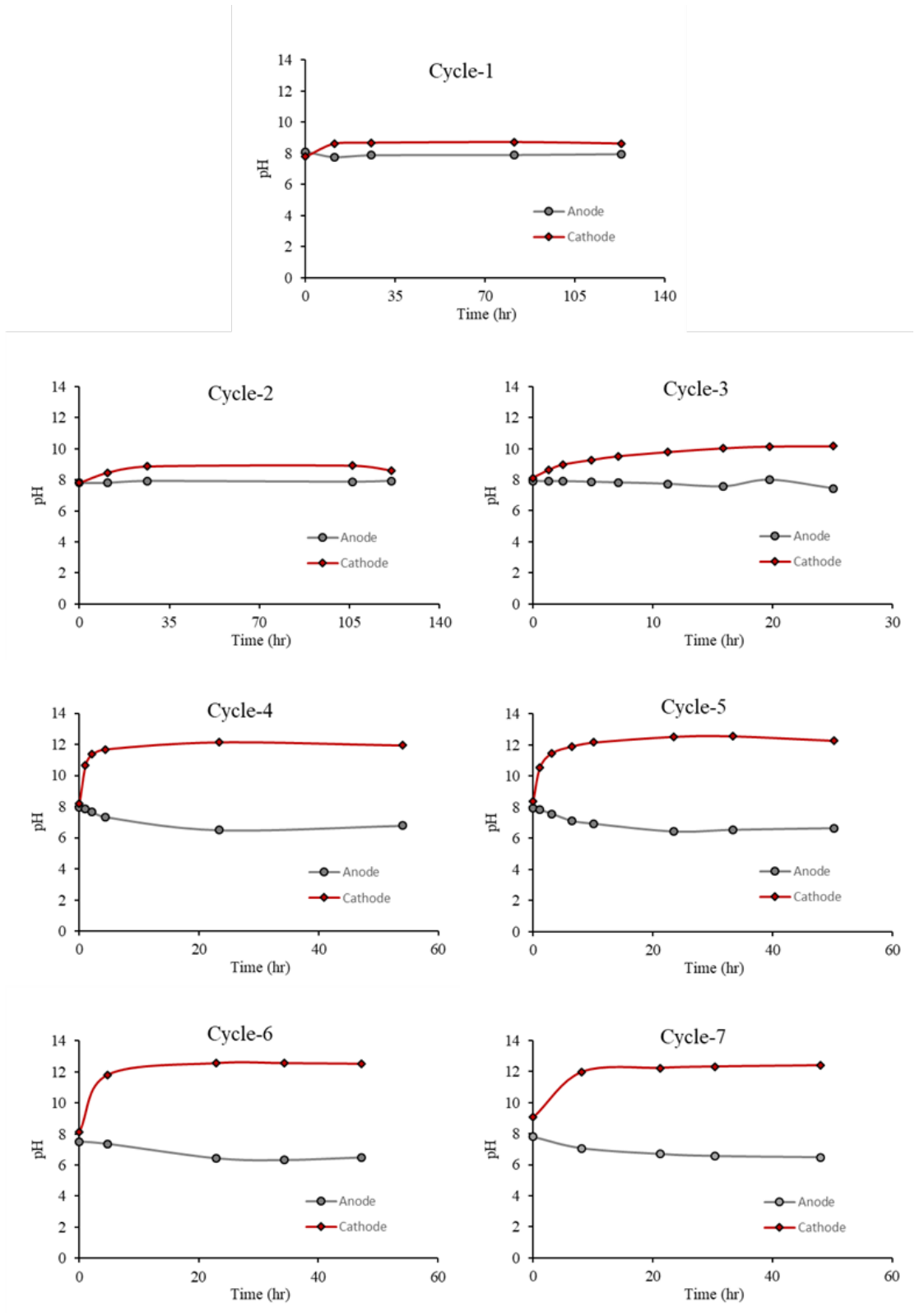
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Fig. S1. Photograph of microbial electrolysis cell (MEC) used in this study.

Fig. S2: pH profiles of anode and cathode chamber during different operating cycles.

Table S1. Summary of pH and conductivity at the end of chemical precipitation experiments with Ca^{2+} and Mg^{2+} salts for phosphorous recovery from MEC effluents.

Set	pH	Conductivity
$Ca^{2+}_{pH=6.8}$ (1)	6.1±0.1	6.89±0.43
$Ca^{2+}_{pH=6.8}$ (2)	5.8±0.1	11.17±0.03
$Ca^{2+}_{pH=6.8}$ (3)	5.7±0.1	15.35±0.04
$Mg^{2+}_{pH=6.8}$ (1)	6.8±0	6.74±0.04
$Mg^{2+}_{pH=6.8}$ (2)	6.7±0	9.23±0.01
$Mg^{2+}_{pH=6.8}$ (3)	6.6±0	11.82±0.15
$Mg^{2+}_{pH=8.6}$ (1)	7.3±0.1	6.15±0.02
$Mg^{2+}_{pH=8.6}$ (2)	7.1±0	8.64±0.01
$Mg^{2+}_{pH=8.6}$ (3)	7±0	11.3±0.01