

1 Impact of lime treatment on tailings dewatering and cap water
2 quality under an oil sands end pit lake scenario

3 Nesma Eltoukhy Allam^a, Nikolas Romaniuk^b, Mike Tate^b, Mohamed N.A. Meshref^{a,c},

4 Bipro R. Dhar^a, Ania C. Ulrich^{a,*}

5 ^aDepartment of Civil and Environmental Engineering, University of Alberta, Edmonton, AB T6G 1H9, Canada

6 ^bGraymont Western Canada Inc., #200-10991 Shellbridge Way, Richmond BC V6X 3C6, Canada

7 ^cPublic Works Department, Faculty of Engineering, Ain Shams University, 1 El Sarayat St., Abbassia, Cairo 11517, Egypt

8

9 **Abstract**

10 Oil sands tailings have a limited ability to dewater and can also release toxic water, so finding a
11 cost-effective tailings reclamation strategy has been challenging. End pit lakes (EPLs) are being
12 examined as one potential approach to reduce fluid fine tailings (FFT) inventories for the oil sands
13 industry. This study investigated the effect of adding lime (650 to 4000 ppm) on FFT dewatering
14 and improving cap water quality under the simulated EPL scenario. The results illustrated that a
15 high lime dose of 3500 ppm achieved the highest FFT water recovery, decreased the cap water
16 alkalinity at the end of the experiment (after 90 d), and increased the possibility of cation exchange
17 (at day 0). In contrast, the degradation of petroleum hydrocarbons was slightly enhanced at low
18 lime dose of 650 ppm in comparison to high doses. In addition, the 650 ppm dose resulted in
19 minimal change in the microbial cell counts at day 90, compared with high lime doses that resulted
20 in a large reduction in the cell counts. Comparisons of toxicity of the FFT pore water after lime
21 addition exhibited higher toxic effects for lime dosages >1600 ppm. Nevertheless, at all lime
22 doses, low cap water toxicity (i.e., <1.0 Toxicity Unit; TU) at day 90 was attained (e.g. for the
23 3500 ppm lime dosage, toxicity was reduced from 0.83 TU (day 14) to 0.58 TU (day 90)). The low
24 water toxicity for the cap water (day 90) can be ascribed to the reduction of cap water pH over
25 time due to the dissolution of atmospheric carbon dioxide into cap water. In this study, the
26 significant implications for the use of lime treatment to improve water quality in EPLs were

27 highlighted. We provided an initial understanding of how lime treatment could benefit the long-
28 term success of FFT remediation within EPLs as self-sustaining aquatic ecosystems in the final
29 reclaimed landscape.

30 **Keywords:** End pit lake (EPL); Oil sands tailings; Fluid fine tailings (FFT); Dewatering; Lime;
31 Cap water quality

32 **1. Introduction**

33 Large volumes of fluid fine tailings (FFT) and oil sands process-affected water (OSPW) are
34 generated from surface mining of oil sands in northern Alberta. The continuous storage of the
35 massive quantities of FFT in engineered tailing ponds (Allen, 2008b; Clemente and Fedorak, 2005;
36 Siddique et al., 2014) requires oil sands operators to use various reclamation strategies to achieve
37 reclamation obligations. Several remediation approaches (e.g., composite tailings, atmospheric
38 drying, etc.) have been studied, however these methods have drawbacks that hinder their
39 applications. Most of the proposed alternatives are either energy-consuming and costly (e.g.,
40 centrifugation; (Wang et al., 2014) or need a large amount of land (e.g., drying requiring freeze-
41 thaw cycles (Proskin et al., 2012) (Allen, 2008a; Board, 2012 ; Hyndman and Sobkowicz, 2010;
42 Vajihinejad and Soares, 2018; Wells and Riley, 2007). Thus, more economical and effective
43 solutions are being investigated to incorporate FFT into oil sands mine closure landscapes
44 (Famakinwa et al., 2018). Water-capped tailings technologies such as end pit lakes (EPLs) are one
45 of those potential approaches to reduce FFT inventories in tailings ponds for subsequent
46 reclamation (Kabwe et al., 2018; Zubot, 2010).

47 EPLs are engineered water bodies created in the post-mining pit by pumping freshwater
48 and OSPW on top of the FFT (Zubot, 2010). EPLs are predicted to slowly develop into self-

49 sustaining aquatic ecosystems (Hrynyshyn, 2012; Zubot, 2010) and act as permanent features in
50 the final reclaimed landscape (Kabwe et al., 2018). In the first field scale demonstration of pit
51 lakes, Base Mine Lake (BML), the FFT was not treated before being deposited into the lake, and
52 the FFT is settling by self-weight consolidation (natural dewatering through gravity settling;
53 (Dompierre, 2016). The FFT (with initial solids content around 10% solids (wt./wt.) as first
54 deposited into tailing ponds) comprises fine clays suspended in water, and the clay particles slowly
55 settle over decades (MacKinnon, 1989).

56 There are key concerns and potential challenges relevant to the environmental development
57 of EPLs (COSIA, 2012). For instance, during the slow settling of FFT, the released pore water
58 contributes OSPW-derived constituents to the EPL cap water (Dompierre et al., 2016a), including
59 the chemical flux of constituents of potential concern (COPCs) (Dompierre and Barbour, 2016;
60 Lawrence et al., 2015). The presence of COPCs (e.g., major inorganic salts, dissolved organic
61 compounds [DOC], naphthenic acids [NAs], and petroleum hydrocarbons [PHCs]) are potentially
62 toxic to aquatic microorganisms (Dompierre et al., 2016a; McQueen et al., 2017; Samadi, 2019;
63 White, 2017; White and Liber, 2018).

64 Earlier studies investigated the influence of vertical mixing (Lawrence et al., 2015) and
65 have used a numerical modeling approach to simulate the movement of COPCs (Dompierre and
66 Barbour, 2016) and identify the key mechanisms controlling mass transport in FFT layers within
67 EPLs. Similarly, White and Liber (2018) monitored the concentrations of inorganic salts in the cap
68 water of BML over three consecutive years and assessed the corresponding toxicological risk of
69 elevated concentrations of inorganic constituents such as sodium (Na^+) and bicarbonate (HCO_3^-)
70 on EPL cap water toxicity (White, 2017; White and Liber, 2018; 2020). The studies reported a
71 sufficient level of *Daphnia*'s populations within the BML surface water with a promising

72 improvement trend of BML surface water quality over time (White and Liber, 2018; 2020).
73 However, this was constrained with the less sensitive toxicity endpoints samples used in the study
74 (White and Liber, 2018) that could necessitate further investigations to claim the development into
75 a robust EPL aquatic ecosystem. Several research groups have looked at the ability of the
76 indigenous microbial community in BML to remediate COPCs (Bowman et al., 2016; Bradford et
77 al., 2017; Richardson et al., 2020; Yu et al., 2019). While investigating methanogenic activity,
78 Samadi et al. (2019) reported an enhancement in FFT dewatering with the addition of
79 hydrocarbons and nutrients due to biogenic gas production. Also, Poon et al. (2018) reported a
80 decrease in the turbidity of the EPL cap water through pH reduction by inducing CO₂. However,
81 that study was focused on improving cap water clarity by enhancing the clay flocculation in BML.

82 Because FFT takes a long time (decades) to settle (Masliyah et al., 2011), coagulation and
83 flocculation were implemented to enhance the dewatering process (Famakinwa et al., 2018; Wang
84 et al., 2015; Wang et al., 2014). Promising results in the acceleration of tailings dewatering were
85 reported using lime or calcium hydroxide [Ca(OH)₂] as a stand-alone coagulant or in combination
86 with gypsum (CaSO₄·2H₂O) or polymers (Baillie and Malmberg, 1969; Beaty and Lane, 1983;
87 Caughill et al., 1993; Chalaturnyk et al., 2002; Ewin et al., 1981; Hamza et al., 1996; Lorentz et
88 al., 2014; Matthews et al., 2002). Lime addition caused an increase in pH and provided divalent
89 calcium ions (Ca²⁺) that improved FFT coagulation (Chalaturnyk et al., 2002; Tate et al., 2017).
90 To date, chemical treatment, particularly lime addition, has not been thoroughly and
91 comprehensively explored in EPL scenarios. Moreover, the effect of the slow dewatering of FFT
92 on the development of EPLs is still unclear (Risacher et al., 2018) and should be investigated
93 further.

94 Based on previous studies, lime addition can be hypothesized to accelerate the dewatering
95 of FFT through cation exchange. The increase in pH resulting from lime addition can reduce
96 methanogenesis in the FFT layer, and therefore reduce the chemical flux of ions and organic
97 constituents to the water cap and enhance the cap water quality in EPLs.

98 Consequently, the key goal of this study is to examine the impact of the addition of various
99 lime doses on FFT dewatering and cap water quality in EPL configurations. The specific objectives
100 are as follows: a) to investigate the feasibility of lime treatment to enhance FFT dewatering in
101 EPLs; b) to evaluate the influence of adding different dosages of lime on the quality of cap water
102 and FFT pore water in EPLs; c) to elucidate the role of cation exchange on the reduction and
103 removal of organic fractions in FFT (DOC and PHCs); and d) to examine the variation of water
104 chemistry (alkalinity, pH, and cations), bacterial cell counts, and toxicity toward *Vibrio fischeri*.

105 **2. Materials and Methods**

106 **2.1 Materials**

107 FFT was collected from an active oil sands tailings pond in northern Alberta. Hydrated
108 lime [Ca(OH)₂] was provided by Graymont Inc. (Calgary, Alberta, Canada). FFT and lime were
109 stored in sealed buckets at 4 °C until use. Raw synthetic OSPW was prepared in 15 L deionized
110 water with: 10.80 g NaHCO₃, 4.35 g Na₂SO₄, 8.70 g NaCl, 0.75 g KCl, 2.10 g MgCl₂·6H₂O, and
111 0.75 g CaCl₂·2H₂O. The specifications for Raw synthetic OSPW were provided by Graymont Inc.,
112 and mimics the solute concentrations in Albian OSPW (Mahdavi, 2014; Poon, 2019). The Raw
113 synthetic OSPW didn't contain any organics to better understand the movement of COPCs from
114 the FFT layer to the cap water.

115 **2.2 Experiment**

116 Four different lime dosages (650, 1600, 3500, and 4000 ppm on a wet weight basis) were
117 used. Lime slurry was prepared by adding hydrated lime to Millipore ultrapure water (18.2 MΩ
118 cm) to achieve a 5% solids slurry (wt./wt.). Each lime dose was added to 1 kg FFT in a glass beaker
119 and mixed with a mechanical agitator at moderate speed (1200 rpm). The initial solids content of
120 the raw FFT was 32.8%. To achieve 50% solids content, both control (no lime) and lime-treated
121 FFT were compressed in an air pressure filter (OFITE; Bench-Mount Filter Press with Hose and
122 Regulator; #140-31) at 100 psi.

123 Experiments were conducted in 1 L glass columns, where 500 ml FFT (50% solids content
124 [wt./wt.]) was placed at the bottom of the column and covered with 500 ml Raw synthetic OSPW.
125 All experiments were conducted at ambient conditions (23°C) using uncovered columns in
126 triplicate, except for the 4000 ppm treatment which had both covered and uncovered conditions.
127 This step was conducted in order to determine the effect of atmospheric CO₂ on the cap water
128 quality and toxicity and to highlight any discrepancies between covered and uncovered conditions.
129 This can be also attributed to the plateau reached at pH 12.5 regardless the increase of lime dose
130 (> 3500 ppm) that requires better understanding about the potential influence of any adverse
131 impact of overdosing.

132 To better understand the effectiveness of lime addition on FFT dewatering, % water
133 recovery (WR) was used as a comparable index in our assessment. The WR was calculated as
134 follows:

$$135 \quad WR\% = \frac{\text{Volume of pore water released at day 90}}{\text{total initial volume of FFT}} \times 100$$

136 The released water from FFT (FFT pore water) at day 0 was collected using the pressure
137 filter, while FFT pore water at day 90 was collected from the FFT samples by centrifugation at
138 8,000 g for 40 min. FFT pore water was collected in 50 ml plastic tubes and stored at 4 °C before
139 further analyses. For cap water analysis, samples were collected directly from the columns every
140 two weeks. The initial FFT and final treated FFT samples at day 90 were collected directly from
141 the columns and stored at -20°C in 50 ml plastic vials for DNA extraction. For PHC analysis, FFT
142 samples were stored at 4 °C until further analysis.

143 **2.3 Chemical analysis**

144 Water samples were analyzed for cations including Ca^{2+} , Na^+ , K^+ , and Mg^{2+} using a
145 Thermo iCAP 6000 series Inductively Coupled Plasma–Optical Emission Spectrometer (ICP-
146 OES, ThermoFisher). A Mettler Toledo DL53 (Mississauga, Canada) was used to measure
147 alkalinity with 0.02 N H_2SO_4 as a titrant, and pH values were measured using an Accumet®
148 Research AR50 (Fisher Scientific, Lenexa, Kansas). DOC was measured using a TOC analyzer
149 (Shimadzu TOC-LCPH, Japan) and a non-purgeable organic carbon method (Brown et al., 2013).
150 Prior to any analysis and unless otherwise specified, liquid samples were filtered with
151 polytetrafluoroethylene (PTFE) syringe filters (Fisher Scientific; 0.22 μm for cation analysis; 0.45
152 μm for DOC analysis). For DOC analysis, all water samples were diluted 10× using ultrapure
153 water.

154 PHCs were analyzed based on the Canada-Wide Standard for Petroleum Hydrocarbons in
155 soil (CCME, 2008). Fractions were grouped as follows: F1 (C6–C10), F2 (C10–C16), F3 (C16–
156 C34), F4 (C34–C50), and F4G-SG (>C50; (CCME, 2008). According to the Dean Stark extraction
157 method (the industry-accepted method) the F4 and F4G-SG fractions were categorized with

158 respect to bitumen content (Dean and Stark, 1920). F1 fractions were non-detectable in all samples
159 and were measured by purge and trap gas chromatography (Column: DB1 0.25 mm ID; 1.0 μ m
160 film; 60 m length). Other fractions (F2, F3, F4, and F4G-SG) were analyzed by Maxxam Analytics
161 (Edmonton, Alberta, Canada). Further details on PHC fractions analysis can be found in the
162 Canada-Wide Standard for Petroleum Hydrocarbons in Soil (CCME, 2008) and elsewhere (Yu et
163 al., 2018a). To determine the statistical differences between the treatment conditions, p-values
164 were calculated using a two tailed t-test in Microsoft Excel. The F-test was initially used to
165 determine the variance among sample pools whether it is equal or unequal. The difference between
166 the treatment conditions is considered significant if p-value < 0.05.

167 **2.4 Toxicity Bioassay**

168 The toxicity of the aqueous samples toward *V. fischeri* was measured with the Microtox[®]
169 bioassay test (Osprey Scientific, Edmonton, Alberta, Canada). Briefly, the 81.9% Basic Test
170 protocol was followed by 5 min and 15 min incubation in the Microtox[®] 500 Analyzer (Azur
171 Environmental time, (Anderson et al., 2011). The light emission was measured and recorded with
172 MicrotoxOmni software to determine the inhibitory concentration resulting in 20% less light
173 emission (IC₂₀) or inhibitory concentration resulting in 50% less light emission (IC₅₀) values.
174 There were no significant differences in the toxicity measured between 5 min and 15 min tests,
175 therefore only the 5 min data is described (Miles et al., 2019). Toxicity units, derived from IC₅₀
176 (TU = 100 \div IC₅₀), were used to represent high-level toxicity trends (Yu et al., 2018a). In previous
177 studies, 1.0 toxicity unit (TU) or lower is considered completely detoxified (Scott et al., 2008; Yu
178 et al., 2018a).

179 Both principal component analysis (PCA) and cluster analysis were employed using
180 Minitab 19 in order to evaluate and highlight any variations between the lime treated conditions
181 and control. Implementing the PCA and cluster analysis in addition to Tukey Pairwise comparisons
182 also supported the determination of clustering, similarity, and significant differences between the
183 treatment conditions (i.e., p-values <0.05 is considered significant different).

184

185 **2.5 DNA Extraction and qPCR analysis**

186 DNA was extracted from the FFT samples (up to 500 mg of FFT used per sample), using
187 the FastDNA™ SPIN Kit of Soil (MP Biomedicals) according to the isolation protocol provided
188 by the supplier. The extraction was conducted at the start and end of the experiment to examine
189 the initial and final microbial population density for all conditions. Microbial cell counts were
190 determined by qPCR amplification of the RNA polymerase beta subunit (*rpoB*) gene, utilizing
191 *rpoB* 1698f (5'-AACATCGGTTTGCTCAAC-3') and *rpoB* 2041r (5'-
192 CGTTGCATGTTGGTACCCAT-3') primers (Brown et al., 2013; Mahdavi et al., 2015; Yu et al.,
193 2018b). Using a Bio-Rad CFX96 optical reaction module conversion for the C1000 Touch thermal
194 cycler, the qPCR assay was performed on DNA extracted samples from the lime treated/ untreated
195 tailings samples. The amplification data were analyzed using Bio-Rad CFX Manager™ 3.0
196 software. All standards and samples were analyzed in triplicate. More details can be found
197 elsewhere (Yu et al., 2018b).

198 **3. Results and discussion**

199 **3.1 Influence of lime treatment on FFT dewatering**

200 The effectiveness of lime addition on the FFT dewatering was assessed based on released
201 water volumes, water recovery, and solids content. The released pore water volumes were 21.7,
202 30.7, 37.7, and 30 ml for the 650, 1600, 3500, and 4000 ppm lime dosages, respectively. Therefore,
203 an increase in the released volume of FFT pore water was observed after lime treatment.

204 To better compare our study with others, the WR was also used as a comparable index. In
205 general, WR increased at day 90 for the lime-treated FFT samples, as illustrated in Table 1. At the
206 3500 ppm dose, the maximum WR was 7.9%. However, increasing the lime dose to 4000 ppm
207 decreased the WR to 6.4%, implying that there is an adverse effect from lime overdosing. This can
208 be attributed to the overdose effect (i.e., 4000 ppm), which might create an extra strength on the
209 solids and resist the self-weight settling. Tate et al., (2017) reported the same notion that increasing
210 lime dosages could lead to an increase in the yield stress of the FFT (i.e., developing strength by
211 lime addition) and hence can enhance the geotechnical properties of FFT. Another explanation
212 could be due to the premature cracking as a result of the reduction of solids percentage happened
213 after lime overdosing (Tate et al., 2017).

214 Comparing our results with previous studies, the 3500 ppm lime treatment achieved a
215 similar WR (7.9%) to Yu et al. (2018a), who reported 8% WR after 100 d using a biological
216 amendment (Ultrazyme). In contrast, Siddique et al. (2014) reported a higher WR of 26% after 90
217 d using hydrolyzed canola meal (i.e., as an organic substrate). For solids content in our study, the
218 highest increase was also observed at a dose of 3500 ppm (the final solids content of the FFT
219 increased to 51.6% at day 90; Table 1). Despite the increase in the solid content, it can be
220 considered marginal. It is worth to note that the initial solids content had a significant impact on
221 the final WR (or the final volume of released water) where the lowest initial solids content led to

222 a high WR (Yu et al., 2018a). The initial solids content in our study was considerably higher (50%)
223 than those used by Siddique et al. (2014); 25%) and Yu et al. (2018a); 34.6%).

224 **3.2 Variations in water chemistry of FFT pore water and cap water**

225 **3.2.1 Alkalinity**

226 As shown in Fig. 1a, pore water alkalinity immediately increased after the addition of high
227 lime doses (day 0; Fig. 1a). For example, after the addition of 1600 or 3500 ppm lime (day 0), the
228 alkalinity increased to 2390 mg/l and 5720 mg/l (measured as CaCO₃), respectively, compared
229 with the control alkalinity of 450 mg/l (measured as CaCO₃). The increased alkalinity at day 0
230 with the higher lime dose could be attributed to the increase of the hydroxide (OH⁻) concentrations,
231 which agrees with Caughill et al. (1993). In contrast, at day 90, the alkalinity sharply decreased at
232 all lime doses, with only a slight change in the control. This could be argued as the pH of the FFT
233 pore water decreased in all the lime-treated columns at day 90 (Fig. S1a). As a result of the
234 dissolution of carbon dioxide CO₂ into the cap water; the pH of the pore water started to decrease,
235 and the formation of carbonate minerals, such as CaCO₃, also reduces the alkalinity (Hrnyshyn,
236 2012).

237 The alkalinity level of the cap water continuously decreased throughout all the lime
238 experiments as well as in the control (Fig. 1b). However, compared with the control after 90 d
239 (315 mg/l measured as CaCO₃), the alkalinity level of the samples treated with 1600 ppm and
240 3500 ppm lime significantly decreased to 170 and 200 mg/l (measured as CaCO₃), respectively.
241 The slight change in cap water alkalinity in the control experiment (20% decrease) could be
242 attributed to methanogenesis in the FFT layer (Dompierre et al., 2016b; Siddique et al., 2014) that
243 can increase and maintain a stable alkalinity level in cap water. Similar observations were

244 reported in BML (White and Liber, 2018) where methanogenesis resulted in elevated and high
245 cap water alkalinity. Based on Fig. S1b, the pH of the control FFT decreased below pH 7.5 at day
246 90. This can be attributed to CO₂ production during methanogenesis (Samadi, 2019; Siddique et
247 al., 2014) or the contribution of atmospheric CO₂ to decrease the pH. Dissolved CO₂ is converted
248 to carbonic acid through hydration and then dissociates to HCO₃⁻ (Li, 2010), which is transported
249 to the cap water through any movement of gas (including CH₄). To confirm such prospect, further
250 investigations are warranted about monitoring of the redox potential for the FFT layer throughout
251 the experiments.

252 An overall decrease in cap water alkalinity in the lime-treated conditions was observed
253 after 90 d, with the highest magnitude of the decrease for the 1600 ppm. For example, the
254 alkalinity decreased after 90 d by 36 %, 58%, and 51% at lime doses of 650, 1600, and 3500 ppm,
255 respectively. The decrease was significantly rapid and sharp in the first 30 d in all lime-treated
256 FFT. A similar trend can be observed for HCO₃⁻ concentrations (Fig. S2). There are two
257 explanations for this decrease. The methanogenesis process may have been negatively influenced
258 by lime addition which inhibited the methanogens (Nyberg et al., 2011). The changes in the
259 microbial community are discussed later (section 3.3). Alternatively, the change in pH after lime
260 addition could have led to the predominance of CO₃²⁻ species (MacKinnon et al., 2001). Although
261 further studies are needed to confirm the underlying mechanisms, these results reveal the positive
262 role of lime addition to maintain an appropriate cap water quality for the aquatic life (White,
263 2017; White and Liber, 2018). The changing pH is correlated with lime dosage, and Fig. S3
264 displays the corresponding effect of lime treatment on the pH of cap water with time.

265 3.2.2 Cations

266 Fig. 2 depicts the change in cation concentrations (Ca^{2+} and Na^+) in the FFT pore water
267 with different lime doses. For the low lime dose (650 ppm; $\text{pH} < 10$), the Ca^{2+} concentration was
268 slightly lower than the control (0 ppm lime) at day 0. Similarly, Ca^{2+} concentrations were slightly
269 higher in the 1600 lime dose at day 0 compared to the control. An immediate sharp increase in
270 the Ca^{2+} concentration at day 0 was observed immediately after the addition of high lime dosages
271 3500 and 4000 ppm ($\text{pH} > 11.5$). At day 90, similar Ca^{2+} concentrations were observed for all
272 lime doses as well as the control. For lime doses 650 and 1600 ppm, there was no significant
273 change in cation concentration ($p > 0.05$), while a significant reduction in cation concentration
274 was observed at high lime doses ($p < 0.05$). The summary of the cation results is presented in
275 Table S1.

276 Elucidating the influence of pH on changes in cations concentrations at days 0 and 90 can
277 lead to a better understanding to propose the mechanisms involved. Briefly, there are two main
278 phases of Ca^{2+} availability based on the consumption of carbonate species. In the first phase, as
279 lime is initially added at day 0, (e.g. $\text{pH} 10$, lime dose 650 ppm), the equilibrium of the
280 bicarbonate-carbonate reaction will be shifted toward CO_3^{2-} . Lime reacts with sodium
281 bicarbonate (NaHCO_3) through water softening reaction to produce sodium hydroxide (NaOH)
282 and insoluble calcium carbonate that precipitate as calcite (CaCO_3 ; (Hamza et al., 1996;
283 MacKinnon et al., 2001). This mechanism explained the lower Ca^{2+} concentrations at day 0 for
284 the 650 ppm lime compared with the control at this pH range (Fig. 2a). This is the dominant
285 reaction pathway up to a pH of approximately 11.5 (lime dose < 1600 ppm), at which the carbonate
286 species are effectively removed. Accordingly, at this phase, no soluble Ca^{2+} is available for cation
287 exchange reactions. It is worth noting that the softening reactions are fast to occur before ion
288 exchange mechanisms. The NaOH produced from water softening reacts with the edges of

289 kaolinite clays, removing hydrogen ions which create new exchange sites that will be occupied
290 by Na^+ . This proposed mechanism can explain the significant reduction in Na^+ concentration at
291 day 0 in the FFT pore water for the 650 ppm compared to control (Fig. 2b). With regards to the
292 second phase (i.e., $\text{pH} > 11.5$, lime dose >1600 ppm), the immediate increase in the Ca^{2+}
293 concentration at day 0 can be attributed to increased solubility of Ca^{2+} from the dissolution of
294 $\text{Ca}(\text{OH})_2$ (MacKinnon et al., 2001; Tate et al., 2016). This provides Ca^{2+} ions that can exchange
295 on mineral surfaces. This can be demonstrated where the soluble Ca^{2+} exchanges and displaces
296 the Na^+ increasing its concentration in the pore water (i.e., Na^+ concentrations are 210, 300, and
297 315 mg/L at 1600, 3500, and 4000 ppm lime dosage respectively).

298 With regards to day 90, we can hypothesize that increasing the $\text{pH} > 12$ (i.e., 3500 ppm
299 and 4000 ppm lime) promote the pozzolanic reaction where soluble Ca^{2+} started to react with
300 dissolved silica and alumina from the clay forming hydrated minerals of calcium aluminate and
301 silicate (Tate et al., 2017; Wang, 2017). This can elucidate the reduction of Ca^{2+} concentration at
302 day 90 for the pore water. Lane (1983) suggested that the pozzolanic reaction can occur with lime
303 addition at high pH in which a stabilization process can occur through lime addition (i.e., the
304 authors suggested coagulation process as a good description of the lime stabilization in soil and
305 tailings). Furthermore, the lime addition can modify particle surface charge and lead to
306 agglomeration and interparticle bonding. In the same sense, the study elucidated the stabilization
307 reactions as the pozzolanic reactions and carbonation reactions.

308 In terms of the cap water, Na^+ concentration was almost stable throughout the experiment
309 at different lime doses (Fig. 3b). In contrast, Ca^{2+} levels decreased in the cap water with increasing
310 lime dose (Fig. 3a). The change in Ca^{2+} concentrations confirms that CO_2 is being absorbed by the
311 system to react with alkaline Ca^{2+} . The Student's t-test results suggested significant differences

312 in Ca^{2+} concentrations at day 90 at high lime doses (>1600 ppm) compared with lower lime doses
313 ($p < 0.05$). Our investigations generally depicted a marginal drop in K^+ concentration for all lime
314 doses throughout the experiment (Fig. S4) that slightly varied with the control. Although this
315 could partially corresponds with the findings of White and Liber (2018), who reported a slight
316 change in K^+ levels in the surface water of BML over three consecutive years, further
317 investigations are warranted to confirm this phenomenon. In summary, confirming cation
318 exchange and pozzolanic reactions occurrence would require a more focused study.

319 **3.3 Variations in bacterial cell counts**

320 The initial indigenous microbial cell counts are represented by the number found in the
321 control sample on day 0 (8.15×10^{10} cells/ μl), and is used to compare the effect of lime treatment
322 on the microorganisms. The addition of lime resulted in an immediate adverse impact on the
323 bacterial cell counts at day 0 after lime addition, and the microbial count was considerably
324 reduced for all lime conditions (Fig. 4). On day 90, the bacterial cell counts were significantly
325 decreased at high lime doses ($\text{pH} > 11.5$), with the greatest reduction of cell counts occurring at
326 4000 ppm lime under covered conditions (4.41×10^4 cells/ μl ; 98% reduction) resulting from the
327 sharp pH increase. In contrast, at a lime dose of 650 ppm ($\text{pH} < 10$) there was a minimal reduction
328 in the microbial cell counts after 90 d, highlighting the adverse impact of increased pH,
329 particularly $\text{pH} > 11.5$, on the cell counts. Overall, there is significant difference in the microbial
330 cell counts for most of the lime-treated FFT (i.e., lime doses > 1600 ppm; $p < 0.05$), but no
331 significant difference in the control or low lime-treated FFT (i.e., $p > 0.05$ for both control and
332 650 ppm lime).

333 Our findings correspond with previous studies that showed a similar adverse impact of a
334 rapid increase in pH (pH > 12) on bacterial cell membranes, which inhibited their growth (Burns
335 and Gremminger, 1994; Nyberg et al., 2011; Wong et al., 2001; Wong and Selvam, 2009).
336 Furthermore, the addition of lime (pH 11) was demonstrated to effectively inactivate pathogens
337 in municipal biosolids (Bennett et al., 2003).

338 **3.4 Impact of lime treatment on organic fractions**

339 To determine the impact of lime treatment on organic fractions, we characterized both
340 PHCs and DOC. Fig. 5a shows the concentrations of PHC fractions after different lime treatments
341 at day 90 compared with the initial concentration in untreated FFT (control) at day 0 and day 90.
342 The greatest reduction in PHC concentrations is observed at 650 ppm lime, with removal
343 efficiencies of 54%, 57%, 60%, and 66% for F2, F3, F4, and bitumen, respectively. However, for
344 lime doses of 1600, 3500, and 4000 ppm, the PHC reductions were minimal compared with the
345 reduction in the control sample after 90 d. The high bacterial cell counts in the sample treated
346 with 650 ppm lime could explain the better degradation of PHCs compared with the degradation
347 in the high lime doses (>1600 ppm).

348 As stated earlier, at a lime dose of 650 ppm, Ca^{2+} reacts with carbonate and precipitates as
349 CaCO_3 (Hamza et al., 1996; MacKinnon et al., 2001). In comparison to control, both F4 and
350 bitumen fractions (i.e., the heaviest fractions of PHCs) might complex with calcite precipitates
351 (CaCO_3) and then settle down. Nevertheless, the F2 fraction (i.e., demonstrated slight reduction,
352 Fig. 5a), which is the highly volatile fraction of PHC (Brickner, 2013), seems to solubilize from
353 the bitumen and dissolve in the FFT pore water.

354 Our findings highlight the negative influence of high doses of lime on microbial cell counts
355 that subsequently affect the degradation of the PHCs. Various studies have reported the
356 effectiveness of various microbial communities such as hydrocarbon-degrading microorganisms,
357 sulfate-reducing bacteria, and iron-reducing bacteria on the degradation of PHCs (Allen et al.,
358 2007; Pandey et al., 2009). Similarly, Yu et al. (2018a) suggested that the hydrocarbons present
359 in tailings serve as a carbon source for the indigenous microorganisms. These findings can explain
360 the reduction of the PHC fractions in the control columns after 90 d.

361 The variations in DOC concentrations in the pore water throughout the experiment are
362 illustrated in Fig. 5b. No significant change was observed between day 0 and day 90 at 650 ppm;
363 however, at lime doses of 1600, 3500, and 4000 ppm, a significant increase in DOC was
364 observed. The changes in DOC agree with the minimal decrease in the PHC fraction; in which
365 at high lime doses, bacterial cell numbers were decreased and were not able to degrade either the
366 PHC or DOC. Additionally, during cation exchange when $\text{pH} > 11.5$, the F2 fraction might
367 dissolve in the FFT pore water and increase the DOC concentration. This mechanism is similar
368 to one proposed by Fine et al. (1997) that highly volatile fractions of PHCs can dissolve into soil
369 pore water or groundwater. The increase in DOC may also be ascribed to the release of organic
370 compounds into the cap water during FFT dewatering (Yu et al., 2018a).

371 **3.5 Toxicity and implications of pH influence**

372 Table 2 summarizes the impact of lime addition on the toxicity to *V. fischeri* of the FFT
373 pore water (days 0 and 90) and cap water (day 14 and day 90). High lime doses (i.e., > 1600 ppm)
374 caused higher toxicity of the FFT pore water (above instrument detection limit) compared with
375 the lower toxicity in the control and in the 650 ppm lime dose (0.1 and 1.2 TU, respectively).

376 These results are consistent with the negative impact of lime addition (i.e., > 1600 ppm) on
377 bacterial cell counts measured in the FFT.

378 With respect to the acute toxicity effect of the cap water to *V. fischeri* over time (Table 2),
379 for lime doses ≤ 1600 ppm (pH ≤ 11.5), the cap water at day 14 was non-toxic after lime addition.
380 At high lime doses (3500 and 4000 ppm; pH > 11.5), the toxicity at day 14 slightly increased to
381 0.8 and 1.1 TU at 3500 and 4000 ppm, respectively, then decreased at day 90 to 0.6 and 0.4 TU
382 for 3500 and 4000 ppm, respectively.

383 Even though the lime treated samples (uncovered conditions after 90 days) in comparison
384 to the control (Table 2), showed a slight increase in toxicity; the toxicity of the cap water for all
385 lime doses did not exceed 1 TU (Table 2). This translates as a nontoxic effect toward *Vibrio*
386 *Fischeri*. To confirm our observations, principal component analysis (PCA, Fig. S5) and cluster
387 analysis (Fig. S6) were assessed to pinpoint the influence of lime treatment on cap water toxicity
388 after 14 and 90 days. The PCA data support very minimal variations and high similarity levels
389 (i.e., statistically confirmed as insignificant different at 95% confidence interval and significance
390 level of 0.05) between the toxicity effect of lime treated samples and control samples in cap water.

391 It is worth noting that our rationale was to monitor the toxicity of the cap water after FFT
392 lime treatment which necessitate using the synthetic OSPW in the experiment without any
393 organics to better understand the movement of COPCs. According to our earlier observations
394 about the adverse effects on WR occurred due to overdosing (i.e., 4000 ppm), we have decided
395 to elucidate the potential influence of any adverse impact of overdosing or discrepancies between
396 covered and uncovered conditions with regards to toxicity. For the 4000 ppm covered condition,
397 a significant increase in toxicity occurred at both days 14 and 90 (14.83 and 16.13 TU,

398 respectively). This can be initially attributed to the significant effect on pH at high lime doses, as
399 shown in Fig. S3. Although at day 14, both of the 4000 ppm conditions (covered and uncovered)
400 had a similar pH (≈ 9.8), the TUs were quite different (1.1 TU for the uncovered condition; 14.8
401 TU for the covered condition). Therefore, high toxicity is not only related to the pH increase in
402 the covered condition, but also possibly to COPCs moving from the FFT into the cap water. These
403 results highlight the significant impact of atmospheric CO₂ to reduce cap water pH and generate
404 a carbonate buffer in the uncovered conditions.

405 In summary, the slight increase in toxicity of cap water samples can be related to the
406 movements of COPCs from FFT to the released pore water and cap water. This would necessitate
407 more toxicological investigations to examine the critical concentrations, species, and the
408 environmental impact of lime treatment on live-dead viability assays. This would develop better
409 understanding of the influence of end pit lake surface water on acute toxicity and in vitro assays
410 (e.g. RAW 264.7 mouse macrophage cell line and the response of bone marrow-derived
411 macrophages).

412 **4. Conclusions**

413 While mimicking the EPL approach, this study investigated lime application to control the
414 water quality in EPLs and aid the dewatering of FFT from oil sands operations. In terms of the
415 better performance of WR, alkalinity of cap water, and cation exchange, lime dose of 3500 ppm
416 achieved 7.9% WR despite the relatively high initial solids content of FFT (50% wt./wt.). The
417 alkalinity of the cap water decreased after adding lime dose of 3500 ppm, compared with the
418 untreated FFT, promoting and maintaining an appropriate, low alkalinity environment for an
419 aquatic ecosystem in EPLs. Furthermore, the cation concentrations indicate that cation exchange
420 occurs at high lime doses (pH > 11.5), which aids in clay settling. However, high lime doses (1600,

421 3500, and 4000 ppm; pH > 10) had a negative impact on microbial cell numbers in the underlying
422 FFT, resulting in minimal reduction of organics (PHC) at high lime doses compared with moderate
423 reduction at a low lime dose (650 ppm). Assessing the toxicity of the FFT pore water after lime
424 treatment demonstrated high toxicity effects to *V. fischeri* because of the increased pH.
425 Nevertheless, the toxicity of the cap water was not affected by the high toxicity of the FFT, except
426 under the covered conditions. This highlights a considerable influence of atmospheric CO₂ on
427 regenerating the carbonate buffer and reducing pH of the cap water over time. As such any pH
428 transport from the FFT will rapidly be neutralized due to the associated impact of dissolution of
429 atmospheric CO₂ into the cap water. Overall, the performance metrics and indicative parameters
430 (e.g. increase the WR% from the FFT, increase the possibility of the cation exchange of Ca²⁺ with
431 the clay particles, and decrease the alkalinity of the cap water) revealed the role of lime addition
432 towards the reclamations benefits within EPLs. Future work is still required to build upon the
433 current findings in order to further optimize the lime treatment and the occurrence of cation
434 exchange. The movement of COPCs into the cap water should be further investigated with a focus
435 on chemical characterization of the FFT organic fractions and their associated potential risks
436 toward the aquatic life in the EPLs. This better understanding will infer how to proceed to larger-
437 scale and successful applications of lime treatment in EPLs.

438 **Conflict of interest**

439 The authors declare that there is no conflict of interest regarding the publication of this
440 research manuscript.

441 **Acknowledgments**

442 The authors acknowledge the financial support of Natural Sciences and Engineering
443 Research Council of Canada (NSERC) Engage grant (EGP 515022-17). This research was also

444 supported by research grants including NSERC Collaborative Research and Development Grants
445 and NSERC scholarship (N.E.A.). The authors truly appreciate the technical support from
446 Graymont and support from their Research Team (Calgary, AB). Also, the authors acknowledge
447 the help and support from the technicians and the lab members of Dr. Ulrich's research group in
448 the Department of Civil and Environmental Engineering at the University of Alberta.

449 **Authorship contribution and Credit Author Statement**

450 **Nesma Eltoukhy Allam:** Investigation; Formal analysis; Methodology; Data curation;
451 Writing. **Nikolas Romaniuk:** Investigation; Methodology, Reviewing & Editing. **Mike Tate:**
452 Conceptualization; Project Administration; Funding acquisition; Reviewing. **Mohamed N. A.**
453 **Meshref:** Formal analysis, Data curation and data visualization, Reviewing & Editing. **Bipro R.**
454 **Dhar:** Supervision, Reviewing & Editing. **Ania C. Ulrich:** Supervision; Conceptualization;
455 Resources; Funding acquisition; Reviewing & Editing.

456 **References**

457 Allen, E.W. 2008a. Process water treatment in Canada's oil sands industry: II. A review of emerging
458 technologies. *Journal of Environmental Engineering and Science* 7(5), 499-524.
459 Allen, E.W. 2008b. Process water treatment in Canada's oil sands industry: I. Target pollutants and
460 treatment objectives. *Journal of Environmental Engineering and Science* 7(2), 123-138.
461 Allen, J.P., Atekwana, E.A., Atekwana, E.A., Duris, J.W., Werkema, D.D. and Rossbach, S. 2007. The
462 microbial community structure in petroleum-contaminated sediments corresponds to geophysical
463 signatures. *Appl. Environ. Microbiol.* 73(9), 2860-2870.
464 Anderson, J.C., Wiseman, S.B., Wang, N., Moustafa, A., Perez-Estrada, L., Gamal El-Din, M., Martin,
465 J.W., Liber, K. and Giesy, J.P. 2011. Effectiveness of Ozonation Treatment in Eliminating
466 Toxicity of Oil Sands Process-Affected Water to *Chironomus dilutus*. *Environ. Sci. Technol.* 46(1),
467 486-493.
468 Baillie, R.A. and Malmberg, E.W. 1969 Removal of clay from the water streams of the hot water process
469 by flocculation, Google Patents.
470 Beaty, N.B. and Lane, M.D. 1983. The polymerization of acetyl-CoA carboxylase. *Journal of Biological*
471 *Chemistry* 258(21), 13051-13055.
472 Bennett, D.D., Higgins, S.E., Moore, R.W., Beltran, R., Caldwell, D.J., Byrd, J.A. and Hargis, B.M. 2003.
473 Effects of lime on *Salmonella enteritidis* survival in vitro. *Journal of applied poultry research* 12(1),
474 65-68.
475 Board, E.R.C. 2012 Energy Resources Conservation Board Tailings Management Assessment Report: Oil
476 Sands Mining Industry , (posted date: June 12, 2013), from Alberta Energy Regulator, Accessed online

477 April 2017: <http://osipfiles.alberta.ca/datasets/158/TailingsManagementAssessmentReport2011-2012.pdf>.

478 Bowman, D.T., Arriaga, D., Morris, P., Risacher, F., Warren, L.A., McCarry, B.E. and Slater, G. 2016.

479 Chemical fingerprinting of naphthenic acids at an oil sands end pit lake by comprehensive two-

480 dimensional gas chromatography/time-of-flight mass spectrometry (GC× GC/TOFMS). AGUFM

481 2016, H23C-1563.

482 Bradford, L.M., Ziolkowski, L.A., Goad, C., Warren, L.A. and Slater, G.F. 2017. Elucidating carbon

483 sources driving microbial metabolism during oil sands reclamation. *Journal of environmental*

484 *management* 188, 246-254.

485 Brickner, H. 2013. The mobility of petroleum hydrocarbons in Athabasca oil sands tailings.

486 Brown, L.D., Pérez-Estrada, L., Wang, N., El-Din, M.G., Martin, J.W., Fedorak, P.M. and Ulrich, A.C.

487 2013. Indigenous microbes survive in situ ozonation improving biodegradation of dissolved

488 organic matter in aged oil sands process-affected waters. *Chemosphere* 93(11), 2748-2755.

489 Burns, H. and Gremminger, L. 1994 Lime and fly ash stabilization of wastewater treatment sludge, Google

490 Patents.

491 Caughill, D.L., Morgenstern, N.R. and Scott, J.D. 1993. Geotechnics of nonsegregating oil sand tailings.

492 *Canadian Geotechnical Journal* 30(5), 801-811.

493 CCME 2008. CANADA-WIDE STANDARD FOR PETROLEUM HYDROCARBONS (PHC) IN SOIL.

494 Chalaturnyk, R.J., Don Scott, J. and Özüm, B. 2002. MANAGEMENT OF OIL SANDS TAILINGS.

495 *Petroleum Science and Technology* 20(9-10), 1025-1046.

496 Clemente, J.S. and Fedorak, P.M. 2005. A review of the occurrence, analyses, toxicity, and biodegradation

497 of naphthenic acids. *Chemosphere* 60(5), 585-600.

498 COSIA 2012 Technical Guide for Fluid Fine Tailings Management pp. Report, 1-131., Canadian Oil Sands

499 Innovation Alliance, Edmonton, AB. August 2012, Edmonton, Alberta, Canada.

500 Dean, E.W. and Stark, D.D. 1920. A Convenient Method for the Determination of Water in Petroleum and

501 Other Organic Emulsions. *Industrial & Engineering Chemistry* 12(5), 486-490.

502 Dompierre, K.A. and Barbour, S.L. 2016. Characterization of physical mass transport through oil sands

503 fluid fine tailings in an end pit lake: a multi-tracer study. *Journal of Contaminant Hydrology* 189,

504 12-26.

505 Dompierre, K.A., Lindsay, M.B.J., Cruz-Hernández, P. and Halferdahl, G.M. 2016a. Initial geochemical

506 characteristics of fluid fine tailings in an oil sands end pit lake. *Science of The Total Environment*

507 556(Supplement C), 196-206.

508 Dompierre, K.A., Lindsay, M.B.J., Cruz-Hernández, P. and Halferdahl, G.M. 2016b. Initial geochemical

509 characteristics of fluid fine tailings in an oil sands end pit lake. *Science of The Total Environment*

510 556, 196-206.

511 Dompierre, K.A.H. 2016. Controls on mass and thermal loading to an oil sands end pit lake from

512 underlying fluid fine tailings.

513 Ewin, G.J., Erno, B.P. and Hepler, L.G. 1981. Clay chemistry: Investigation of thermodynamics of ion

514 exchange reactions by titration calorimetry. *Canadian Journal of Chemistry* 59(20), 2927-2933.

515 Famakinwa, T.E., Su, Y., Wang, J. and Gates, I.D. 2018. An in-situ process to consolidate oil sands mine

516 tailings. *Journal of Environmental Chemical Engineering* 6(2), 3295-3305.

517 Ferris, A.P. and Jepson, W.B. 1975. The exchange capacities of kaolinite and the preparation of homoionic

518 clays. *Journal of Colloid and Interface Science* 51(2), 245-259.

519 Fine, P., Graber, E.R. and Yaron, B. 1997. Soil interactions with petroleum hydrocarbons: abiotic

520 processes. *Soil Technology* 10(2), 133-153.

521 Hamza, H.A., Stanonik, D.J. and Kessick, M.A. 1996. Flocculation of lime-treated oil sands tailings. *Fuel*

522 75(3), 280-284.

523 Hrynshyn, J. 2012. End pit lakes guidance document 2012. Cumulative Environmental Management

524 Association, Fort McMurray, Alberta. CEMA Contract (2010-0016).

525 Hyndman, A. and Sobkowicz, J. 2010, Oil sands tailings: reclamation goals and the state of technology, pp.

526 12-16.

527 Kabwe, L.K., Scott, J.D., Beier, N.A., Wilson, G.W. and Jeeravipoolvarn, S. 2018. Environmental
528 Implications of End Pit Lakes at Oil Sands Mine. *Environmental Geotechnics*, 1-25.

529 Lawrence, G.A., Tedford, E.W. and Pieters, R. 2015. Suspended solids in an end pit lake: potential mixing
530 mechanisms. *Canadian Journal of Civil Engineering* 43(3), 211-217.

531 Li, C. (2010) Methanogenesis in oil sands tailings. [electronic resource] : an analysis of the microbial
532 community involved and its effects on tailings densification, 2010.

533 Lorentz, J., Bara, B., Mikula, R., Leneve, E. and Wang, N. 2014 Oil sands fluid fine tailings dewatering
534 using rim ditching, Google Patents.

535 Ma, C. and Eggleton, R.A. 1999. Cation exchange capacity of kaolinite. *Clays and Clay minerals* 47(2),
536 174-180.

537 MacKinnon, M.D. 1989. Development of the tailings pond at Syncrude's oil sands plant: 1978–1987.
538 *AOSTRA J. Res* 5(2), 109-133.

539 MacKinnon, M.D., Matthews, J.G., Shaw, W.H. and Cuddy, R.G. 2001. Water quality issues associated
540 with composite tailings (CT) technology for managing oil sands tailings. *International Journal of*
541 *Surface Mining, Reclamation and Environment* 15(4), 235-256.

542 Mahdavi, H. (2014) THE APPLICATION OF ALGAE FOR THE REMOVAL OF METALS AND
543 NAPHTHENIC ACIDS FROM OIL SANDS TAILINGS POND WATER.

544 Mahdavi, H., Prasad, V., Liu, Y. and Ulrich, A.C. 2015. In situ biodegradation of naphthenic acids in oil
545 sands tailings pond water using indigenous algae–bacteria consortium. *Bioresource technology*
546 187, 97-105.

547 Masliyeh, J.H., Xu, Z. and Czarnecki, J.A. (2011) Handbook on theory and practice of bitumen recovery
548 from Athabasca oil sands, Kingsley Knowledge Pub.

549 Matthews, J.G., Shaw, W.H., MacKinnon, M.D. and Cuddy, R.G. 2002. Development of composite
550 tailings technology at Syncrude. *International Journal of Surface Mining, Reclamation and*
551 *Environment* 16(1), 24-39.

552 McQueen, A.D., Kinley, C.M., Hendrikse, M., Gaspari, D.P., Calomeni, A.J., Iwinski, K.J., Castle, J.W.,
553 Haakensen, M.C., Peru, K.M. and Headley, J.V. 2017. A risk-based approach for identifying
554 constituents of concern in oil sands process-affected water from the Athabasca Oil Sands region.
555 *Chemosphere* 173, 340-350.

556 Miles, S.M., Hofstetter, S., Edwards, T., Dlusskaya, E., Cologgi, D.L., Gänzle, M. and Ulrich, A.C. 2019.
557 Tolerance and cytotoxicity of naphthenic acids on microorganisms isolated from oil sands process-
558 affected water. *Science of The Total Environment* 695, 133749.

559 Nyberg, K.A., Vinnerås, B., Lewerin, S.S., Kjellberg, E. and Albiñ, A. 2011. Treatment with Ca(OH)₂
560 for inactivation of *Salmonella Typhimurium* and *Enterococcus faecalis* in soil contaminated with
561 infected horse manure. *Journal of Applied Microbiology* 110(6), 1515-1523.

562 Pandey, J., Chauhan, A. and Jain, R.K. 2009. Integrative approaches for assessing the ecological
563 sustainability of in situ bioremediation. *FEMS Microbiology Reviews* 33(2), 324-375.

564 Poon, H.Y. 2019 An Examination of the Effect of Diluent on Microbial Dynamics in Oil Sands Tailings
565 and the Mechanistic Insight on Carbon Dioxide-mediated Turbidity Reduction in Oil Sands Surface
566 Water.

567 Poon, H.Y., Brandon, J.T., Yu, X. and Ulrich, A.C. 2018. Turbidity Mitigation in an Oil Sands Pit Lake
568 through pH Reduction and Fresh Water Addition. *Journal of Environmental Engineering* 144(12),
569 04018127.

570 Proskin, S., Segó, D. and Alostaz, M. 2012. Oil sands MFT properties and freeze-thaw effects. *Journal of*
571 *Cold Regions Engineering* 26(2), 29-54.

572 Richardson, E., Bass, D., Smirnova, A., Paoli, L., Dunfield, P. and Dacks, J.B. 2020. Phylogenetic
573 estimation of community composition and novel eukaryotic lineages in Base Mine Lake: an oil
574 sands tailings reclamation site in Northern Alberta. *Journal of Eukaryotic Microbiology* 67(1), 86-
575 99.

576 Risacher, F.F., Morris, P.K., Arriaga, D., Goad, C., Nelson, T.C., Slater, G.F. and Warren, L.A. 2018. The
577 interplay of methane and ammonia as key oxygen consuming constituents in early stage

578 development of Base Mine Lake, the first demonstration oil sands pit lake. *Applied Geochemistry*
579 93, 49-59.

580 Samadi, N. 2019. Partitioning of inorganic contaminants between fluid fine tailings and cap water under
581 end pit lake scenario: Biological, Chemical and Mineralogical processes.

582 Scott, A.C., Zubot, W., MacKinnon, M.D., Smith, D.W. and Fedorak, P.M. 2008. Ozonation of oil sands
583 process water removes naphthenic acids and toxicity. *Chemosphere* 71(1), 156-160.

584 Siddique, T., Kuznetsov, P., Kuznetsova, A., Arkell, N., Young, R., Li, C., Guigard, S.E., Underwood, E.
585 and Foght, J.M. 2014. Microbially-accelerated consolidation of oil sands tailings. Pathway I:
586 changes in porewater chemistry. *Frontiers in microbiology* 5, 106.

587 Tate, M., Leikam, J., Fox, J. and Romaniuk, N. 2017 Use of calcium hydroxide as a coagulant to improve
588 oil sands tailings treatment. *Proceedings Tailings and Mine Waste 2017, Banff, Alberta, Canada,*
589 November 2017.

590 Tate, M., Leikam, J., ScotT, J.D., Mehranfar, M., Romaniuk, N. and Ozum, B. 2016 Impacts of Calcium
591 Compounds on Oil Sands Water Chemistry. Conference presentation at the Mining and Tailings
592 Waste Conference, keystone, Colorado, USA.

593 Vajihinejad, V. and Soares, J.B.P. 2018. Monitoring polymer flocculation in oil sands tailings: A
594 population balance model approach. *Chemical Engineering Journal* 346, 447-457.

595 Wang, C., Alpatova, A., McPhedran, K.N. and Gamal El-Din, M. 2015. Coagulation/flocculation process
596 with polyaluminum chloride for the remediation of oil sands process-affected water: Performance
597 and mechanism study. *Journal of Environmental Management* 160(Supplement C), 254-262.

598 Wang, C., Harbottle, D., Liu, Q. and Xu, Z. 2014. Current state of fine mineral tailings treatment: A critical
599 review on theory and practice. *Minerals Engineering* 58, 113-131.

600 Wells, P.S. and Riley, D.A. 2007, MFT drying—case study in the use of rheological modification and
601 dewatering of fine tailings through thin lift deposition in the oil sands of Alberta, pp. 13-15.

602 White, K.B. 2017. Characterizing annual changes in the chemistry and toxicity of surface water from Base
603 Mine Lake, an Alberta oil sands end pit lake.

604 White, K.B. and Liber, K. 2018. Early chemical and toxicological risk characterization of inorganic
605 constituents in surface water from the Canadian oil sands first large-scale end pit lake.
606 *Chemosphere* 211, 745-757.

607 White, K.B. and Liber, K. 2020. Chronic Toxicity of Surface Water from a Canadian Oil Sands End Pit
608 Lake to the Freshwater Invertebrates *Chironomus dilutus* and *Ceriodaphnia dubia*. *Archives of*
609 *Environmental Contamination and Toxicology* 78(3), 439-450.

610 Wong, J.W.C., Fang, M. and Jiang, R. 2001. Persistency of bacterial indicators in biosolids stabilization
611 with coal fly ash and lime. *Water environment research* 73(5), 607-611.

612 Wong, J.W.C. and Selvam, A. 2009. Reduction of indicator and pathogenic microorganisms in pig manure
613 through fly ash and lime addition during alkaline stabilization. *Journal of Hazardous Materials*
614 169(1-3), 882-889.

615 Yu, X., Cao, Y., Sampaga, R., Rybiak, S., Burns, T. and Ulrich, A.C. 2018a. Accelerated dewatering and
616 detoxification of oil sands tailings using a biological amendment. *Journal of Environmental*
617 *Engineering* 144(9), 04018091.

618 Yu, X., Lee, K., Ma, B., Asiedu, E. and Ulrich, A.C. 2018b. Indigenous microorganisms residing in oil
619 sands tailings biodegrade residual bitumen. *Chemosphere* 209, 551-559.

620 Yu, X., Lee, K. and Ulrich, A.C. 2019. Model naphthenic acids removal by microalgae and Base Mine
621 Lake cap water microbial inoculum. *Chemosphere* 234, 796-805.

622 Zubot, W. 2010. Water use and aquatic reclamation at Syncrude Canada Ltd. Integrated oil sands mining
623 operation. The Association of Professional Engineers, Geologists, and Geophysicists of Alberta
624 (APEGA).

625

626

