Activity-based cell sorting reveals resistance of functionally degenerate *Nitrospira* during a press disturbance in nitrifying activated sludge

Maxwell B.W. Madill[1,+], Yaqian Luo[1,+], Pranav Sampara[1], Ryan M. Ziels[1,#]

[1] Department of Civil Engineering, University of British Columbia, Vancouver, Canada

[+] Contributed equally to this work.

[#] Corresponding Author:
Email: ziels@mail.ubc.ca

Running Title: BONCAT-FACS of activated sludge nitrifying microbiomes

Word Count of Abstract: 249
Word Count of Importance: 150
Word Count of Text: 4958

Abstract

Managing and engineering activated sludge wastewater treatment microbiomes for low-energy nitrogen removal requires process control strategies to stop the oxidation of ammonium at nitrite. Our ability to out-select nitrite oxidizing bacteria (NOB) from activated sludge is challenged by their metabolic and physiological diversity, warranting measurements of their *in situ* physiology and activity under selective growth pressures. Here, we examined the stability of nitrite oxidation in activated sludge during a press disturbance induced by treating a portion of return activated sludge with a sidestream flow containing free ammonia (FA) at 200 mg NH_3-N/L. The nitrite accumulation ratio peaked at 42% by day 40 in the experimental bioreactor with the press disturbance, while it did not increase in the control bioreactor. A subsequent decrease in nitrite accumulation within the experimental bioreactor coincided with shifts in dominant *Nitrospira* 16S rRNA amplicon sequence variants (ASVs). We applied bioorthogonal non-canonical amino acid tagging (BONCAT) coupled with fluorescence activated cell sorting (FACS) to investigate changes in the translational activity of NOB populations throughout batch exposure to FA. BONCAT-FACS confirmed that the single *Nitrospira* ASV washed-out of the experimental bioreactor had reduced translational activity following exposure to FA, whereas the two *Nitrospira* ASVs that emerged after process acclimation were not impacted by FA. Thus, the coexistence of functionally degenerate and physiologically resistant *Nitrospira* populations provided resilience to the nitrite-oxidizing function during the press disturbance. These results highlight how BONCAT-FACS can resolve ecological niche differentiations within activated sludge and inform strategies to engineer and control microbiome function.

46 **Importance:**
47
48 Nitrogen removal from activated sludge wastewater treatment systems is an energy-intensive
49 process due to the large aeration requirement for nitrification. This energy footprint could be
50 minimized with engineering control strategies that wash-out nitrite oxidizing bacteria (NOB) to limit
51 oxygen demands. However, NOB populations can have a high degree of physiological diversity,
52 and it is currently difficult to decipher the behavior of individual taxa during applied selective
53 pressures. Here, we utilized a new substrate analog probing approach to measure the activity of
54 NOB at the cellular translational level in the face of an applied press disturbance to the activated
55 sludge process. Substrate analog probing corroborated the time-series reactor sampling, showing
56 that coexisting and functionally redundant *Nitrospira* provided resilience to the nitrite oxidation
57 process. Taken together, these results highlight how substrate analog approaches can illuminate
58 *in situ* ecophysiologies within shared niches, and can inform strategies to improve microbiome
59 engineering and management.

Introduction

Relating the in situ physiological responses of individual taxa in the face of environmental perturbations to the resulting microbial community structure and function remains a critical challenge for controlling and engineering microbiomes for desirable ecological processes and outcomes (1–3). Biological wastewater treatment processes are ideal ecosystems to explore such relationships, as environmental conditions can be manipulated, and the community function monitored, in a relatively controlled manner (4). A major goal in the wastewater industry is to engineer microbial bioprocesses to achieve energy-efficient, or even net energy-positive, wastewater treatment (5). A foundational component of achieving that goal is the optimization of mainstream biological nitrogen removal processes, as conventionally this process is the largest consumer of energy and exogenous organic carbon within wastewater treatment plants (WWTPs) (6, 7). Realizing energy-efficient nitrogen removal requires highly finessed and sustained modulation of abundances, activities and interactions of key microbial functional groups to effectively control the global community function and engage the desired nitrogen removal pathway(s) (8, 9). As such pathways typically impose inherent energetic and/or metabolic constraints (10, 11), and often challenge existing community interactions (8, 9), it is critical to fully illuminate the ecophysiological diversity and mechanisms driving niche partitioning within those microbial functional groups, as well as their responses to the applied process control strategies (9).

An appealing strategy to achieve energy-efficient biological nitrogen removal is to limit the nitrification process to nitritation (i.e. oxidation of ammonium to nitrite), as the resulting nitrite can be denitrified directly (25% and 40% net energy and carbon reductions, respectively), and/or provided to anammox bacteria as a growth substrate for autotrophic nitrogen removal (60% and 100% net energy and carbon reductions, respectively) (8, 12). However, achieving stable

86 nitritation in mainstream activated sludge (AS) stands as a major challenge limiting its successful
87 full-scale implementation (13–15). Realizing stable nitritation in mainstream AS relies on
88 engineering control strategies that serve as press disturbances to consistently repress and wash-
89 out nitrite oxidizing bacteria (NOB) while maintaining the activity of ammonium oxidizing bacteria
90 (AOB) (8, 9, 13). The efficacy of a given control strategy is therefore dependent on its ability to
91 create a disturbance that elicits different physiological responses between AOB and NOB.
92 Preliminary success in washing-out NOB from mainstream AS has been achieved using press
93 disturbances that provide a high ammonium residual (13) or control the availability of dissolved
94 oxygen (13, 16, 17) to favour the growth kinetics of AOB over NOB. Additionally, several recently-
95 proposed control strategies have utilized the higher innate sensitivity of NOB to free-ammonia
96 (FA) and free-nitrous acid (FNA) compared to AOB (18–20) to achieve effective NOB inhibition
97 (21–23). Wang et al. (21) demonstrated that a press disturbance induced by exposing a fraction
98 of return sludge to FA-rich sidestream wastewater (210 mg NH_3-N/L) supported successful NOB
99 wash-out in mainstream AS, with nitrite accumulation ratios reaching 80-90%. Despite its potential
100 efficacy for supporting mainstream nitritation, there have been a limited number of studies
101 evaluating the role of niche differentiation and physiological diversity on the stability of nitrite
102 oxidation in the face of a press disturbance from routine FA exposure.
103
104 NOB communities in wastewater treatment often display functional degeneracy, wherein the
105 nitrite oxidation process is distributed among several phylogenetically diverse taxa with varying
106 auxiliary metabolic potentials (24–29). Inherent differences in nitrite oxidation biochemistry and
107 cell morphology play core roles in supporting ecophysiological diversity between NOB genera by
108 influencing their substrate affinities for oxygen and nitrite, and their nitrite oxidation kinetics (24,
109 30, 31). *Nitrospira*, a predominant NOB genus in many WWTP microbiomes (32–34), has
110 demonstrated an extraordinary degree of functional degeneracy, with reports of highly complex
111 and stable communities containing as many as 120 closely-related coexisting strains (26, 27, 34).

112 Considerable ecophysiological diversity may thus exist between *Nitrospira* species/strains to
113 support niche partitioning, which could be supported by distinct oxygen and nitrite preferences
114 (34–36), auxiliary metabolic potentials for utilizing alternative electron acceptors and/or donors
115 (24, 27, 34, 37, 38), and tolerances to challenging environmental conditions including FA (27, 28,
116 39, 40). Exhibited at both the genus and strain-level, such functional degeneracy may enable
117 NOB communities to resist the selective pressures imparted by engineering process control
118 strategies by recruiting functionally redundant, yet physiologically diverse, NOB members (41–
119 43). *In situ* assessments of the metabolically active fraction of nitrifying communities are therefore
120 critical to evaluate the efficacy of mainstream nitritation control strategies and elucidate their
121 associated impacts on functionally degenerate NOB.
122
123 Next-generation substrate analogue probing approaches have recently emerged as powerful
124 tools to decipher the *in situ* physiology of active cells based on their uptake of synthetic analogues
125 of natural biomolecules (44). Bioorthogonal non-canonical amino acid tagging (BONCAT) is a
126 nascent SAP approach to study the physiology of active cells in complex environmental
127 microbiomes (45–48). BONCAT relies on the *in vivo* uptake and incorporation of synthetic amino
128 acids, such as the alkyne-containing analogue of methionine, homopropargylglycine (HPG), into
129 newly synthesized proteins via the native translational machinery, and thereby selectively labels
130 the proteomes of translationally active cells (44, 47). HPG-labeled cells can subsequently be
131 identified by tagging their proteins with azide-modified fluorescent dyes via azide-alkyne click-
132 chemistry, enabling their selective recovery using fluorescence activated cell sorting (FACS) (45,
133 46, 48). To our knowledge, BONCAT, or its paired approach with FACS (BONCAT-FACS), have
134 yet to be applied to study the active fractions of AS microbiomes central to wastewater treatment
135 bioprocesses.
136

137 The objective of this study was to assess the stability of nitrite oxidation in the face of a press
138 disturbance induced by routine exposure of recycled activated sludge with FA as an engineering
139 control strategy to wash-out NOB. We hypothesized that certain members of the active NOB
140 microbial community could acclimate to the applied press disturbance. Two parallel experimental
141 and control AS sequencing batch reactors (SBRs) were operated for ~100 days to investigate the
142 impacts of routine FA exposure as a press disturbance on the NOB community. We applied time-
143 series 16S rRNA gene amplicon sequencing in addition to BONCAT-FACS based activity
144 measurements to elucidate changes in the structure and *in situ* activity of the AS microbiome and
145 nitrifying communities.
146
147 **Results**
148
149 *Partial nitritation performance of activated sludge bioreactors*
150
151 Two AS SBRs treated synthetic mainstream municipal wastewater over two operational phases:
152 the Start-up Phase and the Treatment Phase (Figure S1). Periodic steady-state conditions were
153 presumed over the last 30 days of the 270-day Start-up Phase, during which there were no
154 significant differences in daily effluent concentrations of NH_4^+-N, NO_3^--N, and NO_2^--N between the
155 two SBRs ($p > 0.05$), which averaged 0.1$\pm$0.3 mg NH_4^+-N/L, 19$\pm$2 mg NO_3^--N/L and 0.0$\pm$0.01
156 mg NO_2^--N/L, respectively (Figure 1). There were also no significant differences in the TSS and
157 VSS of mixed liquor between the two systems during the Start-up Phase ($p > 0.05$; Figure S2).
158 Thus, similar stable performance with full nitrification was achieved in both SBRs during the initial
159 Start-up Phase, indicating effective duplication of operating conditions.
160
161 The Treatment Phase was then commenced on day 0 to assess the impacts of a press
162 disturbance induced by routine FA-exposure of return sludge in a side-stream reactor on the
163 nitrifying community structure and activity. Approximately 20% of the return sludge in the

164 treatment SBR was exposed to 200 mg NH_3-N/L as FA at a pH of 9.0 for 24 hrs before being
165 reintroduced into the mainstream SBR on a daily basis, while the same conditions were emulated
166 for the control SBR but without FA added to the side-stream. Additionally, ammonium nitrogen
167 was added (160 mg NH_4^+-N/d) to the control SBR on a daily basis along with the return sludge to
168 achieve equivalent nitrogen loads between the two SBRs. After 10 days of the applied press
169 disturbance, NO_2^--N began to increase in effluent of the treatment SBR, but stayed below
170 detection in the control SBR for the remainder of the Treatment Phase (Figure 1). By day 40,
171 effluent NO_2^--N reached its peak level of 11 mg NO_2^--N/L in the treatment SBR. At the same time,
172 NH_4^+-N accumulated to 3.7 mg NH_4^+-N/L in the treatment SBR between days 37 and 44, yet
173 stayed below 1.2 mg NH_4^+-N/L in the control SBR over the entire Treatment Phase (Figure 1).
174 The accumulation of NO_2^--N in the treatment SBR was transient, however, as the effluent
175 concentration decreased after day 40 and reached a value below detection by day 74 (Figure 1).
176
177 As effluent was sampled from the SBRs on a 24 h basis, and the sidestream return sludge was
178 added once every 24 h, periodic tests were conducted to measure nitrogen species at the end of
179 individual SBR cycles over the course of 24 h to better estimate nitrite accumulation ratios (NARs)
180 (Figure S3). After 20 days of the press-disturbance, the 24 h average NAR was approximately 10
181 times higher in the treatment SBR versus the control (Figure 1C). The NAR reached its peak of
182 41.9% by day 37 in the treatment SBR, aligning with the observed peak in effluent NO_2^--N
183 concentrations. After day 37, the NAR decreased in the treatment SBR, reaching its lowest
184 observed level of 10.9% on day 75. The NAR of the control SBR stayed below 3.2% in the entire
185 Treatment Phase (Figure 1C). This suggests that the nitrite oxidation function was resilient to the
186 press disturbance of routine FA exposure, as the extent of nitrite oxidation inhibition was not
187 sustained after about 40 days.

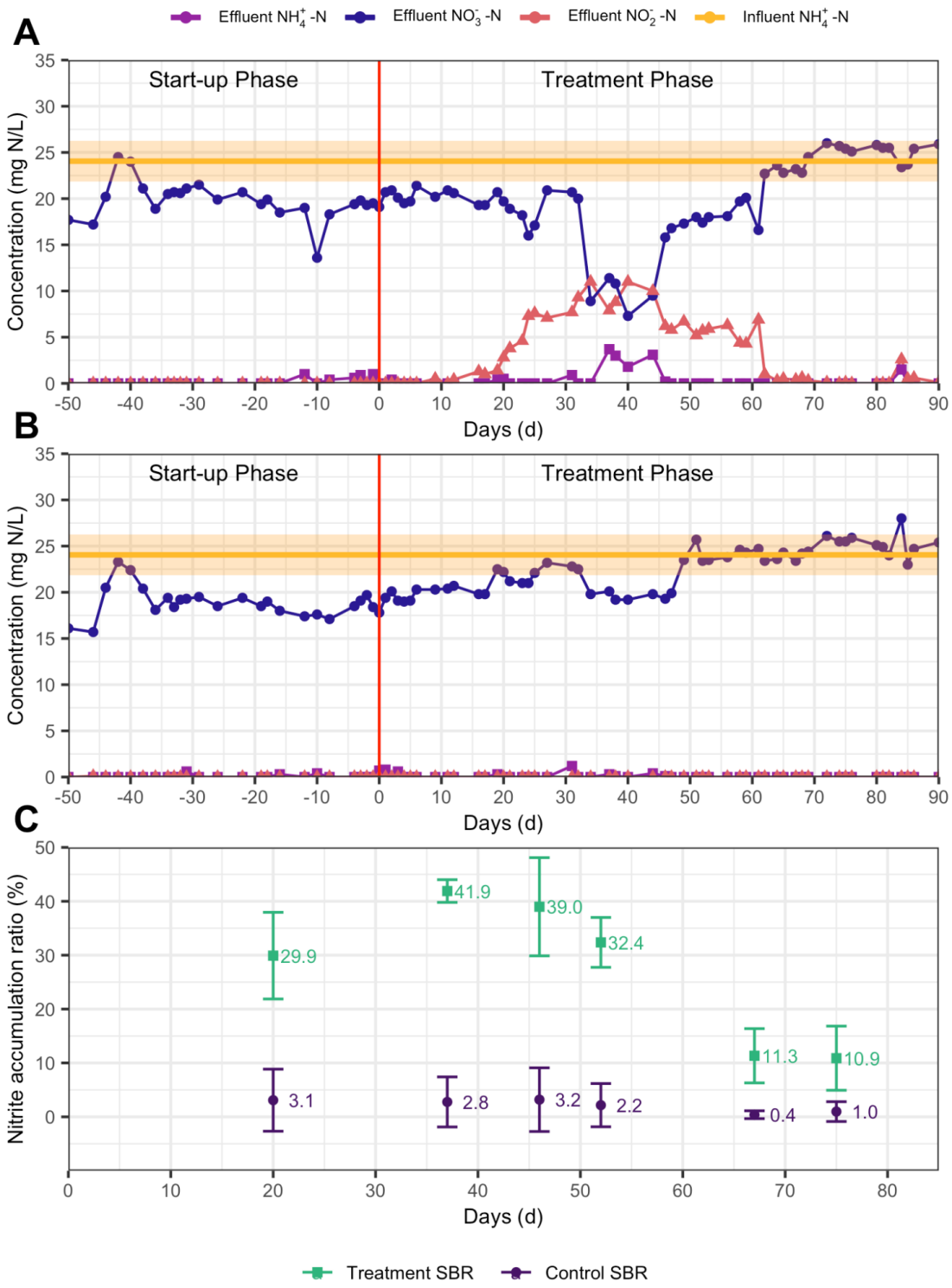


Figure 1: Influent and effluent nitrogen species over the two experimental phases in the (A) treatment sequencing batch reactor (SBR), and (B) control SBR. Shaded orange space represents standard deviation of influent NH_4^+-N. (C) Nitrite accumulation ratio (NAR, %) of treatment and control SBRs over time, based on 24 h monitoring of SBR effluents. NAR is defined

193 as the effluent nitrite (mg N/L) divided by the effluent nitrite plus nitrate (mg N/L), and indicates
194 the level of NOB activity suppression.
195

196 *3.2 Microbial community acclimation to routine FA-exposure*
197
198 Microbial 16S rRNA gene amplicons were denoised into amplicon sequence variants (ASVs) to
199 provide a high-resolution (49, 50) view of how routine FA-exposure impacted the community
200 structure. A total of 3.01 million chimera-free quality-filtered merged reads (Table S1) were
201 denoised into 6,694 ASVs. Over 95% of the 16S rRNA gene amplicons at all time points in both
202 SBRs were comprised of the 8 phyla: *Proteobacteria*, *Bacteroidetes*, *Chloroflexi*, *Nitrospirae*,
203 *Planctomycetes*, *Verrucomicrobia*, *Acidobacteria*, and *Cyanobacteria* (Figure S4). Between 53%
204 to 70% percent of 16S rRNA amplicons were represented by 20 genera across all samples (Figure
205 2). Even at the broad genus level of resolution, there were apparent differences in community
206 profiles between the treatment and control SBRs over time (Figure 2), indicating that routine FA
207 exposure altered the structure of the AS microbiome.

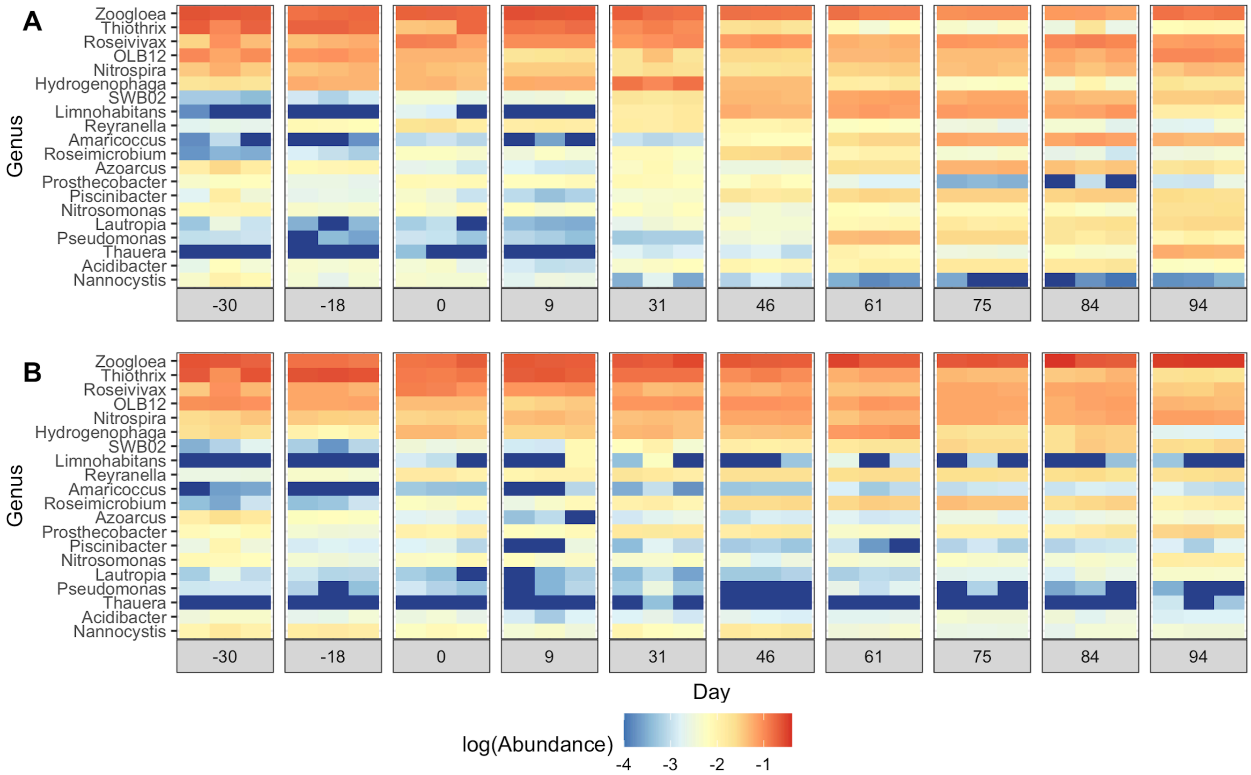


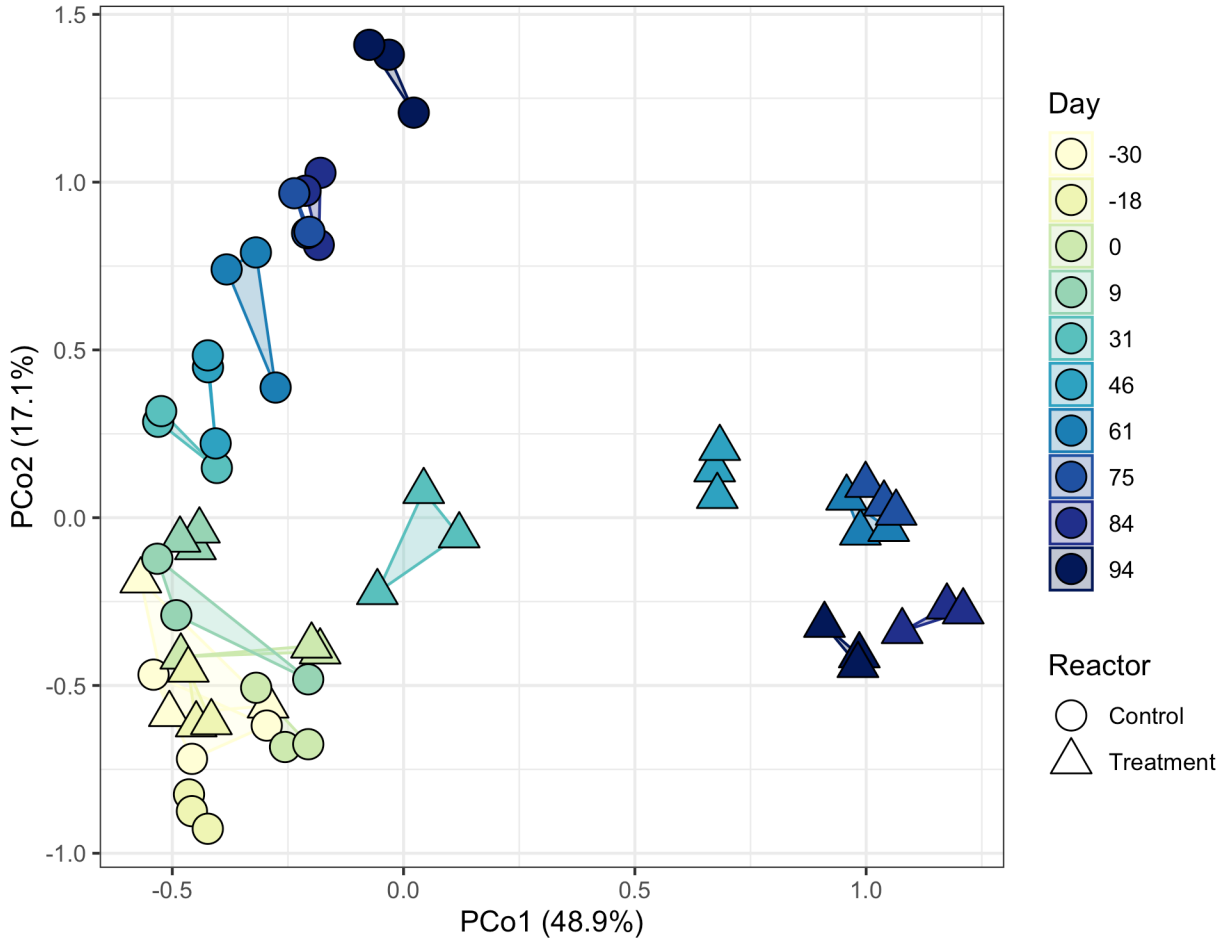
Figure 2: Heatmap of log-scaled relative abundance of top 20 most abundant genera in the (A) treatment SBR and (B) control SBR over the two operational phases. Sequencing results are shown for triplicate DNA extractions on each sampling date. The genera are ordered from highest to lowest cumulative abundance across samples.

At the ASV level, FA-exposure of return sludge led to significant differences in community structure between the two SBRs over time (R^2 = 0.54, p < 0.001, adonis). Principal coordinates analysis (PCoA) of cumulative sum-scaled ASV read counts revealed that community profiles in the two SBRs were similar until day 9, after which the treatment SBR community diverged from the control (Figure 3). Differential abundance analysis showed that there were no statistically different ASVs between the two SBRs on day 0 (p > 0.01, DESeq2), indicating that they were well replicated in the Start-up Phase. By day 46 of the Treatment Phase, around the time that nitrite peaked in the treatment SBR (Figure 1), 105 ASVs spanning 55 genera were differentially abundant between the two SBRs (p < 0.01, DESeq2; Figure S5). The number of ASVs with

224 significant differential abundance between the SBRs continued to increase to a maximum of 166

225 on day 75 of the Treatment Phase (Figure S5).

226



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228 **Figure 3:** Principal coordinates analysis (PCoA) of Bray-Curtis dissimilarities between cumulative
229 sum scaled (CSS) read counts of 16S rRNA ASVs in both SBRs over time. The marker fill
230 represents reactor operating day, and the marker shape corresponds to the SBR. Triplicate DNA
231 extractions for each time point are indicated by a shared polygon. The percentages in
232 parentheses represent the fraction of variance explained by that axis.

233

234

235 *Nitrospira* and *Nitrosomonas* were the only putative NOB and AOB populations detected in the

236 SBRs, respectively (Figure 2). As these reactors were fed with synthetic wastewater, it is likely

237 that these populations originated from the inoculum. Six dominant *Nitrosomonas* ASVs were

238 detected in both SBRs over the two experimental phases (Figure S6). Until day 84, the total
239 *Nitrosomonas* abundance was less than 1% in both SBRs, but increased to over 1.5% in both by
240 day 94 (Figure S12). One *Nitrosomonas* ASV (ASV_36) was differentially abundant between the
241 two SBRs on days 75 and 94 ($p < 0.01$, DESeq2). Three *Nitrospira* ASVs were detected within
242 the two SBRs (Figure 4). In particular, ASV_8 was the dominant *Nitrospira* ASV in both SBRs
243 during the Start-up Phase (before day 0), accounting for 3.3%$\pm$0.8% of 16S rRNA genes on
244 average (Figure 4). By day 46, ASV_8 decreased to 1.6%$\pm$0.1% in the treatment SBR, while it
245 increased to 6.4%$\pm$0.3% within the control. This decrease in ASV_8 abundance coincided with
246 the peak in nitrite accumulation in the treatment SBR (Figure 1a). The abundance of ASV_8 did
247 not significantly change after day 46 for the remainder of the experiment in the treatment SBR (p
248 > 0.01, DESeq2). In contrast, two other *Nitrospira* ASVs (ASV_32 and ASV_47; 99.7% sequence
249 similarity to each other; 94.2% and 94.5% sequence similarities to ASV_8, respectively) were
250 sporadically detected in both SBRs at abundances below 0.4% until day 46, and then increased
251 to a maximum of 1.8%$\pm$0.4% and 1.2%$\pm$0.3% in the treatment SBR by day 84, respectively, but
252 stayed below 0.3% in the control (Figure 4). The abundance of ASV_32 and ASV_47 were
253 significantly higher in the treatment SBR relative to the control SBR by the end of the experiment,
254 while ASV_8 was significantly lower (both days 84 and 94; $p < 0.01$, DESeq2). Phylogenetic
255 analysis based on partial 16S rRNA gene sequences (Figure S7) revealed that ASV_8 was most
256 closely related to *Nitrospira lenta* within lineage II, whereas ASV_32 and ASV_47 were clustered
257 within lineage I of *Nitrospira* (51, 52).
258

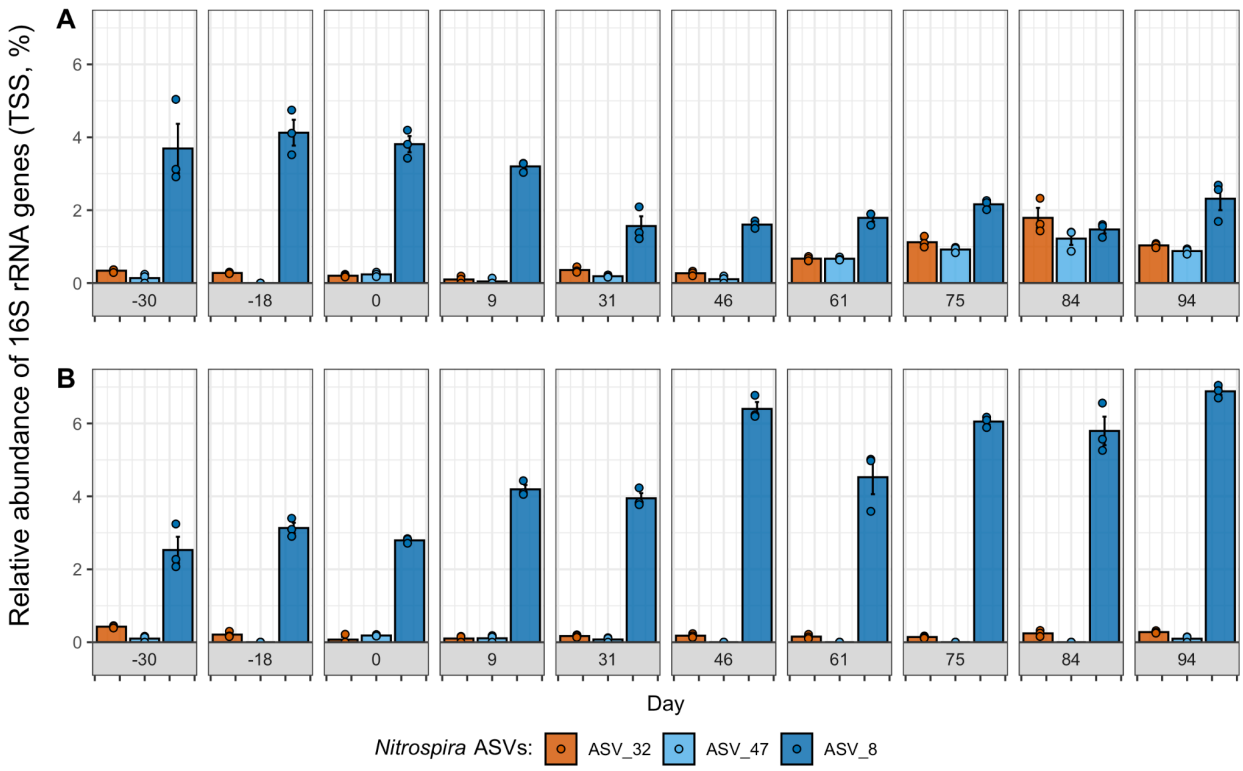

Figure 4: Relative abundance (total sum scaled, TSS) of three dominant *Nitrospira* ASVs over time in the (A) treatment SBR and (B) control SBR. Only *Nitrospira* ASVs detected in more than one sample are shown. Results are shown for triplicate DNA extractions on each day, with the coloured points showing the relative abundance of each ASV in each DNA extraction, the coloured bar showing the mean relative abundance, and the error bars showing the standard error of the mean.

3.3 Substrate analog probing of active nitrifying populations

To decipher the impact of the applied press disturbance on the activity and *in situ* physiology of nitrifying populations within mainstream AS, we conducted BONCAT-FACS on samples collected from nitrifying microcosms seeded with SBR mixed liquor preceding return sludge treatment (R), as well as return sludge from the beginning (15 min after start; S1) and end (24 h after start; S2) of the side-stream return sludge treatment processes for each SBR (Figure 5). Preliminary validation microcosms confirmed the sensitivity of BONCAT in labeling only active cells that had incorporated HPG (Figure S8). Comparison of FACS data between the two SBRs (Figures S9 and S10) revealed that translationally active (i.e. BONCAT+) cell fractions were significantly lower

279 in the S1 and S2 nitrifying microcosms seeded from the FA-exposed treatment SBR relative to
280 those of the control SBR ($p < 0.05$, t-test), but that no significant difference in BONCAT+ cell
281 fractions was observed in the R microcosms seeded with mixed-liquor preceding sludge
282 treatment. (Figure 5). Within the treatment SBR nitrifying microcosms, the fraction of BONCAT+
283 cells in the S1 microcosm was 25% lower than that of the R microcosm without FA-exposure, but
284 the this was not significant ($p = 0.078$; Figure 5). No further reduction in the fraction of BONCAT+
285 cells was observed between microcosms seeded with return sludge from the beginning (S1) and
286 end (S2) of the FA-exposure process (Figure 5). In contrast, the BONCAT+ cell fraction in the
287 microcosm seeded with control SBR return sludge after 24 hrs of its sludge treatment (S2) was
288 significantly higher by 63% compared to that at the beginning of its sludge treatment process (S1;
289 p=0.045; Figure 5).

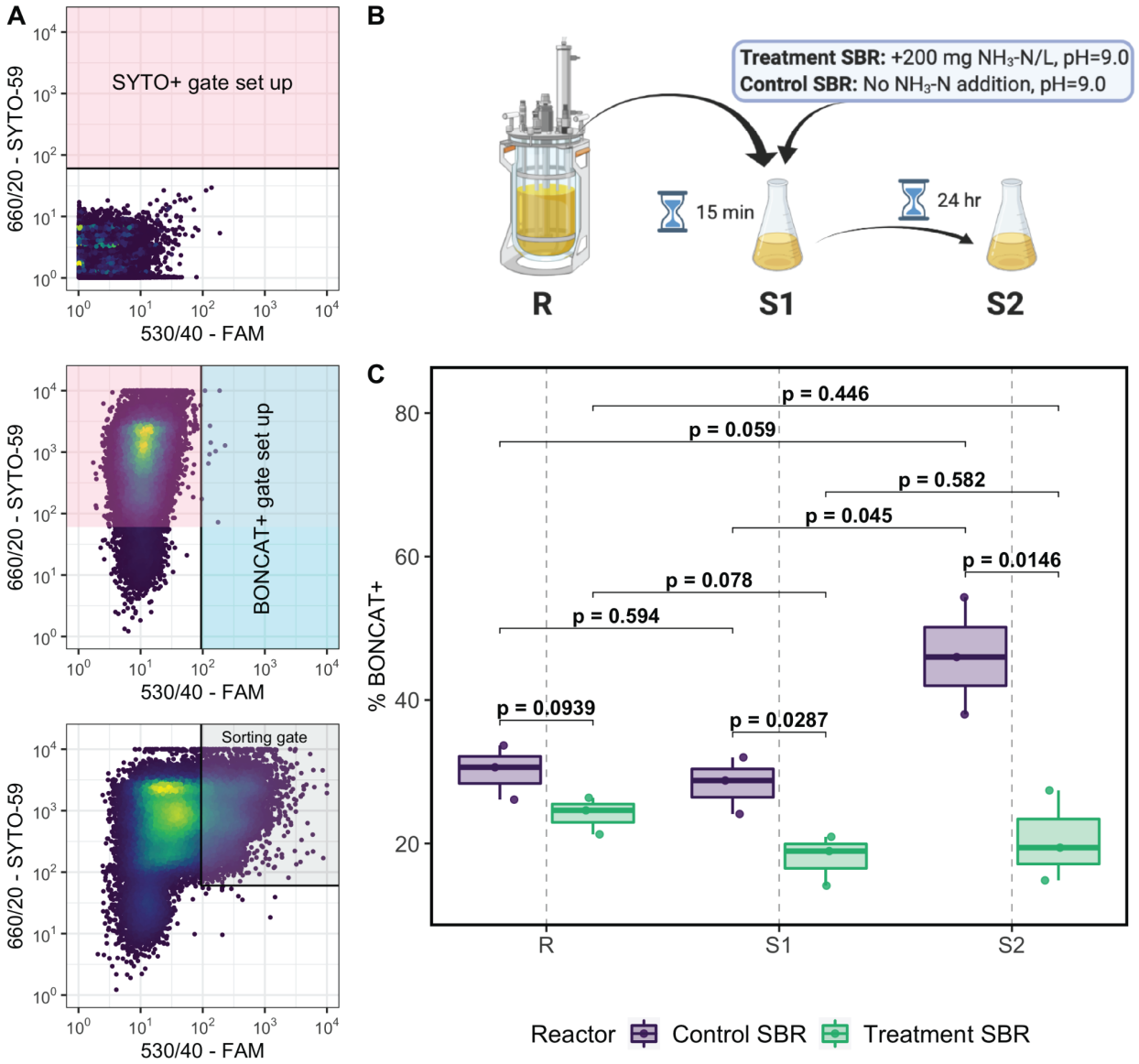

Figure 5: Application of BONCAT-FACS to aerobic nitrifying microcosm samples. (A) Sorting gates were established based on the detection of SYTO+ cells from background particles using unstained HPG-negative cells (top panel), and BONCAT+ cells from non-HPG labelled cells based on their FAM picolyl azide dye fluorescence using SYTO59 stained HPG-negative cells (middle panel), allowing less than 0.5% false positives in each gate. BONCAT+ cell fractions in each HPG-incubated sample were calculated as the fraction of SYTO+ cells residing in the sorting gate (bottom panel). (B) Overview of the return sludge treatment processes showing the origin of samples used to seed the nitrifying microcosms. (C) Comparison of BONCAT+ cell fractions detected in samples prepared from the nitrifying microcosms seeded with mixed-liquor preceding return sludge treatment (R), return sludge after 15 min of side-stream treatment (S1), and return sludge after 24 h of side-stream treatment (S2) from the treatment SBR (bottom) and control SBR (top). Black brackets are shown for comparisons of BONCAT+ cell fractions made between samples, with *p* values calculated using independent t-tests.

304 The microbial community composition observed in the BONCAT+ cell fractions could be attributed
305 to changes in cellular translational activity, as well as any changes that occurred in the bulk
306 community composition throughout the incubation and/or sample processing steps upstream of
307 FACS. To delineate these impacts, we conducted 16S rRNA gene amplicon sequencing on
308 triplicate bulk (i.e. pre-homogenized), post-homogenized and click-chemistry labelled samples
309 from each microcosm, in addition to pre-homogenized bulk samples from HPG-negative control
310 microcosms and SBR mixed liquors prepared with different DNA extraction and PCR amplification
311 procedures. The low-biomass DNA extraction method (prepGEM kit) and the two-step 16S rRNA
312 gene PCR amplification, which were both used to prepare the BONCAT+ libraries, both showed
313 impacts on the community composition relative to respective samples prepared for time-series
314 analysis (i.e. FastDNA Soil kit with one-step PCR; Figure S11). For this reason, the community
315 compositions measured in the BONCAT+ samples were not compared to time-series samples
316 with different DNA extraction and amplification procedures. PCoA revealed that bulk samples
317 from microcosms incubated without HPG were similar to those with HPG, indicating that HPG did
318 not alter the community structure during the 3 h incubation (Figure 6). Bulk samples from the
319 control SBR microcosms (R, S1 and S2) were generally clustered with the mixed liquor sampled
320 directly from the SBR (Figure 6B). In contrast, bulk samples from the treatment SBR microcosms
321 diverged slightly from the bulk SBR community after 24 hrs of FA-exposure (e.g. S2 microcosms;
322 Figure 6A), suggesting that the community composition was altered by exposure to FA. The
323 community compositions of the post-homogenized samples were not identical to the bulk
324 microcosm samples, which could be attributed to cell lysis or the removal of extracellular DNA
325 during the homogenization and click-labeling procedures (45) (Figure 6). Due to the
326 aforementioned findings, the BONCAT+ cell fractions were only compared to their corresponding
327 post-homogenized microcosm samples processed with the same low-biomass DNA extraction
328 method.
329

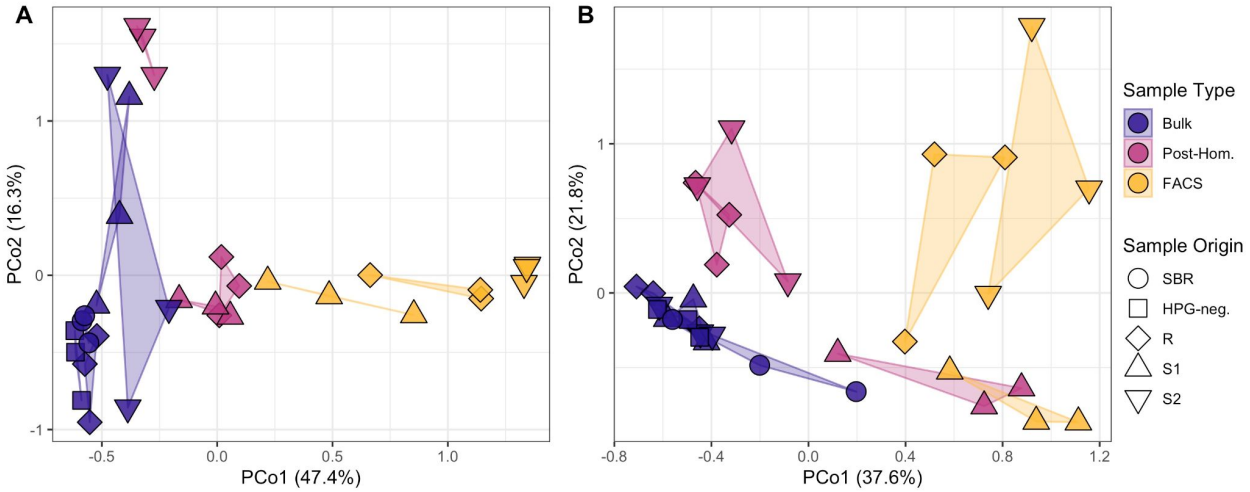

Figure 6. Principal coordinates analysis (PCoA) of Bray-Curtis dissimilarities between cumulative sum scaled (CSS) read counts of 16S rRNA ASVs measured in Bulk, Post-Homogenized (i.e. Post-Hom.) and BONCAT+ (i.e. FACS) samples prepared from nitrifying microcosms seeded with mixed liquor preceding side stream treatment (R), mixed liquor after 15 min of side stream treatment (S1), and mixed liquor after 24 h of side stream treatment (S2), from (A) the treatment SBR and (B) the control SBR. Bulk samples from the HPG-negative R control microcosms (i.e. HPG-neg.) and bulk mixed liquor samples directly from the SBRs (i.e. SBR) were also included. The marker fill represents the preparation procedure for each sample and the marker shape represents the sample origin (microcosm type or SBR mixed liquor). Triplicate samples are indicated by a shared polygon. The percentages in parentheses represent the fraction of variance explained by that coordinate axis.

Comparing the abundance of taxa in a BONCAT+ cell fraction to that in its corresponding bulk community prior to FACS can identify changes in translational activity at the population level (45, 48). For both the treatment and control SBRs, PCoA showed that the largest distance between post-homogenized community compositions and BONCAT+ cell fractions was observed in microcosms seeded with S2 biomass after 24 hrs of FA-exposure (Figure 6). Differential abundance analysis identified 8 and 0 differentially abundant ASVs ($p < 0.01$, DESeq2) in the BONCAT+ fractions of the treatment and control SBR mixed-liquor seeded (R) microcosms, respectively, and 0 differentially abundant ASVs in the BONCAT+ fractions of both SBR microcosms seeded with S1 biomass from the start of return sludge treatment. Conversely, for microcosms seeded with S2 biomass after 24 h of return sludge treatment, 56 and 26 ASVs were differentially abundant in the BONCAT+ fractions of the treatment SBR and control SBR samples,

354 respectively. Within the S2 microcosm BONCAT+ fraction from the treatment SBR, the 56
355 differentially abundant ASVs spanned 26 genera, where 34% of the ASVs were significantly
356 enriched and 66% were significantly reduced, relative to the post-homogenized community
357 (Figure 7). In contrast, the 26 differentially abundant ASVs identified in the S2 microcosm
358 BONCAT+ fraction from the control SBR, which spanned 13 genera, were all (100%) enriched
359 relative to the post-homogenized community (Figure 7). These results suggest that side-stream
360 return sludge treatment caused distinct shifts in the translationally active fraction of the
361 communities in both SBRs, with FA-exposure negatively impacting the activity of a greater
362 number of taxa compared to a similar incubation without FA.

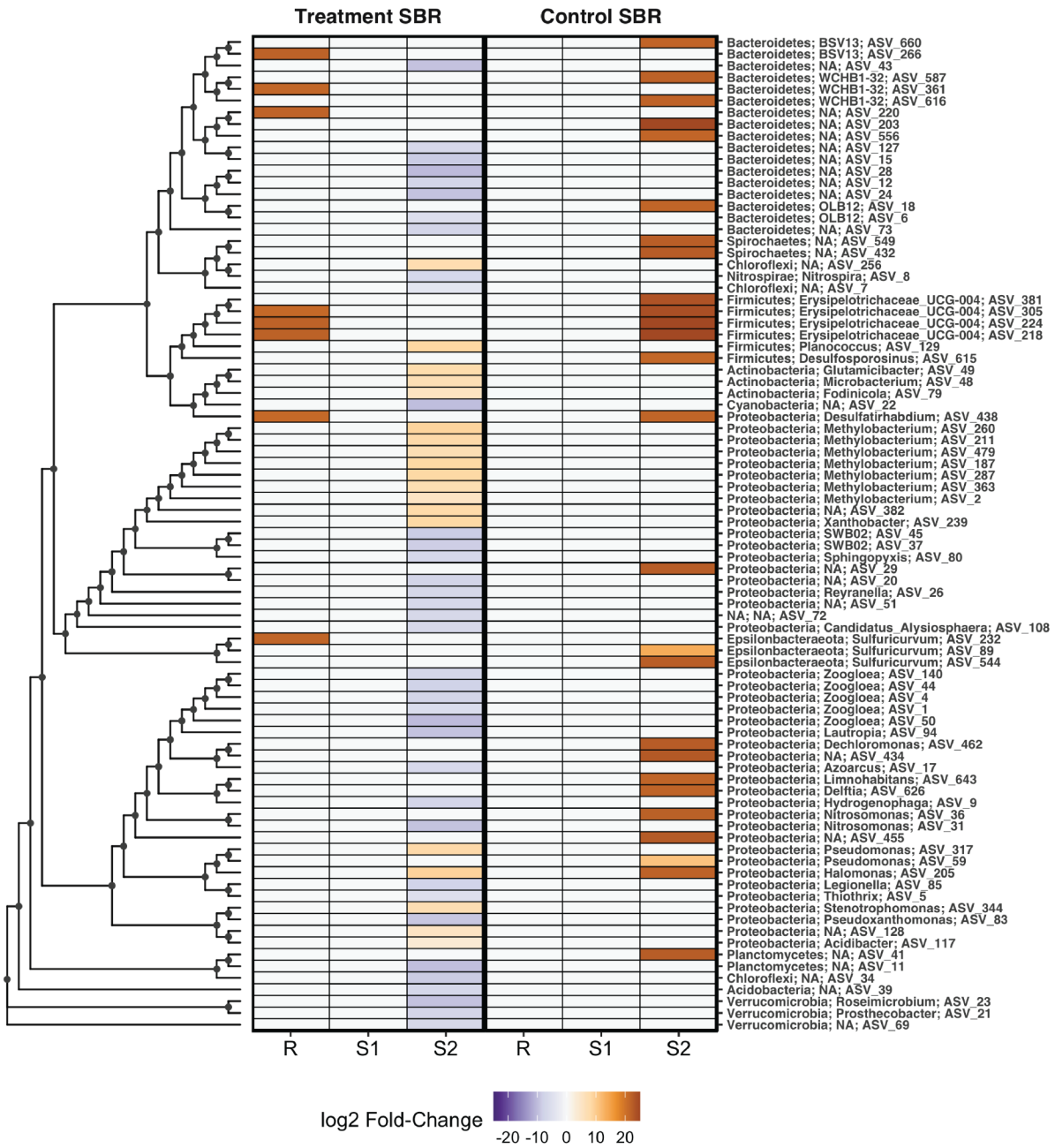

Figure 7: Log_2 fold-change in abundance of all differentially abundant ASVs detected in comparisons between BONCAT+ (i.e. FACS) and Post-Homogenized (i.e. Post-Hom.) fractions prepared from nitrifying microcosms seeded with mixed-liquor preceding return sludge-treatment (R), return sludge after 15 min of side-stream treatment (S1), and return sludge after 24 h of side-stream treatment (S2) from the treatment SBR (left panel) and control SBR (right panel). For each microcosm, log_2 fold-changes are shown only for differentially abundant ASVs, as determined by comparing ASV abundance in triplicate DNA extracts of each fraction using DESeq2 with an adjusted significance level of $p < 0.01$. Log_2 fold-changes that were not significant are set to '0' for visualization purposes. ASVs are ordered based on their phylogenetic distances estimated through multiple sequence alignment using the DECIPHER package (v.2.14.0) (53), and the phylogenetic tree was constructed using a maximum likelihood approach in the phangorn package

374 (v.2.5.5) (54) in the R environment. Tree tip labels (right side of the heatmap) denote the phylum and genus
375 level classifications of each ASV, where NA denotes an unknown taxonomic identity at that level.
376

377 As the BONCAT microcosms were amended with NH_4^+-N as the sole electron donor, it was
378 possible to assess the impact of return sludge treatment on the translational activity of nitrifying
379 populations. The three dominant *Nitrospira* ASVs and the top two dominant *Nitrosomonas* ASVs
380 detected in both SBR BONCAT microcosm sample sets corresponded to the same dominant
381 ASVs detected within both SBR mixed liquors on day 94 of the time-series sampling, when the
382 microcosms were established (Figure 8, Figure S12). The two dominant *Nitrosomonas* ASVs
383 (ASV_31 and ASV_36) were both differentially abundant in the BONCAT+ fractions of S2
384 microcosms ($p < 0.01$, DESeq2; Figure S12), while the observed differences varied by SBR.
385 *Nitrosomonas* ASV_31 was significantly reduced in the BONCAT+ fraction of the S2 microcosm
386 from the treatment SBR, while *Nitrosomonas* ASV_36 was significantly enriched in the BONCAT+
387 fraction of the S2 microcosm of the control SBR (Figure S13). Similar to *Nitrosomonas*, the only
388 significant differential abundance in BONCAT+ fractions for *Nitrospira* ASVs occurred in a S2
389 microcosm (Figure 8). *Nitrospira* ASV_8 was the only differentially abundant NOB in BONCAT+
390 fractions of both SBR microcosms, and was significantly reduced in the S2 microcosm of the
391 treatment SBR ($p < 0.01$, DESeq2). These results indicate that significant reductions in
392 translational activity of nitrifying populations were only observed in return sludge exposed to FA.

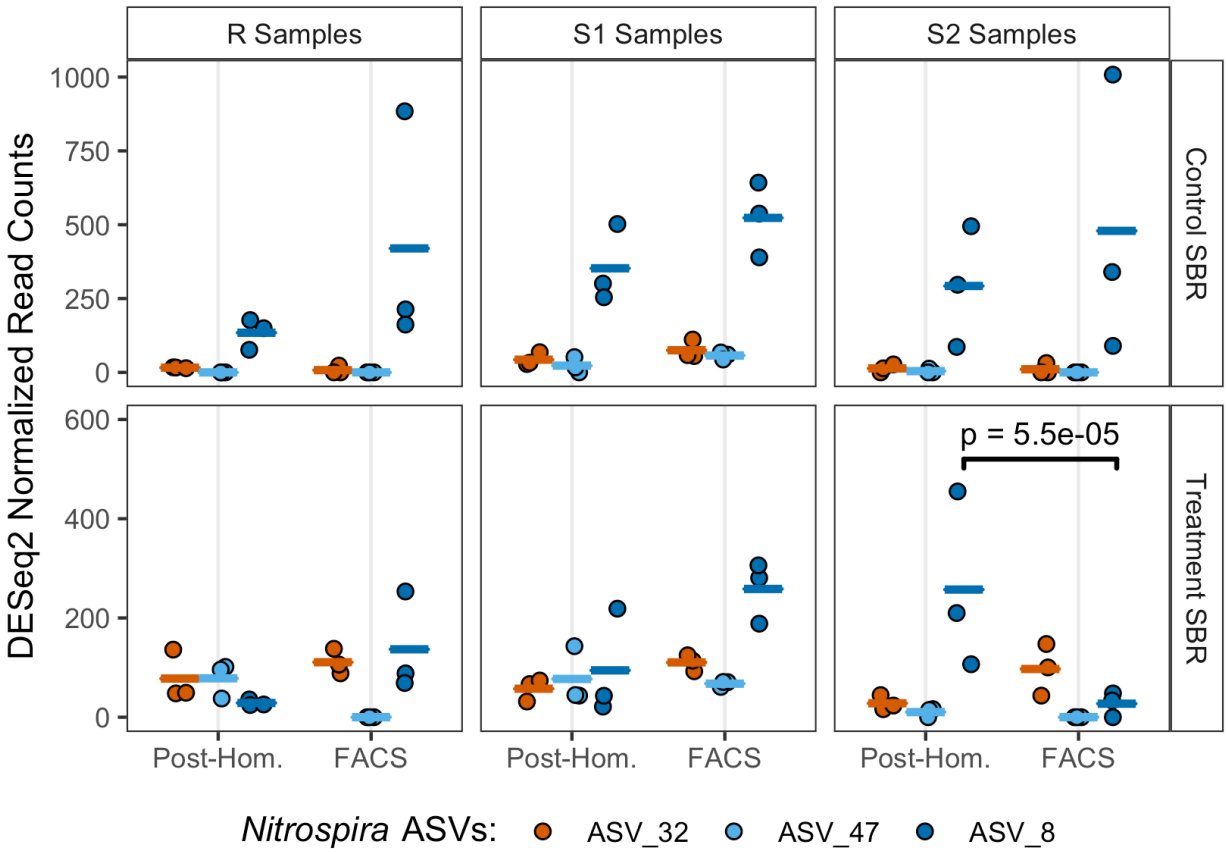

Figure 8: Normalized read counts of three dominant *Nitrospira* ASVs in triplicate BONCAT+ (i.e. FACS) and corresponding Post-Homogenized (i.e. Post-Hom.) and BONCAT+ (i.e. FACS) libraries generated from samples prepared from nitrifying microcosms seeded with mixed-liquor preceding return sludge treatment (R), return sludge after 15 min of side-stream treatment (S1), and return sludge after 24 h of side-stream treatment (S2) from the treatment SBR (bottom) and control SBR (top). Points indicate triplicate normalized read counts per ASV, and the horizontal bars of the same colors represent the sample mean per ASV. The reads were normalized with DESeq2 based on total read-counts per sample. A black bracket between an ASV within two samples represents a significant difference in the mean read abundance, determined with DESeq2 at an adjusted significance level of $p < 0.01$.

Discussion

Implementing process control strategies that achieve consistent modulation of key functional groups remains a critical challenge towards the development of sustainable wastewater treatment biotechnologies. This is particularly true for repressing functionally diverse NOB to promote nitritation for energy-efficient biological nitrogen removal in AS processes. This study

412 demonstrates how underlying shifts in the abundance of physiologically diverse *Nitrospira*
413 populations can confer resilience to the nitrite-oxidizing community during an imposed press
414 disturbance in AS treatment, in this case induced by routine exposure of return sludge to FA. Our
415 results also demonstrate the utility of substrate analog probing approaches like BONCAT to
416 illuminate the *in situ* ecophysiology of shared niches within the activated sludge microbiome, and
417 the associated impacts on process and ecosystem functional stability.
418
419 We observed that routinely exposing ~20% of the return sludge to FA (200 mg NH_3-N/L) in a
420 sidestream reactor with a 24 hr retention time initially reduced nitrite-oxidizing activity in the
421 mainstream reactor, achieving a maximum observed NAR of 42%. However, we observed
422 acclimation of the nitrite-oxidation function after approximately 40 days of the press disturbance,
423 indicated by decreasing NARs within the treatment SBR. This finding is in contrast to those
424 reported by Wang et al. (21) who observed stable repression of NOB activity for over 100 days
425 using a press disturbance of routine FA-exposure at similar conditions (210 mg N/L of FA for 24
426 hr, 22% return sludge exposed). It is important to note that both the pH of the side-stream sludge
427 treatment incubation, and the daily input of total ammonia nitrogen into the mainstream SBR with
428 the return sludge, were controlled across both SBRs. Furthermore, while salinity has been shown
429 to be a driver of Nitrospira population structure (55), the salinity in the side-stream reactor during
430 FA-exposure (~2.5 g/L as Na^+ + Cl^-) was 10-times less than the 50% inhibition level observed for
431 a *Nitrospira* dominated AS community of 30 g/L NaCl (56). Therefore, the inhibition and
432 acclimation of NOB activity to the applied press disturbance can likely be attributed to varying
433 physiological responses of NOB community members to FA-exposure.
434
435 While acclimation of NOB communities in mainstream AS to routine FA-exposure has been
436 previously reported (41, 42, 57), very few studies have directly investigated the role of
437 physiological diversity between NOB members in supporting community-level acclimation. To our

438 knowledge, all previous reports of NOB community acclimation to return sludge FA-exposure
439 have involved shifts in major NOB genera during this press disturbance. Specifically, Li et al. (42)
440 found that *Candidatus* Nitrotoga replaced *Nitrospira* as the dominant NOB in response to return-
441 sludge FA-exposure, while Duan et al. (41) reported a shift from *Nitrospira* to *Nitrobacter* in
442 response to FA-exposure. These reported shifts could be supported by potentially distinct
443 physiological characteristics of *Candidatus* Nitrotoga and *Nitrobacter* compared to *Nitrospira*,
444 such as higher tolerances to FA-inhibition (42, 58) and/or preferences for higher nitrite
445 concentrations (30, 59). In contrast to these genus-level community shifts however, we observed
446 that NOB community acclimation to FA-exposure could occur via shifts between *Nitrospira*
447 sequence variants, specifically from a dominant variant belonging to *Nitrospira* lineage I (ASV_8)
448 to two variants that belonged to *Nitrospira* lineage II (ASV_32 and ASV_47). These findings
449 therefore reveal that physiological diversity at the lineage level, and possibly at the sub-lineage
450 or strain level, within *Nitrospira*-dominated communities can facilitate niche-partitioning and
451 acclimation to FA-exposure as a press-disturbance when applied as an engineering control
452 strategy to promote low-energy nitrogen removal.
453
454 *Nitrospira* species are well-known to be metabolically versatile, with a wide range of functional
455 potentials including nitrite oxidation, hydrogen oxidation, urea conversion, formate oxidation,
456 nitrate reduction, and complete ammonium oxidation (24, 37, 38, 52, 60). This metabolic diversity
457 likely creates opportunities for functionally degenerate *Nitrospira* populations to co-exist within a
458 bioreactor by differentiating in niche space through auxiliary metabolic specializations, while also
459 sharing a common niche space of nitrite oxidation (34, 61). Here, we employed BONCAT-FACS
460 for the first time in an AS wastewater treatment system, to the best of our knowledge, which
461 highlighted differing *in situ* physiologies of three *Nitrospira* variants and resolved their responses
462 to FA-exposure on the level of cellular translational activity. BONCAT-FACS revealed that
463 transitional activity in *Nitrospira* ASV_8 was significantly reduced following exposure to FA for 24

464 h, aligning with the results of the time-series reactor sampling in which *Nitrospira* ASV_8 was
465 washed out of the treatment SBR but remained dominant in the control SBR. In contrast,
466 *Nitrospira* ASV_32 and ASV_47 increased in abundance within the treatment SBR coinciding with
467 the decrease in NAR after day 40, and BONCAT-FACS revealed the translational activity of those
468 variants remained unchanged following FA-exposure. Based on the close alignment of the
469 BONCAT-FACS observations with the trends in the time-series reactor sampling, it can therefore
470 be hypothesized that the wash-out of *Nitrospira* ASV_8 occurred due to its decreased activity in
471 response to the press-disturbance of routine sidestream FA-exposure, which induced a growth
472 lag once it was reintroduced into the mainstream SBR. In contrast, the potentially distinct auxiliary
473 metabolic potentials of *Nitrospira* ASV_32 and ASV_47 may have conferred their physiological
474 resistance to the side-stream FA-exposure, thereby providing these variants with a competitive
475 growth advantage within the mainstream SBR. Physiological tolerance to FA-exposure was
476 identified as a mechanism of niche-partitioning between *Nitrospira* populations of lineages I and
477 II by Ushiki et al. (40), who showed that *Nitrospira* sp. ND1 of lineage I was more sensitive to FA
478 compared to *Nitrospira japonica* of lineage II at 100 mg NH_4^+-N/L and pH 8.0. Here, we observed
479 that *Nitrospira* ASV_32 and ASV_47 of lineage I were physiologically more tolerant to FA
480 compared to *Nitrospira* ASV_8 of lineage II, suggesting that either lineage-specific tolerances are
481 distinct at the higher FA concentration applied here (200 mg NH_3-N/L) or that physiological
482 tolerance to FA is a trait that varies at the strain/species level within *Nitrospira*. Regardless, once
483 *Nitrospira* ASV_32 and ASV_47 grew to high enough abundances in the treatment SBR, they
484 likely contributed to the net oxidation of nitrite and the decrease in the NAR that was observed
485 after day 40 of the Treatment Phase. Therefore, the FA-exposure press disturbance strategy
486 employed in this study acted as a selective pressure that impacted the stability of the NOB
487 community composition, and functional degeneracy within the *Nitrospira* sequence variants likely
488 provided the nitrite-oxidizing community with resilience by shifting activity to physiologically
489 resistant community members.

490

491 The above results highlight the need to combine press disturbances that target the distinct
492 physiological traits of multiple functionally degenerate NOB populations, defined at both the inter
493 and intra genus level, to effectively reduce the aggregate activity of the NOB community for
494 energy-efficient nitrogen removal. For example, if future studies support our finding that lineage I
495 *Nitrospira* are more tolerant to FA than those of lineage II, then combining press-disturbances
496 that further target the physiological traits of lineage I *Nitrospira* along with FA-exposure could
497 prove efficacious. Such strategies could entail the maintenance of high dissolved oxygen
498 concentrations, based on adaptation of lineage I *Nitrospira* to low dissolved oxygen environments
499 (34, 36), or the maintenance of low nitrite concentrations based on their preference for higher
500 nitrite concentrations (34, 40, 51). Measurements of maximal growth rates for these sequence
501 variants could also inform wash-out strategies based on limiting the solids retention time (SRT)
502 (13). Acknowledging the potentially tight tolerances and dynamic nature of nitrite, oxygen, and
503 SRT controls required to target the biokinetics of lineage I *Nitrospira* in full-scale AS systems,
504 combinatorial press-disturbance strategies would likely benefit from control strategies that
505 incorporate frequent community monitoring and biokinetic data of these physiologically diverse
506 NOB populations. These findings also underscore the need for broader applications of *in situ*
507 physiology approaches for elucidating the impacts of NOB out-selection strategies on functionally
508 active NOB members within activated sludge systems.

509

510 Accurately measuring the active biomass fraction in microbial bioprocesses is critical, as many
511 key biokinetic models and process mass balances are based on active biomass concentrations
512 (62). However, conventional indirect approaches for quantifying active biomass based on net
513 substrate utilization and growth yields are unable to resolve the compositional and functional
514 dynamics of active microbial populations. As enzymes are the key catalysts that drive the majority
515 of substrate transformations in microbial bioprocesses, we posit that measures of active biomass

516 should be based on the translationally active microbial cell fraction. The translationally active cell
517 fractions in both SBR microcosms seeded with mixed liquor measured by BONCAT-FACS (R;
518 24.1% ± 2.6% in treatment, 30.1% ± 3.8% in control) were less than the active biomass fraction
519 predicted through steady-state modeling (63) (67-77% VSS-basis; Supplemental Information),
520 which is likely because the microcosms were only supplemented with ammonium as an electron
521 donor. Nonetheless, we observed many translationally active heterotrophs in the BONCAT
522 microcosms, which could have remained active through metabolism of internal carbon reserves,
523 endogenous respiration, or constitutive protein expression. The consistent increase in
524 translational activity across all differentially abundant ASVs detected in the BONCAT+ fraction of
525 the control SBR microcosm seeded with sludge following the 24 h sidestream incubation could
526 thus have been attributed to fermentative metabolism on cell decay products. In contrast, the
527 mixed translational responses of differentially abundant ASVs in the BONCAT+ fraction of the
528 treatment SBR microcosm following sidestream incubation with FA suggests that more complex
529 dynamics between cellular inhibition, decay, and fermentative metabolism were induced by FA-
530 exposure. This analysis highlights the value of BONCAT as an *in situ* physiology approach to
531 directly measure concentrations of translationally active population members in a microbial
532 bioprocess. This approach could therefore be extended to provide actual measurements of the
533 active biomass concentration for use in calibrating and validating higher-resolution process
534 models, which have been called for as tools to optimize energy-efficient nitrogen removal
535 technologies (9) as well as to better resolve biogeochemical cycles in natural ecosystems (64,
536 65). BONCAT-FACS could also be extended to resolve associations in the activity of AS taxa with
537 different substrate preferences (48), thus helping to validate ecological-scale models of the
538 wastewater microbiome (66). Therefore, *in situ* physiology approaches like BONCAT show great
539 promise to help inform new strategies to model, control, and engineer microbiome function in both
540 environmental biotechnologies and natural ecosystems alike.
541

542 **Materials and Methods**

543

544 *Reactor setup, operation and monitoring*

545 Two identical lab-scale sequencing batch reactors (SBRs) with working volumes of 4.28 L were
546 seeded with AS from a pilot-scale SBR at a WWTP in King County (Washington, USA) (Figure
547 S1). The SBR cycles lasted 3 h, including 2 min aerobic feeding, 148 min aerobic reaction, 20
548 min settling, 5 min decanting, and 5 min idle periods. The SBR cycle timing was controlled with
549 ChronTrol XT timers (ChronTrol Corporation, USA), and mixing was provided by overhead mixers.
550 Reactor temperature was maintained at 20 ± 1 ^{o}C using an environmental chamber. The target
551 solid retention times (SRTs) were kept at 10 days for both SBRs throughout the study by wasting
552 a determined amount of biomass based on daily values of total suspended solids (TSS) and
553 volatile suspended solids (VSS) measured in the mixed liquor and effluent streams. The SBRs
554 were fed with a synthetic wastewater containing ammonium chloride as the nitrogen source and
555 sodium acetate and propionic acid as organic carbon sources, producing 24.5 ± 2 mg NH_4^+-N/L
556 and 100 ± 37 mg/L of soluble chemical oxygen demand (sCOD), respectively, to reflect a
557 sCOD/NH_4^+-N ratio of ~4.0 typical of North American wastewaters (63). Macro and micro element
558 components of the synthetic wastewater are detailed in the Supplemental Information.

559 Two operational phases were sequentially conducted: the Start-up Phase and the Treatment
560 Phase. In the Start-up Phase, the two SBRs were operated under the same aerobic conditions
561 without FA-exposure of return sludge to achieve similar nitrification performance. The SBRs were
562 fed with 1.07 L of synthetic wastewater in each SBR cycle using peristaltic pumps
563 (LabF1/YZ1515, Shenchen Precision Pump, China), resulting in a hydraulic retention time (HRT)
564 of 12 h. The pH was not controlled but measured within the range of 6.7-7.5. Aeration was
565 provided by an air pump and dissolved oxygen was not controlled, but ranged from 3-8 mg/L in a
566 typical SBR cycle for both reactors. The Start-up Phase lasted 274 days to establish steady-state
567 conditions.

568 The Treatment Phase was commenced on day 275, hereafter referred to as 'day zero', and lasted
569 94 days. The operational conditions in the Treatment Phase were similar to those in the Start-up
570 Phase, except for the following differences (Figure S1). In the Treatment Phase, 800 mL of mixed
571 liquor was removed from each SBR at the end of the reaction period of a given cycle every 24 h,
572 and was thickened to 50 mL by centrifugation. The thickened return sludge (50 mL) was incubated
573 in an unstirred 200 mL beaker along with 100 mL of media with the same composition as the
574 synthetic feed, but with no COD. For the experimental reactor, termed the treatment SBR, the
575 return sludge treatment solution contained 1060 mg/L NH_4^+-N, with pH adjusted to 9.0 using
576 sodium hydroxide to produce a final FA concentration of 200 mg NH_3-N/L. These FA and total
577 ammonia nitrogen concentrations are within the range observed in anaerobic digester centrate,
578 particularly co-digestors and those with thermal-hydrolysis pre-treatment (67, 68). In the other
579 reactor, termed the control SBR, thickened return sludge was incubated in the same medium at
580 a pH of 9.0 but without ammonium addition. After 24 h of incubation, the 150 mL of treated return
581 sludge was recycled back into the respective SBRs at the start of the next cycle. To maintain
582 consistent nitrogen loadings in the two SBRs, 0.406 g of ammonium chloride was added to the
583 control SBR simultaneously with the treated return sludge. Monitoring experiments lasting 24 h
584 were performed approximately every 10 days to measure composite daily nitrite accumulation
585 ratios (NARs), as described in the Supplemental Information.

586 During the Treatment Phase, effluent nutrient samples were collected just before the addition of
587 treated return sludge. Nitrogen species (ammonium, nitrite, and nitrate), orthophosphate, and
588 soluble COD (sCOD) in effluent samples were monitored three to four times per week. pH and
589 DO were measured at least three times per week. Analytical methods for bioreactor monitoring
590 are described in Table S2.

591 *Microcosms for bioorthogonal non-canonical amino acid tagging (BONCAT)*

592 Three 30 mL samples were collected from each SBR throughout the return sludge treatment cycle
593 commenced on day 94, including: (1) a mixed liquor sample immediately preceding bulk mixed
594 liquor removal for return sludge treatment; (2) a return sludge sample after 15 min of side-stream
595 treatment (S1); and (3) a return sludge sample after 24 h of side-stream treatment (S2). The
596 treated return sludge samples were volume-corrected for the thickening process to attain the
597 same biomass concentrations as the mixed liquor samples. Samples were washed in phosphate
598 buffered saline (1x PBS; filter sterilized) through centrifugation (3000 rpm, 5 min) and
599 resuspension to remove residual growth substrates. All samples were incubated in a media
600 consisting of 25% synthetic wastewater in 1x PBS (vol/vol) without COD or yeast extract, so that
601 NH_4^+-N was the only exogenous electron donor. For BONCAT-labeling microcosms, 15 mL
602 portions of each sample were resuspended in 15 mL of incubation media amended with 1 mM
603 homopropargylglycine (HPG; Click Chemistry Tools, USA) (HPG-positive microcosm), transferred
604 into sterile 125 mL Erlenmeyer flasks, and incubated on an orbital shaker for 3 h at 20 °C and 200
605 rpm. Control microcosms (HPG-negative) for each sample were conducted similarly except
606 without HPG amendment. Following incubations, microcosm samples were washed three times
607 in 1x PBS to remove unincorporated HPG, resuspended in 10% (vol/vol) glycerol in PBS,
608 aliquoted into 1 mL and stored at -80°C until further processing. Details on preliminary BONCAT
609 validation microcosms are provided in the Supplemental Information.

610

611 *BONCAT Sample Preparation and Click-Chemistry*

612 For each microcosm type (R, S1, or S2) for both SBRs, sample preparation and click-chemistry
613 were conducted using triplicate HPG-positive and duplicate HPG-negative microcosm samples.

614 Samples were thawed on ice at 4°C, enzymatically homogenized, subjected to filter-immobilized
615 click-chemistry labeling with the FAM picolyl azide dye (Click Chemistry Tools, USA) closely
616 following the procedure of Couradeau et al. (45), detached from the filter, pre-strained through a
617 30 um mesh filter, and counterstained with SYTO59 (10,000x final dilution; ThermoFisher
618 Scientific, USA) to generate the pre-sort samples for FACS. Details of the sample homogenization
619 and click-chemistry labeling procedures are provided in the Supplemental Information.

620 *Fluorescence-activated cell sorting (FACS)*
621
622 Fluorescence-activated cell sorting was conducted on a BD FACSJazz™ cell sorter (BD
623 Biosciences, USA) calibrated to detect the FAM picolyl azide dye (excitation = 490 nm/emission
624 = 510 nm) and the SYTO59 counterstain dye (excitation = 622 nm/emission = 645 nm). An
625 overview of the FACS gating procedures is provided in the Supplemental Information. Briefly,
626 initial gating was established with side-scatter, forward-scatter and trigger pulse width to exclude
627 large particles and cell aggregates. Sorting gates were set targeting BONCAT-positive
628 (BONCAT+) cell fractions based on background SYTO59 and FAM fluorescence, allowing less
629 than 0.5% of false positives (Figures S13-S14). A total of 100k cells were analyzed from each
630 sample, where cells within the sorting gate (SYTO+, BONCAT+) were sorted into 1.5 mL tubes
631 containing 400 µL of prepGEM Wash Buffer (ZyGEM™, USA), and stored at -80 °C until further
632 analysis.

633
634 *DNA Extraction and 16S rRNA gene amplicon sequencing*
635
636 For time-series analysis of community composition, triplicate 980 µL aliquots of mixed liquor were
637 routinely collected directly from the SBRs at the end of a cycle and flash frozen at -80°C. Mixed
638 liquor samples were thawed on ice, and DNA was extracted using the FastDNA Spin Kit for Soil
639 (MP Biomedicals, USA) with minor modifications (69). For each HPG-positive microcosm type (R,

640 S1, or S2) for both SBRs, 50 µL aliquots were collected from the triplicate pre-homogenized and
641 post-homogenized samples during preparation for FACS, added to 400 µL of prepGEM Wash
642 Buffer, and stored at -80°C. Pre-homogenized samples from HPG-negative R microcosms and
643 bulk mixed liquor samples were prepared similarly. All pre-homogenized, post-homogenized, and
644 BONCAT-FACS samples were extracted using the prepGEM Bacteria kit (ZyGEM™) using a low-
645 biomass input procedure overviewed in the Supplementary Information. DNA concentrations were
646 measured with the Qubit® dsDNA BR and HS Assay Kits and Qubit® fluorometer (ThermoFisher
647 Scientific, USA).
648
649 16S rRNA gene fragments from all DNA extracts were amplified using barcoded primers, 515F
650 and 926R (70), targeting the V4-V5 hypervariable region of the 16S rRNA gene. All samples
651 extracted with the prepGEM kit underwent an initial round of 15-cycles of PCR with non-barcoded
652 515F and 926R primers due to the low-biomass input. One triplicate set of FastDNA extracts from
653 day 94 were also pre-amplified to determine potential biases of that step. Amplified barcoded
654 PCR products were pooled at equimolar concentrations and sequenced on an Illumina MiSeq in
655 paired-end 300 bp mode at the UBC Biofactorial facility.
656
657 Amplicon reads were processed and denoised into amplicon sequencing variants (ASVs) with
658 DADA2 (49) (v.1.12.1) in the R environment. The script used to generate the ASV datasets is
659 provided in Supplemental Information. Denoised sequences were taxonomically classified using
660 the RDB Classifier (71) against the MiDAS 3.0 database (69).
661
662 *Statistical analysis*
663
664 Comparisons of reactor nutrient data were performed with t-tests for N-of-1 trials with serial
665 correlation (72). Amplicon sequencing data was visualized using the tidyverse package (73)
666 (v.1.3.0) in the R environment. Principal coordinates analysis (PCoA) of cumulative-sum scaled

667 (CSS) ASV read counts was performed using the metagenomeSeq (74) (v.1.26.3) and vegan (75)
668 (v.2.5.6) packages in R. PERMANOVA (adonis) was conducted in vegan with 1000 permutations
669 to determine significant differences in reactor communities over time. Differential abundance
670 analysis of ASVs between SBRs and in BONCAT microcosm datasets was performed using
671 DESeq2 (76) (v.1.24.0) using parametric fitting, the Wald significance test, and Benjamin-
672 Hochberg correction for *p* values. A $\log_2$-fold change of an ASV represents the multiplicative effect
673 size for changes in normalized read counts across treatments on a logarithmic scale to base 2.
674 Sequence similarity values between ASVs were calculated using the NCBI Basic Local Alignment
675 Search Tool (77). FACS data was processed using the flowcore package (v.1.52.1) (78) in R.
676

677 **Data Availability**

678 The raw read files of 16S rRNA gene amplicons are available via the NCBI Short Read Archive
679 under BioProject PRJNA693634.

680
681

682 **Acknowledgements**
683 This work was supported by the Natural Sciences and Engineering Research Council (NSERC)
684 of Canada (Discovery Grant). MM was supported by an NSERC CGS-M fellowship.

685

686 **Author Information**
687 MM and YL contributed equally to this work.
688

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