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ARTICLE

Enhancement of the capability in wastewater treatment by adding biochar into a conical-shaped sequencing batch reactor

Heritier Apingien Mufwankolo¹, Rosine Huania², Buy Dui Tan¹, Nguyen Binh Minh¹, Hideaki Sano², Tomoaki Itayama²*

¹ Graduate School of Engineering, Nagasaki University, Nagasaki, 852-8131, Japan

² Graduate School of Integrated Science and Technology, Nagasaki University, Nagasaki, 852-8131, Japan

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ABSTRACT

The present study evaluates the manner in which reactor geometry and corn-cob biochar (CCBC) collectively enhance the performance of sequencing batch reactors (SBRs). Four bench-scale SBRs (1.5 L) were operated in parallel, initially in cylindrical vessels and subsequently in conical vessels, followed by CCBC addition (1.5 g and 3.0 g per reactor, with weekly replenishment). Water quality parameters (TOC, $\text{NH}_4^+ - \text{N}$, $\text{NO}_2^- - \text{N}$, $\text{NO}_3^- - \text{N}$, TN), DO, and MLSS were monitored, and FE-SEM was used to confirm biofilm development on CCBC. In conditions of identical operating cycles, the conical SBR exhibited a higher TOC removal rate ($95 \pm 2\%$, effluent 6.7 mg-C/L) in comparison to the cylindrical SBR ($90 \pm 2\%$, 12.8 mg-C/L). This is indicative of an estimated effluent BOD of 26.3 mg/L as opposed to 49.9 mg/L. The effluent total nitrogen (TN) levels exhibited a decline from 21.0 mg-N/L (cylindrical, 3 cycles/day) to 9.6 mg-N/L (conical, 3 cycles/day) and 9.2 mg-N/L (conical, 4 cycles/day). Following the CCBC addition, a marked decrease in effluent TOC was observed (1.1 and 2.2 mg-C/L for 1.5 g and 3.0 g, respectively), which then stabilized at low levels (3.5 and 2.9 mg-C/L on average). Concurrently, there was a significant reduction in $\text{NH}_4^+ - \text{N}$, $\text{NO}_2^- - \text{N}$, $\text{NO}_3^- - \text{N}$, and TN relative to the control groups. A TN mass balance indicated that adsorption contributed only marginally to long-term T-N removal ($< 0.1\%$), with denitrification instead predominating. This was likely promoted by larger anoxic zones in the conical geometry and by CCBC-supported microhabitats. FE-SEM analysis confirmed progressive biofilm colonization on CCBC. This study demonstrated for the first time that synergistic functions are achieved by placing a high-performance carrier, which combines macro-pores suitable for microbial habitation with high organic matter adsorption capacity, into a conical reactor with excellent flow and sedimentation stabilization. This method is considered to be a highly cost-effective approach for the removal of nutrients and organic matter via SBR in decentralised wastewater treatment systems in small-scale facilities and local hospitals in developing countries.

1. Introduction

In the context of developing countries, the treatment of domestic wastewater and urban sewage is a matter of significant

* Corresponding author.

E-mail address: itayama@nagasaki-u.ac.jp

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concern. A comprehensive analysis of this issue must encompass a multifaceted consideration of cost constraints, fluctuations in load and inflow, and the competencies of the operators. SBR is regarded as a promising solution in such circumstances due to its ability to facilitate equalization, reaction, sedimentation, and discharge within a single tank. Additionally, the system facilitates the concurrent removal of nitrogen (N) and phosphorus (P) through cycle regulation, exhibiting resilience against variations (Askari et al., 2024). In India, the adoption of SBR has expanded concurrently with policy and infrastructure development, and evaluations of sewage treatment plant (STP) performance are being disseminated (Central Pollution Control Board [CPCB], 2021). Moreover, SBR has been demonstrated to apply to small and medium-scale systems in actual wastewater treatment examples from Brazil, achieving removal rates of 83% for soluble chemical oxygen demand (SCOD), 60% for $\text{NH}_4^+ - \text{N}$, and 70% for total suspended solids (TSS) (Fernandes et al., 2013). In addition, designs integrating SBR with Up Flow-Anaerobic Sludge Blanket (UASB) as a secondary treatment for advanced treatment are reported to effectively remove additional organic matter and nutrients and be suitable for phased implementation (Khan et al., 2014).

SBR is regarded as a viable treatment method due to its capacity to adapt to load fluctuations while concomitantly reducing biological oxygen demand (BOD) and total TSS and nutrient levels within a single tank (Amin et al., 2024). For instance, at Basaveshwar Hospital in India (Kolur et al., 2015), SBR demonstrated a removal performance of BOD from 290 to 20 milligrams per liter (mg/L) and chemical oxygen demand (COD) from 435 to 55 mg/L (average), thereby substantiating the efficacy of the system in a practical setting. However, in this case, nitrate nitrogen increased, and denitrification was insufficient, indicating issues with nitrogen removal. Consequently, while the SBR process is already employed as a practical system for diverse wastewater treatment, continuous improvement and refinement are ongoing.

In recent years, there has been a shift in research focus on the reactor geometry itself in SBRs. As demonstrated by a few studies, reactor geometry exerts a significant influence on flow characteristics during aeration, thereby affecting the granularity of aerobic sludge (Hamiruddin et al., 2025; Ren et al., 2009). A particularly noteworthy study is that of Desiredy et al. (2023), which demonstrated the effectiveness of conical reactors in SBRs. The study concluded that the combination of the conical shape and aeration creates an effective velocity gradient for granulation formation, promoting organic removal, nitrogen removal via nitrification and denitrification, and phosphorus removal. However, even in the absence of granular sludge formation, the conical shape is believed to offer advantages in SBR systems that necessitate rapid alternation between aeration and settling. As posited by Khumalo (2022), the conical bottom shape is conducive to stable sludge settling in the SBR treatment of brewery wastewater. However, the superiority of conical reactors over other SBR shapes in terms of organic removal capacity or nitrification/denitrification characteristics has not been directly demonstrated, even in the absence of sludge granulation. Consequently, studies that make a comparison between the treatment capacity of conical and cylindrical reactors of the same size in SBRs are essential.

In addition to reactor configuration, material-based approaches such as the incorporation of adsorptive or catalytic media (e.g., biochar) have also been explored to enhance SBR performance. Shi et al. (2019) examined the impact of BC on phenol treatment in sequencing batch reactors (SBRs), as well as the associated changes in microbial communities. In SBRs containing BC, the rate of phenol degradation was enhanced, concomitant with a decrease in the amount of reactive oxygen species (ROS) within the microbial cells. This finding indicates that the BC adsorbed the phenol, thereby reducing its toxicity to the activated sludge as a whole and protecting the microorganisms within it. Conversely, this adsorption effect led to a significant increase in the *Comamonas* and *Cupriavidus* genera, which are the primary phenol-degrading bacteria, due to the phenol concentration at the BC interface. A study was conducted to investigate the efficacy of an SBR with added biochar in the treatment of wastewater containing moxifloxacin (Du et al., 2025). The removal rate of moxifloxacin for the SBR without biochar was 60.31%, whereas the maximum removal rate achieved by the SBR with added biochar was 73.24%. This finding indicates that SBRs augmented with biochar may be a viable treatment option for hospital wastewater. In the context of petroleum refining wastewater treatment, the incorporation of BC, derived from the carbonization of petroleum sludge, into the SBR system has been shown to expedite granular sludge formation by a period of 15 days. Concurrently, the COD removal rate is enhanced by an average of 3%, the oil content is increased by 4%, and the total nitrogen (TN) removal efficiency is elevated by 10% (X. Wang et al., 2020).

These studies demonstrated that adding biochar to the activated sludge treatment process effectively removes recalcitrant substances, including oil residues and cytotoxic compounds, from wastewater. However, separate research is required to confirm its effectiveness in treating typical wastewater that does not contain these substances. Zhang et al. (2022) conducted a study evaluating the effects of sludge-derived biochar on granule formation processes, stability and wastewater treatment efficiency. Their results suggest that adding biochar to activated sludge promotes the formation of aerobic granules and improves long-term stability. However, no significant effects of biochar addition were observed in terms of organic matter removal, nitrification, or total nitrogen removal. By contrast, an SBR system using rice husk biochar achieved approximately 24% higher $\text{NH}_4^+ - \text{N}$ removal and 15% higher TN removal than the control when treating high-nitrogen livestock wastewater (Pan et al., 2024). These discrepancies in the results of the experiments are likely due to differences in SBR operating conditions and biochar characteristics such as pore size distribution and microbial attachment properties. Therefore, the evaluation of the effectiveness of each selected biochar for the enhancement of the SBR function is essential.

Moreover, when utilising biochar in a practical context, it is imperative to select inexpensive and readily available feedstock (Ravindran et al., 2024). Given the diverse effects of biochar observed in SBR systems, identifying cost-effective and sustainable biochar sources is crucial for practical implementation. From this standpoint, non-edible components, such as corn cobs, can be regarded as valuable biochar feedstock.

Indeed, corn is a grain that is cultivated extensively around the world, including in African countries (FAO, 2023; IITA, 2022). Furthermore, these corn cobs are generally disposed of as low-value waste, which renders them readily available and inexpensive as raw materials. Indeed, a substantial body of research has been conducted on the adsorption of various substances using biochar derived from corn cobs (Dang et al., 2021; Obey et al., 2022a, Obey et al, 2022b). However, no studies have evaluated the performance of adding corn-derived biochar to SBR as a solution in rural areas of developing countries. The economic viability of constructing large-scale wastewater treatment facilities is low.

In summarizing the aforementioned points, it is evident that despite the ongoing research into the reactor shape of SBRs and biochar addition, two significant research gaps remain. Firstly, it is necessary to demonstrate that the conical reactor shape effectively enhances treatment performance, including organic and nitrogen removal, even in conditions where sludge does not granulate. Secondly, it is necessary to demonstrate that the addition of CCBC to conical SBRs improves treatment performance for typical municipal sewage and domestic wastewater that does not contain recalcitrant or hazardous substances. The objective of this study is to address these research questions through experimental methods.

2. Materials and methods

2.1 Preparation of materials

In this study, synthetic wastewater (SWW) was used as the influent for the sequencing batch reactors (SBRs). The detailed composition of the SWW is provided in Table 1. The feed solution was freshly prepared for 48 hours use and stored at 2 °C until feeding. Activated sludge was obtained from a wastewater

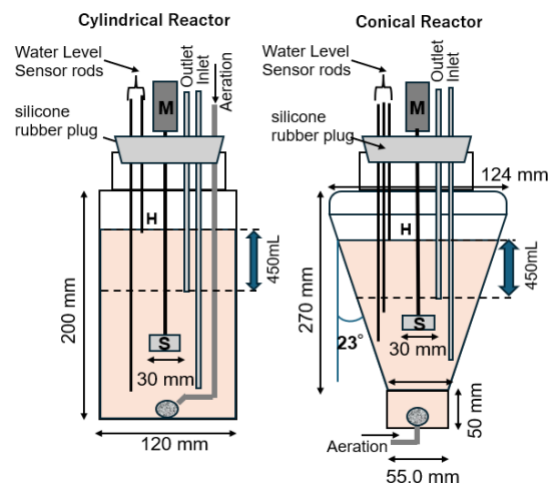
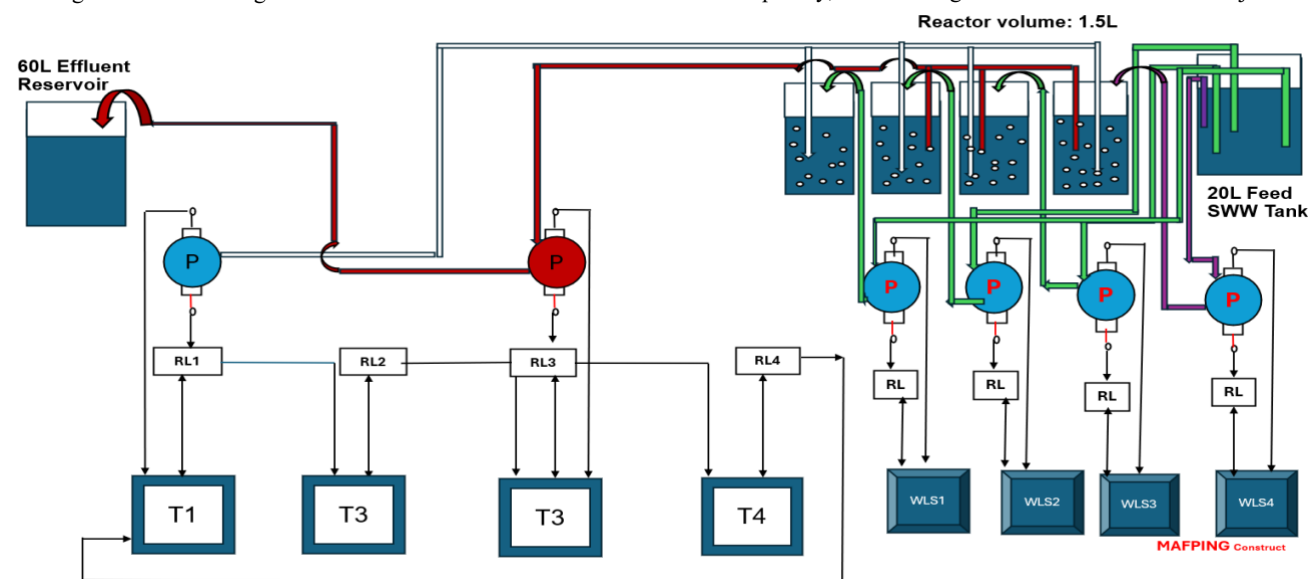


Figure 1 cylindrical shape and conical shape reactors

After a three-day days of sludge cultivation utilizing synthetic wastewater with adequate aeration, the sludge was transferred to the SBRs. Corn cobs obtained from a local farm in Shimabara City, Nagasaki Prefecture, Japan, were processed through a series of steps. Initially, the cobs were cut into smaller fragments. Then, they were subjected to an air-drying process for a period of three weeks. The prepared corn-cob samples, with a mass of 30 grams, were subsequently exposed to a process of thermal decomposition, commonly referred to as pyrolysis, in the context of an electric furnace. The pyrolysis process was conducted at a temperature of 800°C for two hours in argon gas at a flow rate of 5 L/min, accompanied by an air flow rate of 0.1 L/min. The recovery of the biochar as weight base was measured. Subsequently, the resulting carbonized biochar was subjected to



treatment plant serving a food-processing facility and promptly transported to the laboratory for use as inoculum.

additional crushing in a stainless-steel mortar and then sieved through a stainless-steel

Figure 2 Schematic diagram of sequence control system of SBRs: T1, T2, T3, and T4 represent timer numbers from 1 to 4: T1 monitors

the Air blower, T2 operates during sedimentation, T3 monitors water suction pump, T4 operates while water level sensors control supply water pump; RL1, RL2, RL3, and RL4 stand for Relay number 1 to 4; P stands for pump. WLS1, WLS2, WLS3, WLS4 represent Water level sensor from 1 to 4; Water Level Sensor runs the supply water pump (ON when WLSE2 is out of water and OFF when WLSE1 is contacted by water). A power converter is used as with Timer T3. 20L Feed SWW Tank contains the prepared synthetic wastewater stored at 3°C.

sieve to select particles ranging in size from 1 mm to 3 mm. Then the surface area of the obtained biochar was evaluated by the iodine adsorption method (Mianowski et al., 2007). Subsequently, the unused samples of CCBC grains were subjected to a process of washing with pure water. Thereafter, the CCBCs were dried once more at 60 °C under low pressure for a period of 24 hours. Thereafter, they were stored in a dry box until the SEM image could be captured. It is noteworthy that CCBC samples collected from SBRs employed the identical dry method as the preparation for the SEM. The SEM used was a field emission scanning electron microscope (FE-SEM, J S M - I T 8 0 0 S H L , JEOL, Ltd., Japan).

2.2 Experiment setups of the SBRs and the operation of the SBRs

Two distinct geometric shapes, cylindrical and conical, were selected for the examination of shape effects. The 2 L glass reagent bottle (AS one co., Japan) was utilized for the cylindrical reactor. The conical shape reactor was simply manufactured from the 2 L polypropylene triangular flask (AS one co., Japan), as illustrated in Figure 1. The effective capacity of both types of reactor was 1.5 L. The four cylindrical or conical reactors (R1, R2, R3, and R4) were arranged in the same sequence control system, as depicted in Figure 2.

The sequence control system was composed of four timer sets (H3CR-A, Omron Co., Japan) and a floatless water level controller (61F-GP-NH AC100V, Omron Co., Japan). These components were utilized to establish the temporal parameters for the operation of the feeding, suction, and aeration pumps as illustrated in Figure 2. The switching of each pump was achieved via each electromagnetic relay (MY-4, Omron Co., Japan).

The SWW feed volume was set at 450 mL, as ascertained by the water level sensor rod of H-level, and the volume of effluent removal was configured by the tip position of the outlet glass pipe, as illustrated in Figure 1. As demonstrated in Figure 1, the mixing screw (S) was rotated at 90 revolutions per minute (rpm) by the geared motor (M) during the mixing phase.

Table 1 Components of Artificial Wastewater

Compounds	Conc.(mg/L)
Glucose	500
Peptone	20
Yeast Extract	20
Beef Extract	20
KH ₂ PO ₄	10
K ₂ HPO ₄	50
NaCl	5
NaHCO ₃	350
(NH ₄) ₂ SO ₄	225
MgSO ₄ ·7H ₂ O	5
Ca(NO ₃) ₂	7
FeSO ₄ ·7H ₂ O	5

CaCl ₂	5
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This setup allowed for an 8-hour cycle operation to be completed in the first experiment for 200 days by using cylindrical reactors, including the acclimatization period of the activated sludge. The process involved the following steps: feeding SWW for 15 minutes, mixing without aeration for 1-hour, aerobic reaction by aeration for 5 hours and 30 minutes, settling for 1 hour, and removing the effluent from the clarified layer for 15 minutes. In the second experiment, which lasted 45 days, the conical reactors operated in the same sequence method as in the first experiment. In the third experiment, the conical reactors were operated for 60 days more, but the cycle was changed to a 6-hour cycle. The cycle included a feeding phase (15 minutes), a mixing phase without aeration (45 minutes), an aerobic reaction phase with aeration (3.8 hours), a settling phase (1 hour), and a 10-minute effluent removal phase of the clarified layer.

In the fourth experiment, which was conducted after a period of approximately 120 days, 1.5 grams and 3.0 grams of CCBC granules were added to two reaction tanks (R2 and R4), respectively. This choice corresponds to 1g/L and 2g/L of the concentration of CCBC in the reactor. The range of dosage was determined based on the previous report (Kim et al, 2022). Two control reaction tanks (R1 and R3) were configured without the addition of CCBC and were instead filled with activated sludge alone. To maintain stable CCBC levels within the reaction tank, the amount of CCBC lost during periodic sludge removal was replenished on a weekly basis with an equal volume of new material. The quantity of CCBC lost due to sludge removal was measured on a weekly basis by determining the biochar content in the sludge suspension discharged from each of the four reactors. Specifically, the sludge suspension was discharged automatically from each reactor on a daily basis and stored in a refrigerator for a period of one week. Following the one-week suspension of sludge, biochar particles were separated by stainless steel mesh of two sizes, 1 mm (for bigger particles) and 0.25 mm (for smaller particles), respectively. To achieve the removal of adhering biomass and solids, the particles were subjected to a thorough washing process on each stainless-steel mesh. The separated material was then transferred to a pre-weighed glass fibre filter, following which it was dried at 105°C for a period of two hours, and weighed. This measured weight represented the biochar that was lost during the week.

2.3. Water quality analysis

The measurements of mixed liquor suspended solids (MLSS), pH, and dissolved oxygen (DO) were daily conducted by inserting each sensor into each reactor at 2 hours after the commencement of the aeration phase. The measurements were conducted using the MLSS sensor (SS-10F, Kasahara Chemical Instruments Co., Japan), the pH sensor (HM-30P model, TOA DKK Co., Japan), and DO sensor (YSI ProSolo, Xylem Inc., USA). The calibration of the

MLSS sensor was conducted using sludge samples, which were subsequently measured by means of a weight method. In this period, biochar grains with a density ranging from 0 to 2 g/L were incorporated into the sludge, with the aim of assessing the biochar's effect on the sensor value. To estimate the MLSS contribution from activated sludge, the following regression formula from our preliminary study was applied.

$$\text{MLSS (mg. L}^{-1}\text{)} = 2.412 * \text{MLSS}_{\text{sensor}} - 129.8 \text{ CCBC(g. L}^{-1}\text{)} - 2493 \quad (1)$$

$$R^2 = 0.96, \text{ p-value} < 1.772 * 10^{-6}$$

The effluent water sample from each reactor was collected at regular intervals for the purpose of analysing water quality parameters. The frequency of effluent water collection was determined to be three or two times per week. Water samples for the analysis of dissolved species were filtered through a glass fibre filter (GA55, Advantest Co., Japan) before storage at -20°C . Photometric analyses of $\text{NO}_2^- - \text{N}$, $\text{NO}_3^- - \text{N}$, and $\text{NH}_4^+ - \text{N}$ were conducted using commercial kits (Ramda-9000 series, Kyoritsu Chemical-check Lab., Corp., Japan) and a UV-spectrophotometer (UV-1800, SHIMADZU, JAPAN). The analyses were conducted in accordance with the manufacturer's instructions. Total organic carbon (TOC) content of the samples was analyzed using TOC-L (Shimadzu, Kyoto, Japan) following the manufacturer's protocol. Ultrasonic treatment (Nippon Seiki Co., Ltd., Japan) was applied to samples containing large flocs. DOC was quantified using the same method as for TOC by analysing filtered samples using TOC-L. T-N of the effluent was measured by Hosomi's method (Hosomi and Sudo, 1986).

3. Results and Discussion

3.1. Pyrolysis of corn cobs

In this study, it was necessary to prepare a sufficient quantity of the CCBC for the initial addition and continuous addition to supply the reduced biochar granules by extracting the sludge in the SBR operation. Obey et al. (2024) reported that a high adsorption capacity was achieved for CCBC when the temperature was set at 900°C and the pyrolysis time was 2 hours. The discussion addressed the potential implications of air leakage in the electric furnace during the pyrolysis process, suggesting that this could enhance surface area and adsorption capacity. In the preliminary test of the pyrolysis of corn cobs, the condition of 900°C and 2 hours with 0.2 L/min of air flow and 5 L/min of argon gas flow was utilized, resulting in a 10% recovery of biochar. It is imperative to consider the loss incurred during the sieving process, which necessitates a recovery of at least 25%. Therefore, our experiment involved subjecting the material to a temperature of 800°C , accompanied by an air flow of 0.1 L/min for a duration of two hours. This procedure resulted in a recovery of 25%, thereby demonstrating the efficacy of the process. Consequently, 60 grams or more of the CCBCs were obtained from 200 grams of dried corn cobs under the specified pyrolysis condition. Whilst the BET method is generally employed to ascertain the surface area, the iodine adsorption can be utilized as an alternative technique for measuring the surface area of biochar (Dang et al., 2021). In fact,

Mianovsky (2007) and Saka (2012) reported a favourable linear correlation between the two methods (Saka, 2012; Mianovsky et al., 2007). The surface area of the biochar obtained was determined by the iodine method. The result of the experiment was a measurement of $140 \text{ m}^2/\text{g}$.

3.2 A comparison of the performance of cylindrical and conical SBRs.

3.2.1 Changes of Temperature pH, DO and MLSS

Before presenting the results, the selected operating conditions are summarised in Figure 3. In this study, four SBRs were operated in parallel under identical conditions. Prior to biochar incorporation, a thorough assessment was performed to evaluate potential differences in treated water quality among the reactors. With the need to experimentally demonstrate whether the conical reactor configuration can enhance treatment performance—particularly for organic and nitrogen removal—even without well-defined sludge granules and comparative studies between cylindrical and conical SBRs are scarce, especially regarding how aeration and mixing affect nutrient removal and energy efficiency, which are key to reducing operational costs in activated sludge systems, these inter-reactor differences must be assessed with precision to facilitate an accurate evaluation of the effect of biochar addition. The mixed liquor suspended solids (MLSS) in the reactors were maintained at approximately 5,500 mg/L, where MLSS was estimated from the MLSS sensor read value by the equation (1). The temperature was maintained within the range of $20\text{--}25^{\circ}\text{C}$, and the pH was adjusted to the range of 6.5–7.7. As demonstrated in Figure 4, the air flow rate was modified in a stepwise manner to assess the response of dissolved oxygen and other water quality parameters.

Although not explicitly addressed in the data, the topic of odour generation is introduced. This phenomenon can be attributed to the presence of sulfur-reducing bacteria (SRB). The process of sulfate reduction, which is catalyzed by hydrogen sulfide (H_2S), results in the generation of noxious odours and poses potential toxicity risks. The present study observed complete suppression of odour in the conical reactor in comparison with the cylindrical one. This phenomenon is hypothesised to arise from diminished H_2S release, consequent to enhanced gas-liquid separation and mixing dynamics (Andriamanohiarisoamanana, 2016; Liu et al., 2024;). Effective odour management is essential for maintaining reactor stability, as well as for mitigating adverse effects on surrounding areas and avoiding resident complaints. As Zarra et al. (2008) demonstrate, suppressing odour emissions is a critical design consideration in SBRs (Sequencing Batch Reactors) and other biological wastewater treatment systems.

3.2.2 Comparison of the removal of organic matter

To demonstrate the difference in organic matter removal capacity between cylindrical and conical SBRs, the total organic carbon (TOC) concentration (including fine solid particles) in the influent and effluent water was measured. It should be noted that Figure 4 also includes the period following biochar addition; the

effect of biochar will be discussed in the subsequent chapter. The influent TOC concentration averaged 164.4 mg/L across all reactors. Overall, the conical SBR demonstrated superior TOC removal efficiency ($95 \pm 2\%$) in comparison to the cylindrical SBR ($90 \pm 2\%$). This difference was found to be statistically significant at the 0.05 level of significance. This improvement corresponded to effluent TOC concentrations of 6.7 mg/L in the conical reactor versus 12.8 mg/L in the cylindrical reactor.

In this study, BOD in the effluent couldn't be measured in the regular water quality monitoring because it demands the time cost. Therefore, BOD and TOC were simultaneously measured for several samples to conduct the regression for evaluating BOD from TOC values. The obtained formula is as follows:

$$\text{BOD} = 3.9\text{TOC} (R^2 = 0.98) \quad (2)$$

Therefore, the biological oxygen demand (BOD) of 49.9 mg/L was estimated from the total organic carbon (TOC) of 12.8 mg/L by the cylindrical reactor, and the BOD of 26.3 mg/L was estimated from the TOC of 6.7 mg/L by the conical reactor. It is evident that the cylindrical reactor has not been successful in meeting the BOD standard of 30 mg/L in discharge from WWTP by US-EPA, though the conical reactor has been shown to satisfy this standard (U.S. Environmental Protection Agency [EPA], 2023).

3.2.3 Comparison of nitrification and denitrification

The subsequent section presents the results concerning nitrification and denitrification. As demonstrated in Figures 5A and 5B, alterations in nitrite concentration are distinctly discernible in both the cylindrical reactor and the conical reactor. The origin of the time axis illustrated in Figure 5B corresponds to the transition point from the cylindrical reactor to the conical reactor. In both periods, the levels of nitrite, nitrate and ammonia were measured at the end of the aeration phase. The aeration phase lasted for 5.5 hours in the cylinder reactor and 3.8 hours in the conical reactor. During the preliminary phase of the experiment conducted in the cylindrical reactor, there was a gradual decrease in nitrate concentration from approximately 6 mg-N/L to 2 mg-N/L over a period of 60 days. The concentration, as demonstrated in Figures 6A and 7A, a synchronous decrease in nitrate, ammonia, and nitrite. The decline in ammonia concentration is attributable to the activity of ammonia-oxidizing bacteria (AOB). Denitrification during the settling phase of the SBR process plays a pivotal role in achieving the desired nitrate reduction.

It is a widely accepted conventional theory that the nitrification process in activated sludge is primarily governed by the activity of nitrite-oxidizing bacteria (NOB) (Sepehri & Sarrafzadeh, 2019); (Daims et al., 2015). In comparison to AOB, NOB demonstrates a reduced growth rate, and their initial nitrite oxidation activity is inadequate. However, the gradual increase in NOB suggests a slow, progressive decrease in nitrite concentration in the effluent. However, the nitrite concentration in the cylindrical reactor did not decrease below approximately 2 mg-N/L. The persistence at this concentration level indicates that the nitrification process in the activated sludge remained inadequate (Su et al., 2023). Following the enhancement of the aeration process, the levels of dissolved

oxygen (DO) were confirmed to have increased to approximately 6 mg/L by day 150. However, the nitrite concentration remained at approximately 2 mg-N/L. This outcome is presumably ascribable to the cylindrical configuration of the reactor. In this context, the configuration of the cylindrical reactor assumes significant importance. We observed that sludge accumulates in the corners at the bottom, potentially forming localized areas with low dissolved oxygen. It is hypothesized that these areas inhibit the nitrite oxidation process. Consequently, the transfer of the sludge to a conical reactor without corners at the bottom resolved the accumulation problem. The sequence of events depicted in Figure 5B commences five days after the concluding plot point in Figure 5A. In addition, Figures 6A and 6B, as well as Figures 7A and 7B, exhibit a similar relationship. The results of the study demonstrated a gradual decline in nitrite levels over 50 days. This finding suggests that the uniform agitation of the sludge within the conical reactor facilitated the permeation of dissolved oxygen throughout the entire activated sludge mass, thereby enabling the initiation of slow proliferation by nitrite-oxidizing bacteria. It has been established that nitrite-oxidising bacteria exhibit a reduced nitrogen utilization rate and a more languid growth pattern in comparison to ammonia-oxidising bacteria (Sepehri, A., & Sarrafzadeh, M. H. (2019). Consequently, this delayed response can be considered a rational outcome.

As demonstrated in Figure 8, the conical reactor consistently exhibits higher dissolved oxygen concentrations than the cylindrical reactor at equivalent aeration intensity. These observations are consistent with the findings of previous studies, which have demonstrated a relationship between reactor geometry and aeration rate, thereby confirming improvements in volumetric oxygen transfer coefficient, standard oxygen transfer efficiency (SOTE), and oxygen mass transfer performance in conical reactors (Adachi, 2015; Klöckner et al., 2013; Klymenko et al., 2025).

The enhanced oxygen supply efficiency also exerted a favourable influence on ammonia oxidation. Indeed, Figure 7A demonstrates that, after 5.5 hours of aeration in the cylindrical reactor, the averaged ammonia concentration was 1.5 mg-N/L over a period of 50 to 200 days. Conversely, as demonstrated in Figure 7B, the conical reactor exhibited an average of merely 0.25 mg-N/L, even after a duration of 3.8 hours of aeration. The experimental results unequivocally demonstrate the necessity of appropriate shape design (Díez et al., 2007).

In a standard nitrogen removal process, organic nitrogen and ammonia present within the total nitrogen components are converted to nitrate during the nitrification process. Subsequently, the nitrate is converted to nitrogen gas via the denitrification process. As shown in Figure 7A, the nitrate concentration in the cylindrical reactor exceeded 50 mg-N/L during the initial stage. Considering the total nitrogen concentration of the synthetic influent wastewater (66.9 mg-N/L), it is evident that the majority of nitrogen was retained during the nitrification stage. This result indicates that denitrification activity was minimal during the initial stage. After 50 days, during the steady-state phase of the cylindrical reactor, the nitrate concentration in the effluent decreased and was confirmed to reach approximately 15 mg-N/L. Subsequently, transitioning the reactor shape from cylindrical to conical further enhanced denitrification activity, accelerating the reduction of nitrate.

As illustrated in Figure 9A, the TN levels in the effluent underwent significant alterations. The trends exhibited by the effluent TN from cylindrical and conical reactors exhibited a high degree of similarity to the trends observed in nitrate levels in the effluent. The TN of the influent SWW was found to be 66.9 mg-N/L for all reactors. Furthermore, the average effluent concentration for four cylindrical reactors with three cycle operations per day was 21.0 mg-N/L (S.D. 1.7 mg-N/L) in the stationary state after 80 days from the start day until 200 days. Following the alteration of the conical reactor within the same cycle, a decline in the effluent TN was observed, reaching a stationary level of 9.6 mg-N/L (S.D. 0.2 mg-N/L). Subsequent adjustment of the cycle number to four per day resulted in a decrease in the conical reactor to 9.2 mg-N/L (S.D. 0.1 mg-N/L). The removal percentages TN were evaluated as 68% (cylinder, three cycles), 86% (conical, three cycles) and 86% (conical, four cycles), respectively. The alteration in the cycle number in operation is indicative of the modification in the flow rate per day. To elaborate, the treatment capacity for the three cycles is 1.35 L/day, whereas for the four cycles, it is 1.8 L/day. The mean daily removal of TN was 61.9 mg-N/day for three-cycle operation of the cylindrical reactor, 77.3 mg-N/day for three-cycle operation of the conical reactor, and 103.7 mg-N/day for four-cycle operation, respectively. However, to ascertain whether the conical reactor shape enhanced the denitrification reaction, it is also necessary to consider the contribution to TN removal by discharging sludge from each reactor. For the duration of the same term of the cylindrical reactors with three cycles, 180.3 mg of sludge were removed on a daily basis from the reactor. This is equivalent to 16.2 mg-N per day, as indicated by the daily total nitrogen removal. In the course of the conical reactor experiment, which comprised three cycles, a total of 243.8 mg of sludge was discarded, along with 136.5 mg of sludge for each of the four cycles. The values thus correspond to daily nitrogen removal rates of 21.9 mg-N and 12.3 mg-N, respectively. In this estimation, the typical factor 0.09 was applied to convert from a MLSS value of activated sludge to a TN (PON) value (Erkan et al., 2017; Myszograj & Pluciennik-Koropczuk, 2020).

Denitrification and sludge waste were identified as the primary factors influencing TN removal. The daily TN removal rate for the cylindrical reactor (across three cycles) by the denitrification was estimated at 45.7 mg-N/day (= 61.9-16.2). For the conical reactor, TN removal rates of 55.4 mg-N/day (= 77.3-21.9) and 112.8 mg-N/day (= 125.1-12.3) were observed for three- and four-cycle operations, respectively. It was thus determined that the denitrification efficiency of the conical-shaped reactor exceeded that of the cylindrical-shaped reactor.

The process of denitrification can only be achieved under conditions of minimal or absent dissolved oxygen (DO), typically below 1.0 mg/L (Oh & Silverstein, 1999.; P. Zhang & Qi, 2007; X. Zhang et al., 2017). In SBR systems, such anoxic conditions were readily established by stopping aeration. This enabled denitrifying bacterial communities to utilize nitrate as the terminal electron acceptor (Toledo-Alarcón et al., 2025). In this study, the anoxic phase was initiated immediately following the cessation of air supply. The average dissolved oxygen (DO) concentration in the

intermediate layer of the accumulated sludge in the cylindrical reactor was 1.2-0.9 mg/L 20 minutes after cessation. In the conical reactor, the average DO concentration in the intermediate layer was found to be 0.4-0.5 mg/L 20 minutes after cessation. The DO concentration in the surface layer was found to be within the range of 1.2-1.9 mg/L at that time.

Furthermore, the depth of the settled sludge in the cylindrical reactor after 20 minutes was approximately 5 cm. The volume of the settled sludge was determined to be 550 millilitres, based on the reactor size shown in Figure 1. Conversely, in the conical reactor, the depth was approximately 17 cm, and the volume of the settled sludge was calculated to be 930 mL. As depicted in Figure 3c, the MLSS of the sludge exhibited relative constancy upon transitioning from the cylindrical reactor to the conical reactor, thereby indicating that the microbial biomass involved in denitrification remains consistent in both reactors. Therefore, if denitrification occurs within the settled sludge zone, it is hypothesized that denitrification will proceed with an efficiency that is 1.7 times higher based on this volume ratio. Furthermore, the cone-shaped reactor has been found to have a lower DO than the cylindrical reactor, suggesting a higher level of denitrification efficiency. The disparity in DO distribution within the settled sludge between the cylindrical and cone-shaped reactors is a highly intriguing issue. However, this necessitates the mathematical treatment of the reactive settling process, encompassing the sludge settling process, and demands the utilization of highly sophisticated mathematical modelling techniques, necessitating the collaboration of computational fluid dynamics experts (Bürger et al., 2022). Conversely, the discrepancy in the size of the settled sludge between the conical and cylindrical reactors, despite equivalent MLSS, can be rationalized by existing research that pertains to the size of sludge flocs in both reactors. Desiredy et al. (2023) denotes the observation that the substantial velocity gradient within the conical reactor fosters the formation of granules. Notwithstanding the absence of granulation in this study, an augmentation in sludge floc size was observed following the transition to the conical reactor. This is hypothesized to have resulted in an increase in the volume of settled sludge. Furthermore, within a certain range of velocity gradients, an increase in gradient size has been shown to increase collision frequency and promote floc growth (Aktas et al., 2013). However, if the velocity gradient is excessively large, shear destruction occurs and flocs become smaller. It is therefore vital to adjust the aeration rate in the conical reactor. Moreover, an increase in floc size has been shown to be associated with an increase in the size of the anaerobic zones within the flocs, which may also favour the denitrification process (Peng et al., 2024).

Concurrently, the residual TOC remained above the stoichiometric organic carbon demand required for complete denitrification in both reactors, thereby providing sufficient electron donors to sustain microbial metabolism. In this study, as previously discussed, the effluent from the conical reactor exhibited an estimated biological oxygen demand (BOD) of approximately 25 mg L⁻¹. According to the *Nutrient Control Design Manual* published by the U.S. Environmental Protection Agency (2009), the removal of 1 mg-N of nitrate requires

approximately 4 mg L^{-1} of biodegradable organic matter (as BOD), and extending the duration of the anaerobic phase can facilitate up to 6 mg-N L^{-1} of additional nitrate removal from the effluent. This correspondence suggests that the BOD levels observed in the conical configuration were sufficient to sustain effective denitrification under the applied operating conditions. These findings are consistent with previous reports indicating that denitrification performance in SBRs is contingent on the balance between aerobic and anaerobic phases, the availability of readily biodegradable organic carbon, and cycle period control (Jiang et al., 2024). However, in this study, to clearly demonstrate the effect of biochar addition, the anaerobic phase duration was deliberately maintained at one hour, thereby sustaining an intentionally insufficient denitrification state.

Finally, in this section, it is also important to note that denitrification plays a crucial role in preventing alkalinity recovery and the subsequent decrease in pH that is caused by nitrate reduction (Henze, 1997; Lin et al., 2021).

3.3. The effect of the biochar addition

3.3.1 The effect to water qualities and the organic matter removal

As demonstrated in Figure 3, fluctuations in temperature, pH, MLSS, and DO were observed during the period after CCBC addition. During this period, temporal fluctuations in temperature decreased, but a gradual temperature increase was observed. The dissolved oxygen concentration was found to stabilize at approximately 6 mg/L , with no significant differences observed between SBRs with or without biochar addition. Concurrently, the suspended solids concentration in the SBR (R4), where CCBC (3 g) was added initially, was higher than in the SBR (R2) where 1.5 g was added, and the SBRs (R1 and R3) where no CCBC was added. It is important to note that in this case, CCBC was added weekly to replenish the biochar lost during the discharge of excess sludge from each SBR. The weekly average addition rate was 0.18 g for R2 and 0.23 g for R4.

Furthermore, an elevated pH level was observed in the biochar addition reactor in comparison to the no-biochar reactor. The underlying reason for the observed increase in MLSS in the presence of biochar is relatively straightforward. On the other hand, the increase of pH by the biochar addition could be caused by the ash component.

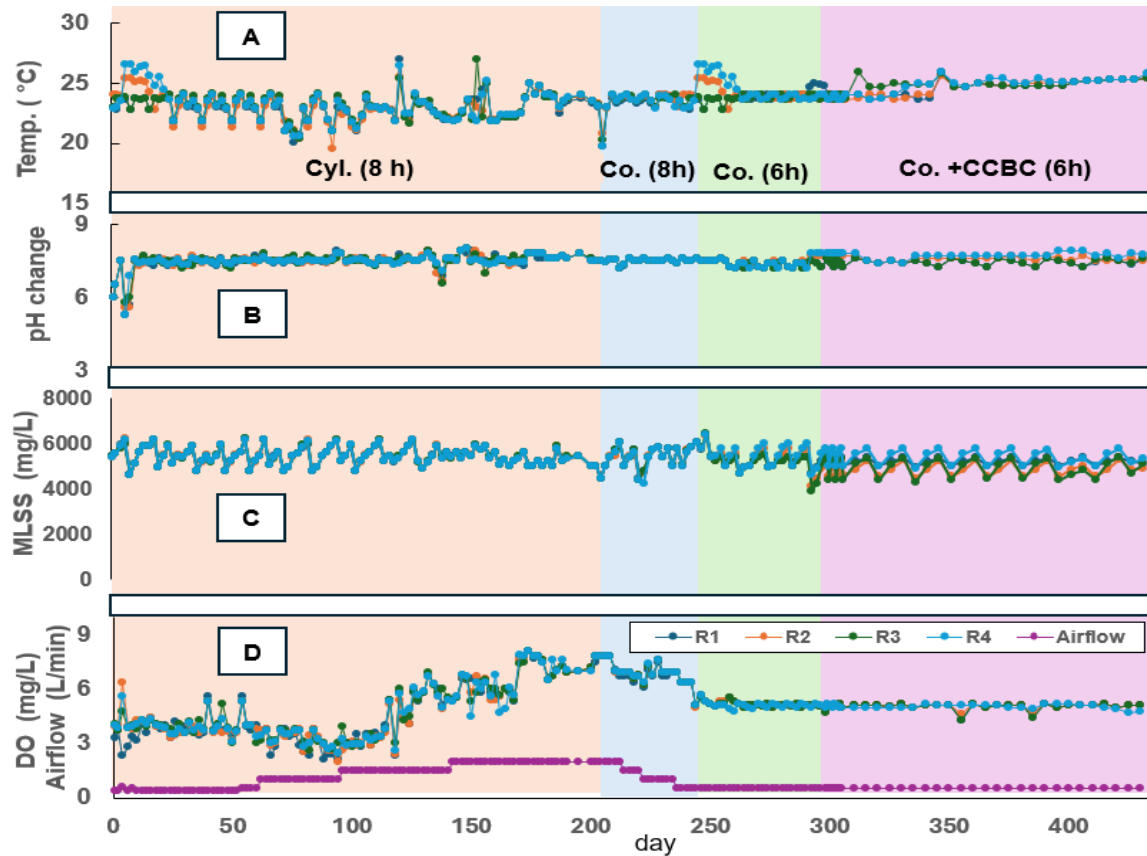


Figure 3 Observed data during full experiment operation time from cylindrical shape reactors to conical reactors and CCBC addition of A) Temperature, B) pH, C) MLSS, and D) DO and Airflow.

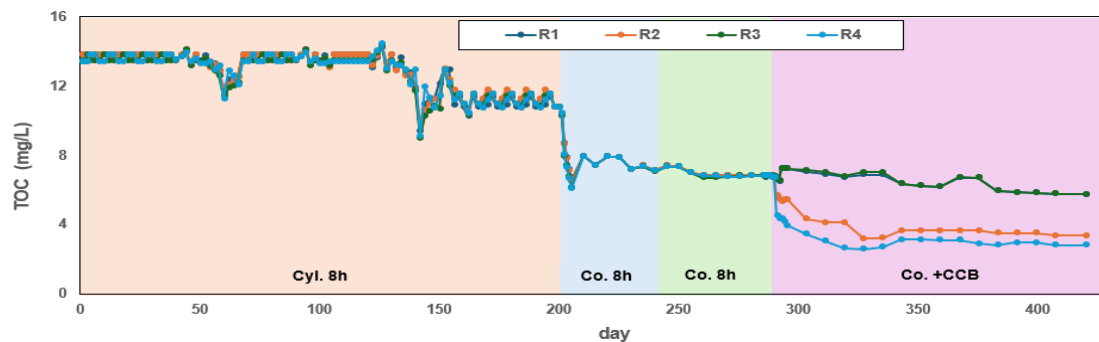


Figure 4 TOC concentration changes of the effluent from the full experiment from cylindrical shape reactors to conical reactors to CCBC addition.

Table 2 Mass balance of T-N from CCBC addition to completion. The SBR experiment continued for 129 days after CCBC addition. The conversion factor 0.09 TN(mg-N) / Sludge (mg) was applied.

SBR	Control (R1 and R3)	R2	R4
TN removal (mg-N)	13612.99	14121.08	14517.86
Initial Addition of CCBC (g)	0.00	1.50	3.00
Total Addition of CCBC (g)	0.00	5.22	6.94
TN Adsorption (mg-N)	0.00	14.09	18.74
Difference of sludge amount between initial and final (mg)	665.90	749.32	559.28
Accumulated TN as sludge (mg-N)	59.93	67.44	50.34
Waste sludge (mg)	6538.44	9983.90	12051.30
TN removal through the waste sludge (mg-N)	588.46	898.55	1084.62
TN removal by denitrification (mg-N)	12964.60	13141.00	13364.17
Enhancement for denitrification by biochar addition (mg-N)		176.40	399.56

Despite undergoing washing with pure water, it is acknowledged that biochar may contain trace amounts of alkali metals and alkaline earth metals derived from ash. These residues have been observed to induce a modest increase in alkalinity and elevate the pH level. Moreover, the elimination of nitrate ions through processes such as denitrification has been shown to result in an increase in alkalinity, consequently leading to an elevation in pH.

As demonstrated in Figures 5 to 9, the significant responses to the biochar addition were revealed in the temporal changes of the TOC, nitrite, nitrate, ammonia, and TN. These water quality

parameters immediately decreased when CCBC was added to the SBRs.

The effluent TOC concentration of R2-SBR (with 1.5 g of CCBC added) decreased from 6.7 mg C/L to 5.6 mg C/L (a decrease of 1.1 mg C/L), and the TOC of R4-SBR (with 3 g of CCBC added) decreased from 6.7 mg C/L to 4.5 mg C/L (a decrease of 2.2 mg C/L). Given the proportionality of the decrease in relation to the amount of CCBC added, and the observed prompt response, it can be reasonably concluded that this decrease was achieved through CCBC adsorption. While a decline in TOC was observed in the presence of CCBC, the TOC in both control SBRs

devoid of biochar (R1 and R3) exhibited a minor increase. It is hypothesised that a minor disruption may have impacted both reactors during the biochar addition process.

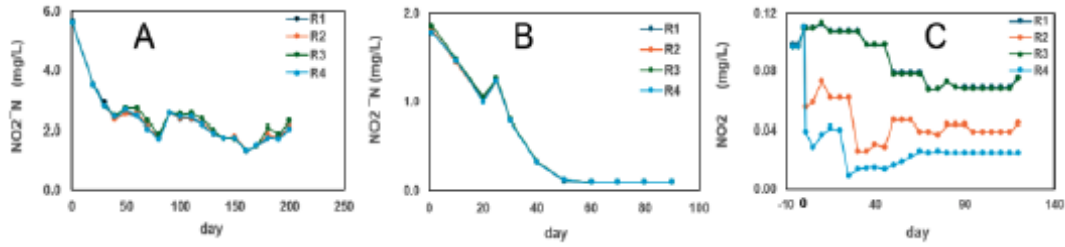


Figure 5 Nitrite removal performance during the start-up of the SBR process of: (A) Cylindrical reactors, (B) Conical reactors, (C) CCBC amended-Conic reactors.

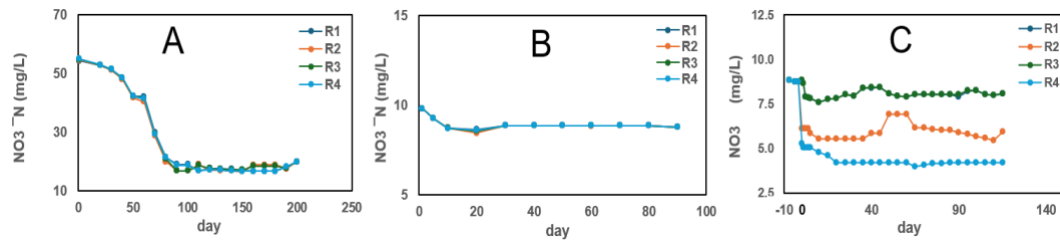


Figure 6 $\text{NO}_3^- \text{N}$ removal performance during the start-up of the SBR process of: (A) Cylindrical reactors, (B) Conical reactors, (C) CCBC amended-Conical reactors.

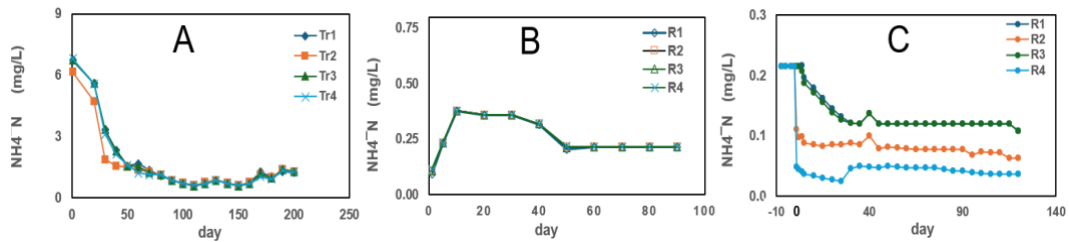


Figure 7 $\text{NH}_4^+ \text{N}$ removal performance during the start-up of the SBR process of: (A) cylindrical reactors, (B) Conical reactors, (C) CCBC amended-Conical reactors.

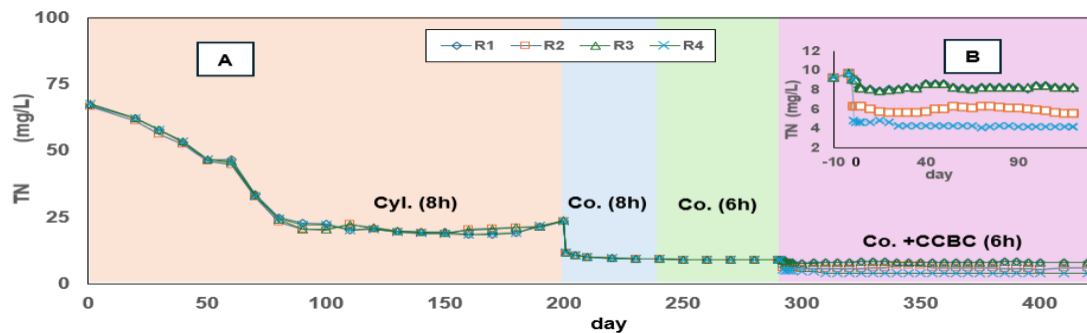


Figure 9 A) TN concentration of the effluent of the full experiment from Cylindric shape reactors to conic reactors to CCBC addition. B) Magnified TN concentration in CCBC amended-SBRs phase.

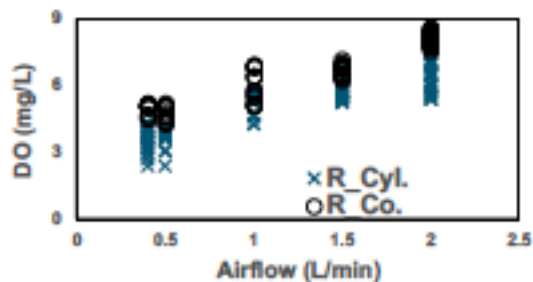


Figure 8 Observed data for Dissolved oxygen relationship and dependency with airflow speed.

After the adsorption by the biochar, the gradual decrease of TOC was observed in the R2 and the R4 for 40 days, and both suggested the enhancement of biological activity by the biochar addition. This phenomenon is likely linked to enhanced microbial colonization and metabolic activity, since biochar offers a high surface area and redox-active sites that stimulate biomass growth and respiration (Jiang et al., 2024).

From 40 days after the CCBC addition up to the conclusion of the study, the average TOC in the effluent from the R2 and R4 reactors was 3.5 mg-C/L (as BOD 14 mg/L) and 2.9 mg-C/L (as BOD 11 mg/L), respectively. In contrast, the control reactors R1 and R3 demonstrated 6.2 mg-C/L (as BOD 24 mg/L). The hypothesis that this significant effect could be sustained was supported by the CCBC weekly added to the R2 and R4. Conversely, the TOC in the effluents from the control SBRs, as illustrated, gradually decreased. As illustrated in Figure 3A, there is a slight increase in temperature. This has led to the hypothesis that the microbiological activity in the activated sludge may be enhanced, thereby facilitating effective degradation of organic matter depending on the temperature.

The underlying mechanisms of organic removal in SBRs are well established, as microbial consortia utilize organic carbon as a primary substrate during aerobic and anoxic phases, leading to effective mineralization and stabilization of wastewater organics (Tchobanoglous et al., 2014; Liu & Tay, 2004). In addition to microbial metabolism, the incorporation of biochar into SBRs provides supplementary removal pathways. Specifically, mechanisms such as molecular–biochar electrostatic interactions, hydrogen bonding, π – π stacking interactions, and surface complexation have been reported to facilitate the sorption and subsequent reduction of dissolved organic matter in biochar-amended systems (Gupta et al., 2016). These synergistic processes not only enhance total organic carbon (TOC) removal but also stabilize reactor performance by providing adsorption sites that buffer against fluctuations in organic loadings and by promoting the enrichment of microbial consortia within biochar pores (Hu et al., 2005). Hence, the integration of biochar with conventional SBR treatment represents a promising approach to improve organic pollutant removal efficiency under varying operational conditions.

3.3.2 The effect on the nitrogen removal

As demonstrated in Figures 5C, 6C, and 7C, the removal of nitrite, nitrate, and ammonia in the SBR after CCBC addition is evident. In accordance with the preceding hypothesis, the low concentrations in the effluent from reactors R2 and R4, where biochar was added, decreased more rapidly compared to the control groups R1 and R3. As demonstrated, the mean nitrate concentration was 5.95 mg-N/L in R2 and 4.11 mg-N/L in R4, in comparison to 8.35 mg-N/L in R1 and R3. Ahead of the implementation of CCBC, ammonia and nitrite concentrations in all SBRs were within acceptable low ranges. However, following the CCBC addition, these concentrations underwent a precipitous decline in R2 and R4. The mean nitrite concentration was determined to be 0.047 mg-N/L in R2. The mean ammonia concentration was found to be 0.081 mg-N/L in R2 and 0.041 mg-N/L in R4, while the mean in R1 and R3 was 0.171 mg-N/L.

During the initial adsorption phase, when biochar was first introduced into the SBR, rapid removal efficiency for each nitrogen species was demonstrated. Indeed, preliminary studies on CCBC revealed adsorption capacities of 3.39 mg-N/g for nitrate and 3.64 mg-N/L for ammonia under incubation conditions of pH 7.5–6.8 for 24 hours. The process of ammonia removal may be explained by the theory of CCBC adsorption. Conversely, the nitrate reduction achieved by adding 1.5 g of CCBC to the SBR (1.5 L) was estimated to be 3.39 mg-N/L as the reduction concentration. The addition of 3.0 g of CCBC was predicted to result in the removal of 6.78 mg-N/L of nitrate via the adsorption process. This estimated result indicates that the adsorption process in the SRB fell below the efficiency predicted by the ideal adsorption experiment. This outcome is hypothesised to be attributable to the presence of competing substances inhibiting the adsorption process (Xiong et al., 2022; J. Yang et al., 2017). In addition, despite the regular provision of CCBC granules every week, it is highly probable that the adsorption capacity of biochar for nitrate was inadequate to sustain the low concentration in the effluent. Consequently, a TN-based material balance assessment is imperative for comprehensive analysis.

The overall TN removal was calculated for R2-SBR, R4-SBR and the two controls after the addition of CCBC until the end. For this calculation, it was assumed that the stable inflow of TN was 66.9 mg/L of SWW. As the two controls (R1 and R3) demonstrated almost identical values, we used the average of the two. As illustrated in Table 2, each element in the mass balance was calculated. To estimate denitrification as the major effect for TN removal, the following mass balance equation was assumed:

$$\sum \text{TN removal} = \sum \text{Denitrification} + \sum \text{TN in the wasted sludge} + \sum \text{Adsorption by Biochar} + \sum \text{TN in the increased sludge}$$

The values of the terms in this equation were calculated over the 129 days from the commencement of CCBC addition to the conclusion of the experiment. The Σ symbol is employed to denote the summation calculation over the specified period. To estimate the total nitrogen (TN) content in the waste sludge from the discharged sludge, a conversion factor of 0.09 (mg-N/MLSS) was applied, as outlined in Section 3.2.2. In addition, it is necessary to determine an adsorption capacity of 2.7 mg-N/g-CCBC (= 2.7

mg-N/L $\times$ SBR capacity 1.5 L / CCBC 1.5 g). This is due to the assumption that the addition of 1.5 g of CCBC to the R2-SBR resulted in a reduction of TN by 2.7 mg-N/L through adsorption. Utilizing the aforementioned calculation method, the TN adsorption capacity in the R4-SBR was evaluated as 1.85 mg-N/g. In this case, to evaluate the denitrification process more stringently, the larger capacity of 2.7 mg-N/g-CCBC was adopted.

As demonstrated in Table 2, TN removal was predominantly accomplished through denitrification in all cases. The second largest contribution was attributed to TN removal via sludge discharge, accounting for approximately 5–8% of the total. However, in both R2-SBR and R4-SBR, the adsorption effect of CCBC was found to be largely ineffective for TN removal during the period, with the adsorption-based TN removal rate remaining at approximately 0.1% in each case. As demonstrated in Figure 9, the rapid decrease in TN can be attributed to the adsorption process. However, it is evident that adsorption rapidly reaches its maximum capacity, i.e., saturation. Consequently, it is hypothesized that the incorporation of biochar expedites the denitrification reaction. The hypothesis that denitrification is enhanced is predicated on the premise that this process occurs within the microhabitat of the bacterial consortium located on and within the biochar. The SEM observations further confirmed microbial colonization dynamics on CCBC grains. In the SEM images of unused CCBC, as exhibited in Figure 10, white spots were observed in the blue closed curves. These may be hydroxide metals, which have been shown to strongly reflect electron beams of the FE-SEM. In the remaining images of CCBC grains from the R2 and R4 samples, the presence of white spots was confirmed. However, it was determined that these were not attributable to microbes or biofilms.

One month after the CCBC addition, SEM analysis of R2 S-T and R4 S-T samples confirmed the presence of microorganisms on the material surface and within the pores. The red closed curves in these images are indicative of microorganisms. As illustrated in Figure 10 (R2 L-T and R4 L-T), the results of the observation of microorganisms attached to the CCBC surface over a period of three months are demonstrated. The biofilm-like layer forming on the surface appears to be more developed on L-T than on S-T. The presence of numerous detached biofilms and shrunken microorganisms is thought to have rendered the observation process particularly arduous. However, the SEM images are analogous to biofilm images on biochar reported in other studies, clearly indicating that microorganisms inhabit the biochar (Changlor et al., 2025). Given the prevalence of denitrification among heterotrophic bacteria, it is reasonable to hypothesise that denitrification is also occurring among the observed microorganisms (Ouyang et al., 2024; Sui et al., 2024; Zumft, 1997). The potential enhancement of denitrification may be attributed to the simultaneous adsorption of organic matter and nitrate, as demonstrated in Figures 4 and 6C. In studies of bioactive carbon (BAC), adsorption has been shown to maintain high concentrations of nitrate and dissolved organic matter (DOM) in the vicinity of the CCBC surface. It is hypothesised that this will synergistically enhance bacterial activity related to denitrification (X. Yang et al., 2025).

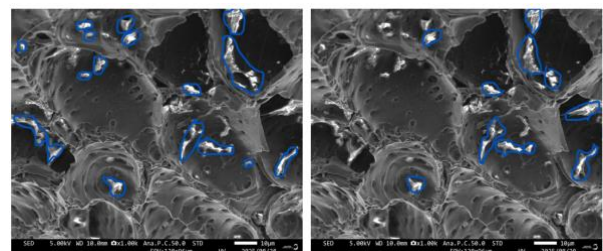
As posited by Shi, S. et al. (2019), the promotion effect of organic accumulation on the surface and biodegradation by bacteria in the biofilm on phenol degradation in BC-added SBR has

been demonstrated. This mechanism has been considered to be effective even for substances that are not difficult to degrade, such as phenol. Consequently, it is hypothesized that the incorporation of biochar into the SBR process promotes the long-term stable removal of organic matter through a dual mechanism of adsorption and biodegradation. Furthermore, it is hypothesized that the enhanced fluidization and efficient oxygen supply provided by the conical reactor further enhanced the effectiveness of the biochar.

Recent studies have indicated that the process of sludge granulation can be significantly enhanced by the introduction of activated carbon of an appropriate particle size. As demonstrated by Zhou et al., (2015), activated carbon with a diameter of 0.2 mm was found to function as a nucleation site. This process was found to accelerate floc–carbon coaggregation and aerobic granule formation. These granules have the potential to enhance the quality of the effluent water. Conversely, reactors that were amended with larger particles (0.6 mm) did not demonstrate a substantial enhancement in granulation, which is presumably attributable to hydrodynamic effects. Increased size of particles has been shown to result in greater velocity field differences between flocs and carbon particles, thereby reducing the stability and lifespan of floc–carbon aggregates. It can thus be concluded that the size of the particles is of crucial importance in determining whether activated carbon can effectively serve as a nucleating agent for granulation. However, the biochar-amended SBRs demonstrated the capacity to effectively remove organic material and total nitrogen (TN). It is hypothesised that the macro pores observed in Figure 10 may provide a suitable habitat for protozoa and rotifers (Thies & Rillig, 2009). It is hypothesised that these micro-organisms may have a significant impact on the performance of SRBs with biochar grains. The present study aims to demonstrate the effect of biochar for the improvement of water quality in the effluent from the SBRs. Further study is required to investigate this system from the perspective of microbiological study. This knowledge is of significant value, as it is instrumental in the development of more efficient systems.

The findings, which have been revealed for the first time in this study, are expected to lead to practical applications demonstrating high cost-effectiveness, particularly in small-scale, decentralized wastewater treatment systems (e.g., small hospitals in rural areas of developing countries, industrial facilities, and livestock facility effluent).

Unused CCBC



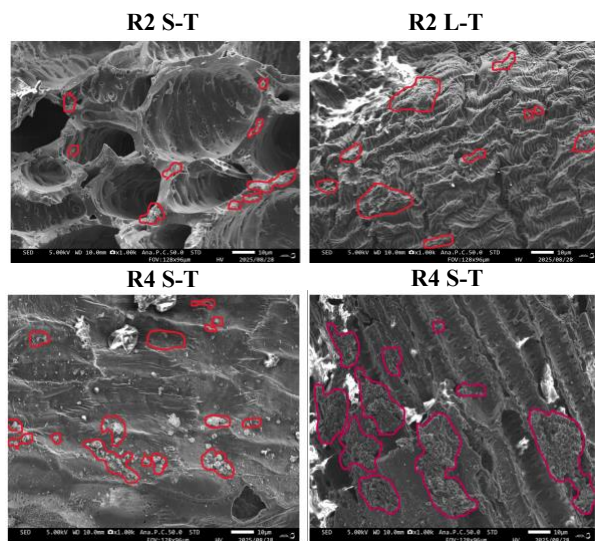


Figure 10 CCBC surface view: Before addition in the reactors (Unused CCBC, and sludge flocs (biofilm) formed due to bacterial coagulation on the CCBC surface in Reactor 2 and 4 for short-term operation (R2 S-T, R2 S-T) and long-term operation (R2 L-T, R4 L-T).

Markers are used to show the Ca or K or Mg deposited metal (in white) then red for Biofilm or bacteria floc (Changlor et al., 2025; Zama et al., 2022)

Conclusion

The conical SBR consistently outperformed the cylindrical reactor under equivalent operating conditions, achieving lower effluent TOC (6.7 mg-C/L vs. 12.8 mg-C/L) and TN (9.2–9.6 mg-N/L vs. 21.0 mg-N/L). The addition of corn-cob biochar (CCBC) led to an immediate reduction in TOC through adsorption, while long-term nitrogen removal was primarily driven by enhanced denitrification, with adsorption contributing less than 0.1%.

The conical shape of the reactor led to the formation of a substantial anaerobic layer following the cessation of aeration, thereby enhancing denitrification efficiency during the fixed anaerobic period. Furthermore, FE-SEM analysis confirmed the presence of microbial colonization within the CCBC macro-pores. This finding confirmed that microorganisms functioning synergistically with the adsorption capacity of CCBC were active in organic matter and nitrogen removal. In addition, the conical shape has been demonstrated to encourage the removal of organic matter and nitrification by enhancing flow conditions and augmenting oxygen supply to microorganisms. Consequently, this study is pioneering in demonstrating the efficacy of an SBR equipped with a conical reactor incorporating CCBC.

The results of this study provide practical outcomes for decentralized or resource-constrained wastewater treatment systems. Specifically, the integration of a conical SBR design with CCBC has been demonstrated as a viable approach to enhance effluent quality, stabilize reactor performance, and optimize

nutrient removal without relying on complex infrastructure.

Consequently, future research endeavors should incorporate these objectives and conduct thorough evaluations of specific wastewater. For instance, the SBR system established in this research as a means to remove pharmaceuticals containing antimicrobial agents from hospital wastewater is of significant value as a wastewater treatment solution for small, underdeveloped hospitals in developing countries.

Credit author statement

The corresponding author hereby confirms that the given information is correct as per his knowledge. Even though at various stages of work contribution roles were often interchanged, the table below presents the primary contribution of individual author. Conceptualization: Heritier A. Mufwankolo and Tomoaki Itayama; Original draft writing: Heritier A. Mufwankolo; literature search: Heritier A. Mufwankolo, Rosine Huania, Buy Dui TAN, Nguyen Binh MINH; original draft writing: Heritier A. Mufwankolo; Data curation, Formal analysis, Investigation, Visualization: Heritier A. Mufwankolo and Tomoaki Itayama; Review & editing: Heritier A. Mufwankolo, Rosine Huania, Itayama Tomoaki, Hideaki Sano; Resources, Validation and guidance; Hideaki Sano, Itayama Tomoaki.

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