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Fluidized-Bed Bioreactor Applications for Biological Wastewater Treatment: A Review of Research and Developments

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ABSTRACT

Wastewater treatment is a process that is vital to protecting both the environment and human health. At present, the most cost-effective way of treating wastewater is with biological treatment processes such as the activated sludge process, despite their long operating times. However, population increases have created a demand for more efficient means of wastewater treatment. Fluidization has been demonstrated to increase the efficiency of many processes in chemical and biochemical engineering, but it has not been widely used in large-scale wastewater treatment. At the University of Western Ontario, the circulating fluidized-bed bioreactor (CFBBR) was developed for treating wastewater. In this process, carrier particles develop a biofilm composed of bacteria and other microbes. The excellent mixing and mass transfer characteristics inherent to fluidization make this process very effective at treating both municipal and industrial wastewater. Studies of lab- and pilot-scale systems showed that the CFBBR can remove over 90% of the influent organic matter and 80% of the nitrogen, and produces less than one-third as much biological sludge as the activated sludge process. Due to its high efficiency, the CFBBR can also be used to treat wastewaters with high organic solid concentrations, which are more difficult to treat with conventional methods because they require longer residence times; the CFBBR can also be used to reduce the system size and footprint. In addition, it is much better at handling and recovering from dynamic loadings (i.e., varying influent volume and concentrations) than current systems. Overall, the CFBBR has been shown to be a very effective means of treating wastewater, and to be capable of treating larger volumes of wastewater using a smaller reactor volume and a shorter residence time. In addition, its compact design holds potential for more geographically localized and isolated wastewater treatment systems.

1. Introduction

1.1. Biological wastewater treatment

Wastewater treatment is an important process for protecting both the environment and human health. Pollutants and bacteria in wastewater can cause severe damage to water resources, resulting in further damage to humans and other animals that come in contact with tainted water. For centuries, humans were able to simply release their waste into the environment with little or no effect, as the environment was able to take up the pollutants. With the growth of the human population, however, this is no longer possible; nature's capacity for taking up pollutants has long been exceeded, and wastewater must be treated before release, or risk damaging both humans and the environment. Today, wastewater is collected from buildings (residential, industrial, business, medical, etc.) and enters sewer pipe systems. It then flows through pipes and pumping stations (which keep the flow moving) until it reaches a treatment plant. Many established processes are capable of treating wastewater effectively. However, as the human population continues to

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increase, more wastewater will be produced, creating a greater demand for treatment [1].

To meet the ever-increasing demand for wastewater treatment, new plants will need to be constructed and existing plants will require upgrades and expansions. These new plants and expansions will take up more space in population centers; however, as cities expand, less space will be available for treatment plants. To combat this problem, more efficient treatment processes will be needed that are capable of treating larger volumes of wastewater in less time than conventional methods. One technology that has been shown to have high efficiency in wastewater treatment is the circulating fluidized-bed bioreactor (CFBBR) [2], which was developed at the University of Western Ontario, Canada. This review covers the research that was done at the university on lab- and pilot-scale CFBBR systems used for the treatment of municipal wastewater (MWW) and of various industrial wastewaters.

The primary pollutants that must be removed from wastewater are carbon, nitrogen, and phosphorus, including organic compounds, ammonia, phosphates, and many other pollutants. Particulate and colloidal solids must also be removed. Finally, harmful pathogens need to be stabilized and/or destroyed [1].

The conventional layout of a wastewater treatment plant (WWTP) starts with primary treatment, which removes large solids through a physical separation process such as screening and gravity settling. This is followed by secondary treatment, where most of the biological treatment occurs. Finally, the wastewater moves to tertiary treatment, where it is chemically polished and disinfected (if necessary). Fig. 1 shows the basic layout of a WWTP that uses an activated sludge system.

Biological treatment processes are employed in the secondary treatment section. Biological treatment is carried out by microbial growth contained within bioreactors; the microbes consume the pollutants through their metabolic processes. Biological treatment generally comes in two forms: suspended and attached growth. In suspended growth, the microbial colonies (flocs) freely swim/float in the mixed liquor. Mixing is mechanically induced, either by impellers or by air flow from the bottom. The most well-known suspended growth process is the activated sludge process (Fig. 1). Attached growth, also called fixed film, is characterized by a biofilm that is composed of bacteria, particulates, extracellular polymers, and gels growing on a support media (Fig. 2). The typical carrier media used for attached growth are rock or plastic. Ideal carriers are porous and have rough surfaces, allowing for more effective attachment than smooth, non-porous surfaces [1].

The four main processes that are carried out in general wastewater treatment are aerobic organic oxidation, nitrification, denitrification, and biological phosphorus removal. Through these processes, most of the carbon, nitrogen, and phosphorus are removed. These processes are carried out by different types of bacteria and require different environmental conditions and substrates [1]. The two main classes of bacteria involved are categorized based on the type of carbon they consume for cell growth: Heterotrophic bacteria consume organic carbon, whereas autotrophic bacteria consume inorganic carbon. The three main environmental conditions are aerobic (presence of oxygen), anoxic (presence of nitrates, little to no oxygen), and anaerobic (no oxygen or nitrates) conditions [1].

1.1.1. Aerobic organic oxidation

Heterotrophic bacteria oxidize organic material to gain energy and use it for biomass synthesis. The basic reaction is as follows:

$$C_aH_bO_cN_d\ (\text{organic material}) + O_2 \rightarrow CO_2 + H_2O + NH_3 \tag{1}$$

As seen in reaction (1), the organic material (e.g., $C_5H_7O_3N$) is broken down into carbon dioxide, water, and ammonia, using oxygen gas as the oxidizing agent [1].

1.1.2. Nitrification

Autotrophic bacteria carry out a carbon-fixation process using ammonia as the electron donor to convert inorganic carbon into organic carbon compounds. This reduction-oxidation reaction oxidizes the ammonia to form nitrites and then further oxidizes the nitrites to form nitrates. The same reaction is used by bacteria to gain energy for other cellular functions. Due to the lower growth yields and rates of autotrophic bacteria, most of the biodegradable organics in the liquor must be removed first. Otherwise, heterotrophic bacteria will dominate the growth and outcompete the nitrifying bacteria, leading to washout of the nitrifiers.

$$NH_4^+ + HCO_3^- + O_2 + CO_2 \rightarrow NO_3^- + H_2O + C_5H_7O_2N\ (\text{biomass}) \tag{2}$$

$$NH_4^+ + O_2 \rightarrow NO_3^- + H^+ + H_2O \tag{3}$$

Reaction (2) shows ammonia being used as an electron donor to reduce inorganic carbon (HCO_3^- and CO_2) into organic carbon. Some of the ammonia is also incorporated into the new biomass. Reaction (3) shows the overall oxidation of ammonia to form nitrates [1].

1.1.3. Denitrification

Certain bacteria have a nitrate reductase enzyme in their electron transport chain that allows them to substitute nitrates for oxygen as the electron acceptor. Through this process, nitrates are reduced in a series of reactions to diatomic nitrogen, which then bubbles out of the water due to its low solubility. It should be noted that this process can only occur in low-oxygen and high-nitrate concentrations (anoxic conditions); otherwise, the nitrate reductase enzyme will be inhibited [1].

$$NO_3^- + C_aH_bO_cN_d \rightarrow CO_2 + H_2O + NH_3 + N_2 + OH^- \tag{4}$$

Reaction (4) is similar to reaction (1) in that organic material is

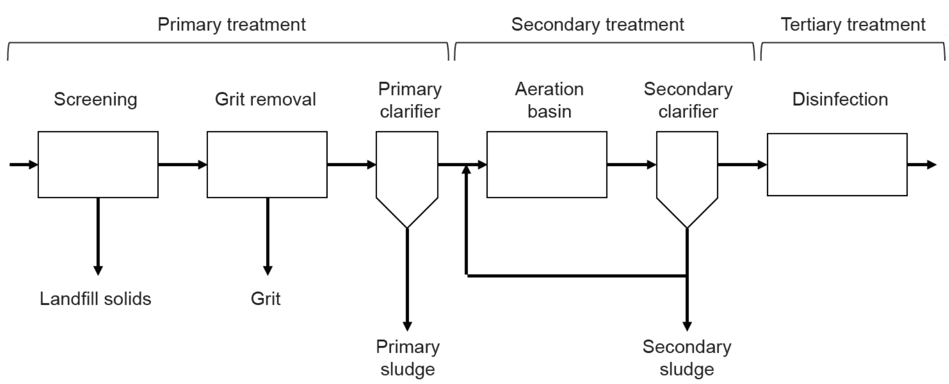

Fig. 1. Layout of a conventional WWTP.

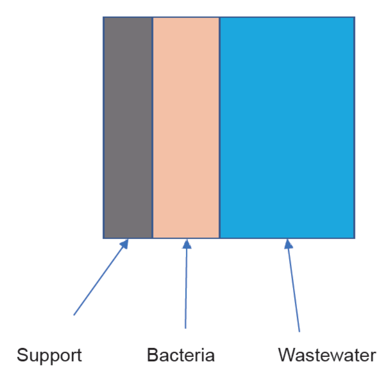

Fig. 2. Attached-growth biological treatment.

oxidized into carbon dioxide, water, and ammonia. However, in this case, nitrates have replaced oxygen as the electron acceptor. When the nitrates are reduced, they become diatomic nitrogen (N_2) and leave the system as nitrogen gas [1].

1.1.4. Biological phosphorus removal

In addition to phosphorus removal by biomass synthesis, enhanced biological phosphorus removal (EBPR) occurs when large concentrations of phosphorus are present. EBPR is a two-stage process that is carried out by a group of bacteria called polyphosphate accumulating organisms (PAOs). These microbes are capable of storing large amounts of phosphates in the form of polyphosphate granules. This is a method of energy storage that is used to replace adenosine triphosphate when the aerobic metabolic pathways are not functional (i.e., in the absence of oxygen).

The first stage is an anaerobic process, in which the PAOs use their stored phosphate to take up and store organic material (acetate and short fatty acids), while simultaneously releasing phosphates into the water. The second stage is an aerobic process, in which the PAOs use the stored fatty acids as an energy source for taking up the phosphates in the water, and store the phosphates as polyphosphate granules. The microbes are then settled in the clarifier and recycled to the start of the process, and any excess sludge is removed. In EBPR, phosphates are ultimately removed in the waste sludge stored in the PAOs [1].

1.2. Fluidized-bed bioreactors

Fluidization is a process in which the upward flow of a fluid suspends a bed of particles. Fluidization offers many advantages, including excellent mixing, increased mass transfer, large specific surface area, and uniform particle and temperature distributions. Fluidization was first used in the 1920s for coal gasification [3]. Its second major application was fluidized catalytic cracking, which was developed in the 1940s [4]. Both processes utilized gas-solid fluidization, which has since been developed and applied to many other processes. Liquid-solid (LS) and gas-liquid-solid (GLS) three-phase fluidization were developed later, and have been proven to have great potential and application in biochemical processes [5]. The basic outline and function of these two forms of fluidization are covered in the next two subsections.

1.2.1. Fluidization

LS fluidization works by means of an upward-moving liquid stream that suspends and/or entrains a bed of solid particles. Although several fluidization regimes exist, the two regimes used in wastewater processes are the conventional and circulating regimes. Fig. 3 and Fig. 4 [5] provide basic depictions of conventional and circulating fluidization systems, respectively. In conventional fluidization, the liquid velocity is insufficient to entrain the particles and wash them out of the column [3]. In circulating fluidization, a high liquid velocity is used to carry the particles to the top of the column and then return them to the bottom via a recycle line or column [6].

GLS three-phase fluidization has the same general layout as LS fluidization, except with the addition of an air distributor as well as the liquid distributor. In GLS fluidization, both the liquid stream and gas bubbles fluidize the particles. Like LS fluidization, GLS fluidization can operate in both conventional and circulating regimes [7]. However, depending on the specific requirements of the process, only one of the columns may have a gas distributor; thus, only one of the columns may operate with GLS fluidization. The process discussed in this paper follows this setup.

1.2.2. Principle of the fluidized-bed bioreactor

The fluidized-bed bioreactor (FBBR) is an application of the LS fluidized bed. These bioreactors can be run in a single or double column system, depending on the treatment process being carried out. The FBBR is an attached-growth process. The microbes attach to the fluidized media and form a biofilm on the surface (Fig. 5) [1]. Fluidization in the column is caused by the recirculating wastewater and/or by the air stream, if the process includes aeration [5].

Like all fluidization processes, the excellent mixing, increased mass transfer, and enlarged surface area in the FBBR process enhance its function. The use of smaller particles than those in other attached-growth systems such as integrated fixed-film activated sludge (IFAS) and moving-bed bioreactor (MBBR) systems, coupled with excellent microbial attachment characteristics, results in much thicker biofilms; hence, the surface area of the film exposed to the water is much higher than in traditional attached-growth processes. The increased contact between the wastewater substrates and the biofilm also allows this process to break down larger compounds that are typically more difficult to treat. In addition, the FBBR has proved to be capable of handling larger loadings and operating at lower hydraulic retention times than a typical bioreactor.

2. Circulating fluidized-bed bioreactor

The CFBBR system developed at the University of Western Ontario

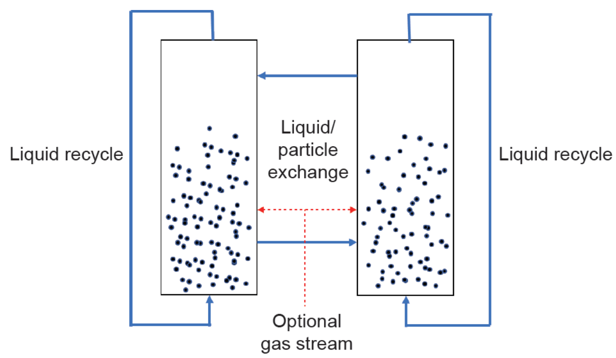

Fig. 3. Conventional twin fluidized-bed system. Adapted from Ref. [5].

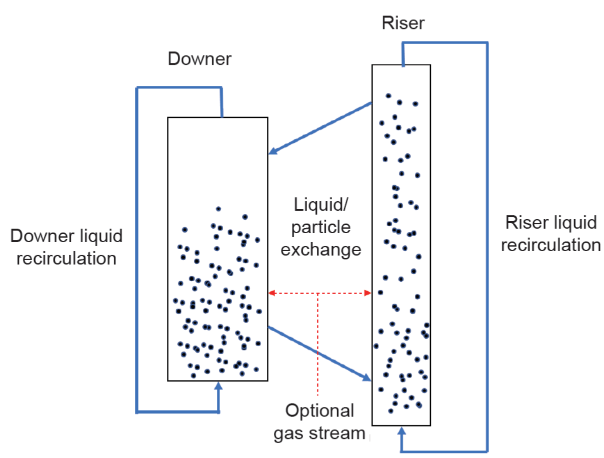

Fig. 4. Layout of a liquid fluidized bed with particle circulation. Adapted from Ref. [5].

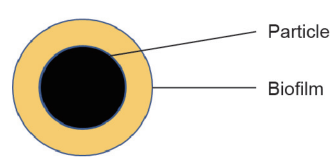

Fig. 5. Particle biofilm.

is a twin-column system that is capable of maintaining two different environments, which is advantageous for biological treatment [5]. The CFBBR has an aerobic column (medium to high oxygen) and an anoxic column (low oxygen, high nitrates), enabling it to achieve nitrification and denitrification in the same process. It can also be run with particle exchange between the two columns in order to enhance phosphorus removal due to transfer between aerobic and anaerobic environments [8]. The CFBBR has been tested in lab- and pilot-scale reactors to treat MWW and leachate.

2.1. Scales of research studies

2.1.1. Lab scale

CFBBR-1

The CFBBR was first tested with a lab-scale reactor (CFBBR-1) consisting of a riser, downer, and liquid-solid separator at the top of each column. A schematic of the system is shown in Fig. 6 [9]. The configuration of CFFBR-1 is similar to the circulating fluidization system shown in Fig. 4, which operates with particle circulation between the riser and downer. The system operates with the riser in the circulating fluidization regime and the downer in the conventional regime, with the liquid and particles being separated at the top in the LS separators. The particles at the top of the riser are transferred to the downer. Because the particles are more tightly packed in the downer than in the riser, the biofilm-rich particles transferred to the downer from the riser will lose their biofilm due to shear and abrasion as the particles collide with each other. As a result, the loss of biomass increases the density of the particles, enhancing their downward flow through the conventional fluidized bed. The particles in the bottom of the downer are recirculated to the riser bottom to begin the cycle again. The liquid at the top of the downer enters an LS separator, where most of the suspended solids (volatile suspended solids (VSS) and total suspended solids (TSS)) are separated for sludge wasting. The remaining nitrate-rich liquid is circulated back to the downer for fluidization and to the riser for fluidization and denitrification. In this apparatus, lava rock was used as the carrier media. The average particle diameter was 0.67 mm with a bulk and true density of 1720 kg·m^{-3} and 2560 kg·m^{-3}, respectively. The lava rock had an approximate specific surface area of 9000 m^2·m^{-3} [9].

The system is designed so that the downer operates under aerobic conditions (three-phase fluidization) to achieve the biological organic oxidation and nitrification of ammonia. The riser operates under anoxic conditions (two-phase fluidization) to achieve the denitrification of nitrates. The system has also demonstrated potential for EBPR, albeit not to the same degree as a system that is specifically designed for EBPR. This is because the CFBBR lacks a true anaerobic zone, which is required for EBPR [1]. Table 1 [9] provides a summary of the influent and effluent qualities of the CFBBR.

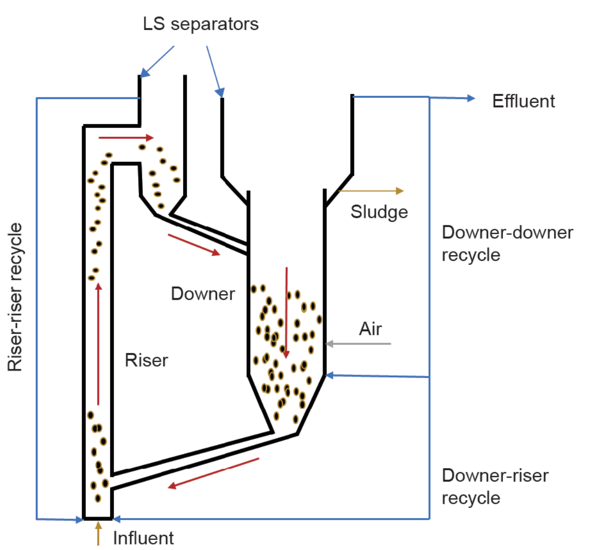

Fig. 6. Diagram of the CFBBR design showing the directions of the gas, liquid, and solid flow [9].

CFBBR-2

The other system tested at the lab scale is the twin fluidized-bed bioreactor (CFBBR-2). Fig. 7 [10] provides a diagram of this system. Like the CFBBR-1, the CFBBR-2 consists of two columns: one aerobic and one anoxic. However, these columns are the same height, and both operate with conventional fluidization, similar to the configuration shown in Fig. 3. Since both columns operate in the conventional fluidization regime, no continuous particle exchange occurs. The CFBBR-2 was designed after the discovery that particle circulation does not play a significant factor in the CFBBR treatment performance. Circulation of the particles is only necessary if enhanced phosphorus removal is required. When necessary, particle circulation between the riser and downer can be carried out by using impellers at the top and bottom of the columns to periodically transfer the particles, making particle circulation independent within the process. Particles at the bottom of the aerobic column would thus be transferred to the anoxic column and those at the top of the anoxic column would be transferred to the aerobic column [10]. This particular system has two columns of identical shape and volume; however, the column sizes can vary depending on the required hydraulic retention time (HRT) for each column.

Because this system operates with conventional fluidization in both

Table 1
Influent and effluent qualities of the CFBBR (unit: mg·L^{-1}) [9].

Parameter	Influent	Effluent
COD	273	26
SCOD	73	21
NH_4^+-N	19	0.7
NO_3^--N	0.5	6.5
TN	31.2	8.6
TP	3.8	0.8
TSS	144	4
VSS	118	3

COD: chemical oxygen demand; SCOD: soluble chemical oxygen demand; TN: total nitrogen; TP: total phosphorus.

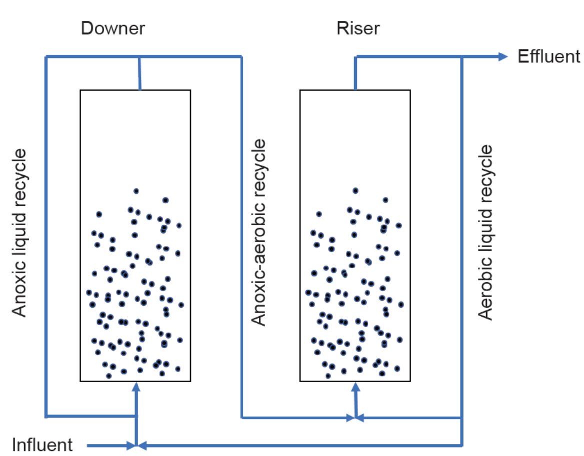

Fig. 7. Diagram of the CFBBR-2 system [10].

columns, the shear rate on the biofilm is lower than in the CFBBR-1. This led to a much lower detachment rate and longer solids retention time (SRT), culminating in a much lower observed biomass yield for the overall system. The observed solids yield ranged from 0.06–0.071 g(VSS)·g(COD)$^{-1}$ (COD: chemical oxygen demand) [10], which was significantly lower than the yields seen in CFBBR-1 [0.12–0.16 g(VSS)·g(COD)$^{-1}$] [9]. The CFBBR-2 system was shown to have similar biological nutrient removal (BNR) performance and effluent quality to that of the CFBBR-1. Table 2 [10] summarizes the influent and effluent quality of the CFBBR-2.

The CFBBR-2 system was tested at various organic loading rates (OLRs) with synthetic loading of 1.3, 1.7, and 2.3 kg(COD)·(m^3·d)$^{-1}$. The effluent quality and BNR efficiency were similar for all of these loadings. Above an OLR of 2.3 kg(COD)·(m^3·d)$^{-1}$, the COD removal efficiency began to decrease due to increased shear on the particles and subsequent biomass detachment. However, in the lower OLRs, the detachment rate was measured to be much lower than that of the CFBBR-1 at both the lab and pilot scale, giving it a comparatively longer SRT [10].

2.1.2. Pilot scale

Following success at the lab scale, a pilot-scale system was established and tested at the Adelaide Wastewater Treatment Plant in London, Canada. As one of the City of London's six wastewater treatment plants, Adelaide treats an annual average of 27 500 m^3·d^{-1} [11]. The system had the same general configuration, layout, and operation as the lab-scale system. Fig. 8 shows the design of the pilot-scale system [12].

The carrier media for this system was lava rock, with a similar particle diameter (an average of 0.67 mm) and bulk density (1720 kg·m^{-3}) as those used in the lab-scale system. The pilot-scale CFBBR was designed to treat 5 m^3·d^{-1} of primary influent and achieved removal efficiencies that were close to those of the lab-scale system [12]. An exceptional aspect of this system was demonstrated by its effluent quality: The VSS and phosphorus concentrations were low enough to meet the required secondary effluent quality without the need for secondary clarification or chemical phosphorus removal. This system's ability to handle high-solid feeds and produce low-solid effluent could allow future WWTPs to reduce the size and cost of their clarifiers [13]. Table 3 [12] provides the treatment data from all three phases of the study. The influent flow rates for Phases I–III were 2880, 4320, and 5800 L·d^{-1} on average, respectively. Table 4 [9,10,12,14,15] provides a full summary of the lab- and pilot-scale BNR efficiencies along with those of alternative technologies and methods.

Typical activated sludge processes operate at aerobic HRTs of 4–24 h [1], so the fact that the CFBBR achieves comparable nutrient removal efficiencies at considerably lower HRTs clearly demonstrates this system's effectiveness for BNR.

2.2. Response to dynamic loading conditions

One crucial aspect of a wastewater system is its ability to handle dynamic loadings while still treating wastewater effectively and maintaining sufficient BNR. There are two common forms of dynamic loading. The first is a sudden increased flow with similar nutrient loading as before, resulting in a larger volume of diluted wastewater.

Table 2
Influent and effluent quality of the CFBBR-2 system (unit: mg·L^{-1}) [10].

Parameter	Influent	Effluent
COD	262	20
SCOD	234	9.5
NH_4^+-N	26.1	0.5
NO_3^--N	0.7	3.9
TN	29.5	5.4
TP	4.4	3.8
TSS	27	16.3
VSS	19	12

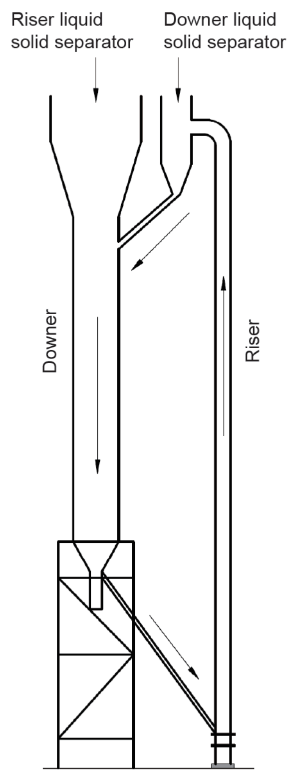

Fig. 8. Configuration of the pilot-scale CFBBR [12].

Table 3
Influent and effluent data of the pilot-scale CFBBR study (unit: mg·L^{-1}) [12].

Parameter	Phase I (2880 L·d^{-1})		Phase II (4320 L·d^{-1})		Phase III (5800 L·d^{-1})	
	Influent	Effluent	Influent	Effluent	Influent	Effluent
TCOD	332 ± 42	26 ± 3	349 ± 38	39 ± 8	496 ± 152	45 ± 7
SCOD	71 ± 14	13 ± 4	100 ± 16	15 ± 4	117 ± 23	23 ± 5
NH_4^+-N	22.1 ± 5.2	1.2 ± 0.5	24.6 ± 2.9	0.9 ± 0.3	25.8 ± 1.1	9.5 ± 0.9
NO_3^{dfff-}-N	0.9 ± 0.6	3.6 ± 1.2	0.4 ± 0.1	4.7 ± 1.3	0.4 ± 0.1	2.8 ± 0.6
TP	4.9 ± 1	1 ± 0.1	4.2 ± 0.8	1.2 ± 0.2	5.9 ± 0.6	1.2 ± 0.4
TSS	217 ± 27	11 ± 2	219 ± 26	22 ± 6	443 ± 174	27 ± 6
VSS	174 ± 28	9 ± 2	171 ± 23	16 ± 5	315 ± 106	21 ± 6

TCOD: total chemical oxygen demand.

An example of this would be wet weather flows [16]. The other form is organic shock loading, in which there is a sharp increase in the organics and/or solids concentration in water, while the volume remains unchanged [1]. Both forms of dynamic loading were tested in the pilot-scale system at the Adelaide Wastewater Treatment Plant.

2.2.1. Wet weather flows

Wet weather flows are a challenge for any plant in an area with frequent rain and snow. The increased volume of wastewater flowing through the same units results in a reduced residence time, thereby lowering the removal efficiency of the system. This causes the effluent to have higher than usual concentrations of pollutants. It is also possible in extreme cases for water to be sent through a bypass, forgoing any treatment at all. Both scenarios can be damaging to the environment unless handled properly [1].

Wet weather flows were simulated in the pilot-scale CFBBR at the Adelaide Wastewater Treatment Plant. Clean tap water was added to the influent to increase the volumetric loading, thereby simulating wet weather flows. The baseline flow rate started at 5 $m^3 \cdot d^{-1}$ of de-gritted MWW. The clean water was then added to increase the total flow to 10 $m^3 \cdot d^{-1}$ and again to 20 $m^3 \cdot d^{-1}$. Each of these increased flows was maintained for 4 h, the average time for an increased wet weather flow [17]. Considering that the system was designed for a 5 $m^3 \cdot d^{-1}$ flow rate, the flows of 10 $m^3 \cdot d^{-1}$ and 20 $m^3 \cdot d^{-1}$ correspond to peak flow factors of 2 and 4, respectively. A peak flow factor of 4 is a common design parameter when accounting for wet weather flows in the design of a wastewater treatment system [17].

As shown in Section 2.4.2, the steady-state COD, total nitrogen (TN), and total phosphorus (TP) removal efficiencies in the pilot were approximately 90%, 80%, and 70%, respectively. During the dynamic testing, there was a measurable decrease in effluent quality and organics and nutrient removal, but the levels remained within acceptable limits. This result was somewhat expected, since a decrease in efficiency is the main effect of hydraulic overloading [16]. The steady-state and dynamic effluent quality and BNR efficiencies are summarized in Table 5 and Table 6 [17].

The organic and nutrient removal efficiency did drop during the simulated wet weather flows. At twice the typical flow rate (10 $m^3 \cdot d^{-1}$), the removal efficiencies and effluent quality were within acceptable parameters [18]. However, at four times the flow rate (20 $m^3 \cdot d^{-1}$), the removal efficiencies effluent quality became too poor and no longer met acceptable standards. This result indicates that the maximum allowable wet weather flow is somewhere between 10 $m^3 \cdot d^{-1}$ and 20 $m^3 \cdot d^{-1}$; that is, perhaps three times the baseline flow rate (15 $m^3 \cdot d^{-1}$). The CFBBR could continue operating without the need for secondary clarification or chemical addition for phosphorus removal.

When the CFBBR's response to dynamic loading was compared with those of other fixed-film processes, the CFBBR was found to have similar effluent quality and removal efficiency. Table 7 [17,19–21] compares the treatment efficiency of several different processes with the CFBBR's performance.

Aside from the immediate response of a system to dynamic loading, the other important factor is the system recovery—that is, how quickly the system returns to its steady-state effluent quality and removal efficiency. Past studies on conventional processes show that they can take anywhere from 7 d to 15 d to recover from sustained peaking factors of 2.5 for 2 h [1]. The study on the CFBBR's recovery from hydraulic overloading showed that the system almost fully recovered within 24 h of the end of the peak flow. Table 8 [17] shows the changes in the attached biofilm, nitrification rate, and denitrification rate that were measured before, during, and 24 h after hydraulic overloading.

2.2.2. Organic shock loading

Sharp increases in organic concentrations disrupt the biological processes occurring in the system. A large increase in biodegradable organic pollutants without a corresponding increase in available oxygen will result in the domination of non-nitrifying heterotrophs over the nitrifying autotrophs, due to the higher biomass yields and faster utilization rates of aerobic heterotrophs. This occurrence leads to washout (loss) of nitrifiers and to an overall decrease of nitrification efficiency. A large loss of nitrifiers can be difficult to recover from, due to their slow growth rates. Ultimately, there will be an increase in effluent COD and ammonia because of the drop in nutrient removal efficiency [1].

During a lab-scale study using the CFBBR-2 to treat synthetic

Table 4
Summary of BNR performance.

Name	Source	HRT (h)	EBCT (h)	SRT (d)	OLR [$kg \cdot (m^3 \cdot d)^{-1}$]	COD (%)	N (%)	P (%)	Biomass yields [$mg(VSS) \cdot mg(COD)^{-1}$]
CFBBR-1	[9]	2.04	0.82	44–56	3.36	91	78	85	0.12–0.135
CFBBR-2	[10]	2.88	0.98	72–108	2.23	97	84	12	0.071
Pilot CFBBR	[12]	2.03	1.5	20–39	4.12	90	80	70	0.12–0.16
UASB	[14]	3.2	—	—	2.6	34	—	—	—
AnMBR	[15]	7.92	—	—	5.9–19.8	58	—	—	—

HRT = $V_{Reactor}/Q$; EBCT = $V_{Compacted\ bed}/Q$ (empty bed contact time); UASB: upflow anaerobic sludge blanket; AnMBR: anaerobic membrane bioreactor.

Table 5
Summary of steady-state and dynamic loading effluent quality in the pilot-scale CFBBR (unit: $mg \cdot L^{-1}$) [17].

Parameter	5 $m^3 \cdot d^{-1}$		10 $m^3 \cdot d^{-1}$		20 $m^3 \cdot d^{-1}$	
	Influent	Effluent	Influent[a]	Effluent	Influent[a]	Effluent
TCOD	578	41	289	64.2	144.5	63
SCOD	192	20	96	24.5	48	22
NH_4^+-N	35.2	0.9	17.6	2	9.8	3.4
NO_3^--N	< 0.06	5.4	< 0.03	5.7	< 0.2	6.9
PO_4^--P	—	< 1	—	0.5	—	0.4
TP	12.5	1.3	6.3	1.8	3.2	2.7
TSS	443	32	221.5	—	111	38
VSS	339	22	169.5	—	85	—

[a] Estimated from 5 $m^3 \cdot d^{-1}$ influent data.

wastewater, the system's response to organic shock loading was tested by increasing the influent COD in a stepwise fashion. The influent COD concentration started at 420 mg·L^{-1} and was then increased to 720 mg·L^{-1} for 4.5 h, followed by a further increase to 1200 mg·L^{-1} for 4 h. 1200 mg·L^{-1} corresponded to an ultimate OLR of 13.2 kg(COD)·(m^3·d)$^{-1}$. Liquid circulation and aeration rates remained unchanged during the shock loadings [22].

During testing, as expected, the nitrification efficiencies dropped from 95% to 49% due to heterotrophs dominating growth and to the limitations of dissolved oxygen (DO). The DO was measured in the riser and downer as 0 mg·L^{-1} and 2.5 mg·L^{-1}, respectively, at its lowest. During steady-state operation, the DO was measured as 0.3 mg·L^{-1} and 4.9 mg·L^{-1} in the riser and downer, respectively. The COD removal also dropped, although not as much as the nitrification efficiency, from 93.4% to 64.1%. Decreases in COD removal and in nitrification efficiency were seen in the effluent when both COD removal and nitrification efficiency sharply increased at the same time, 1.8 h into the test.

Batch-specific nitrification rate tests verified the decreased nitrification efficiency, which showed a 15% decrease in activity after the 10 h carbon shock load. This result indicates a 15% washout of the nitrifying biomass during the shock load. Despite the changes in biomass activity, the total amount of attached biomass measured in the system did not materially change during the shock load.

2.3. Water reuse

In addition to meeting typical secondary effluent quality standards without the need for additional treatment (clarification or chemical addition), the CFBBR was shown to generate an effluent that might be usable for non-potable reuse applications such as agricultural irrigation or industrial uses. In order for treated wastewater to be reclaimed, it must have reasonable disinfection characteristics—meaning that it must be easily disinfected without requiring large amounts of chemicals or energy. The two main requirements that must be met for reasonable disinfection are biochemical oxygen demand and TSS concentrations of less than 30 mg·L^{-1} [18], because meeting these requirements makes ultraviolet disinfection reasonable [9]. The CFBBR is capable of meeting or closely approaching these requirements during its steady-state operation. With some additional treatment, such as clarification or chemical addition, the effluent from dynamic loadings could also meet this standard.

2.4. Additional design considerations, issues, and challenges

2.4.1. Worm predation

The SRT in a wastewater treatment system can have a considerable impact on the solids yield. Longer SRTs typically lead to lower yields, since the biomass decays to a greater extent [23]. The solids yield can also be affected by the presence of larger organisms in the system that are capable of consuming the microbes, such as protozoans, metazoans, and oligochaete worms. These predators are aided by longer SRTs because a longer SRT gives them more time to consume the microbes [24,25]. In the past two decades, developments have been made in using worms for solids yield reduction in wastewater systems. In most cases, a separate worm reactor is employed between the activated sludge basin and the secondary clarifier [26].

The effect of worm predation in the CFBBR was studied at the lab scale. In this system, the worms were active in the downer, consuming the biofilm as the particles moved down the column. When the particles moved back to the riser, the biofilm would regrow until the particles returned to the downer to continue the cycle [27].

Overall, it was found that simultaneous COD and nitrogen removal could be achieved with worm predation integrated into the system. The BNR efficiencies were consistent with those of past studies and the system showed greatly reduced solids yields due to the worm predation. The study revealed an observed solids yield of 0.082 g(VSS)·g(COD)$^{-1}$.

2.4.2. Effects of carbon-to-nitrogen ratio on BNR efficiency

To study the effects of the carbon-to-nitrogen ratio, a lab-scale study was conducted to examine how varying COD loading with constant nitrogen loading would affect the simultaneous COD and nitrogen removal. Since a high-COD concentration reduces nitrifier activity, fewer nitrates will be produced, subsequently enhancing

Table 6
Summary of dynamic loading BNR efficiency in the pilot-scale CFBBR [17].

BNR efficiency	5 m^3·d^{-1}	10 m^3·d^{-1}	20 m^3·d^{-1}
COD removal (%)	90	75	49
N removal (%)	80	39	23
P removal (%)	70	43	16

Table 7
Comparison of dynamic loading effluent and nutrient removal percentages.

Process	Source	HRT (h)	Influent (^{1}COD, ^{2}NH$_4$, ^{3}TSS, ^{4}TN) (mg·L^{-1})	Effluent (^{1}COD, ^{2}NH$_4$, ^{3}TSS, ^{4}TN) (mg·L^{-1})	Removal (^{1}COD, ^{2}TN, ^{3}TP)
Submerged fixed-film	[19]	3.2	^{1}450, ^{3}120,^{4}80	^{1}65, ^{2}11, ^{3}19	^{1}90%, ^{2}80%
		0.7	—	^{1}110, ^{2}55, ^{3}30	^{1}75%, ^{2}20%
Moving bed	[20]	1.4	^{1}527, ^{2}18.5	^{1}121, ^{2}11, ^{3}53	^{1}75%
		0.4	—	^{1}230, ^{2}18, ^{3}104	^{1}56%
Biological aerated filter	[21]	2.0	^{1}235	^{1}57, ^{3}19	^{1}85%
		0.8	—	^{1}138, ^{3}41	^{1}35%
CFBBR	[17]	3.2	^{1}578, ^{3}443, ^{4}61	^{1}47, ^{2}1, ^{3}31	^{1}90%, ^{2}80%, ^{3}70%
		0.8	—	^{1}65, ^{2}4.7, ^{3}50	^{1}49%, ^{2}23%, ^{3}16%

Table 8
Biomass characteristics during the dynamic loading study [17].

Parameter	Before overload	During overload	24 h after overload
Anoxic biofilm [mg(VSS)·g(particles)$^{-1}$]	16.7	15.4	15.6
Aerobic biofilm [mg(VSS)·g(particles)$^{-1}$]	6.9	6.2	6.3
Nitrification {g(NH$_4$)·[g(VSS)·d^{-1}]$^{-1}$}	0.12	0.08	0.1
Denitrification {g(NO$_3$)·[g(VSS)·d^{-1}]$^{-1}$}	0.34	0.28	0.31

the denitrifier activity [1]. COD/nitrogen (COD/N) ratios of 10∶1, 6∶1, and 4∶1 were tested at the same empty bed contact time (EBCT) (0.82 h). The total COD removal did not vary much between the three phases, achieving above 90% removal throughout. However, the amount of COD oxidation occurring in the riser compared to the downer changed between the phases. At a COD/N ratio of 10∶1, approximately 37% of the COD was oxidized in the riser. This relatively low value was due to that phase having the lowest amount of nitrates produced from nitrification. A COD/N ratio of 4∶1 resulted in approximately 57% of the COD being oxidized in the riser. This higher value was due to a higher amount of nitrates being produced during nitrification. It was also shown that as the ratio became smaller, the amount of nitrogen removal decreased, with ratios of 10∶1, 6∶1, and 4∶1 achieving TN removals of ~91%, ~82%, and ~71%, respectively. The first two phases both reached acceptable effluent quality, while the third phase did not, and would need additional treatment to reach an acceptable effluent quality [18]. All three phases showed low solid yields, a finding that was consistent with the other CFBBR studies; the yields of the three phases ranged from 0.11–0.15 mg(VSS)·mg(COD)$^{-1}$ [28].

3. High-strength wastewater treatment

In addition to MWW, the CFBBR technology has been applied to the treatment of landfill leachate and to rendering waste. An aerobic platform called the anaerobic fluidized-bed bioreactor (AnFBR) was applied to the treatment of wastewater sludges (primary and secondary) and of thin stillage from bioethanol.

3.1. Application of CFBBR in its basic form

3.1.1. Landfill leachate

Landfill leachate forms when organic waste in landfills is broken down by the bacteria that are present and mixes with water, producing a high-concentration solution of soluble COD, ammonia, and other pollutants. Because of its toxicity, treating landfill leachate effectively is of high importance. The high concentrations of COD, ammonia, and heavy metals, along with many other pollutants, that are present in landfill leachate can seriously damage the environment if they are not properly treated and removed. In addition, the low carbon-to-nitrogen ratio of landfill leachate makes biological treatment challenging. As discharge limits become increasingly stringent, conventional biological treatment paired with physical and chemical treatment methods may no longer be effective enough for the treatment of landfill leachate [29].

In addition to being tested as a means of treating MWW, the pilot-scale CFBBR located at the Adelaide Wastewater Treatment Plant was tested as a means of treating landfill leachate. Its integration of aerobic and anoxic conditions into a single process made it a suitable candidate to achieve higher required standards of treatment. The physical operation of the CFBBR system was the same as the operation when treating MWW. The anoxic riser operated in the fast fluidization regime and the aerobic downer operated in the conventional regime. For leachate treatment, the CFBBR was not run with particle circulation [30].

The pilot-scale CFBBR was tested at various loadings and corresponding HRTs with leachate taken from the W12A landfill in London, Canada. Table 9 [30] shows the three flow rates that were used and their corresponding loading values. Table 10 [30] shows the average influent and effluent quality from each stage.

The CFBBR showed very low solid yields. For Phases I–III, the yields were 0.13, 0.15, and 0.16 g(VSS)·g(COD)$^{-1}$, respectively; these are similar to the yields that were obtained when treating MWW in the CFBBR. In the second phase, at an OLR of 2.15, the CFBBR achieved COD, nitrogen, and phosphorus removal efficiencies of ~85%, ~80%, and ~70%, respectively. These removal efficiencies are similar to those that were obtained when treating MWW in the CFBBR. However, the actual effluent concentrations from the treated leachate were higher, given the higher influent concentrations. Table 11 [30–34] compares the COD removal efficiencies of other treatment methods with those of the CFBBR.

3.1.2. Rendering waste

The CFBBR was also used to treat another high-strength wastewater: rendering wastewater. Rendering comes from the livestock-farming and food-processing industry; it is produced when organic wastes are mixed together to form wastewater with high organic and nutrient concentrations. Like all high-strength wastewater, rendering wastewater must meet certain effluent quality standards before it can be discharged into municipal sewers [35]. For this study, a lab-scale reactor of the CFBBR-1 configuration was built at a rendering facility in Hamilton, Canada, using lava rock as the carrier media (0.67 mm in diameter, 1720 kg·m^{-3} in bulk density). The study was carried out in three phases with varying influent flows and OLRs [36]. Table 12 [36] provides a summary of the operating parameters of the reactor.

The CFBBR showed excellent performance in treating the rendering waste. In Phase I, which had the highest OLR in the study, the COD removal efficiency was above 90% and the nitrogen removal efficiency was 79%. The solids yields in this test were similar to those in the other studies using the CFBBR, with an average yield of 0.12 g(VSS)·g(COD)$^{-1}$. Table 13 [36] provides the influent and effluent

Table 9
CFBBR operating conditions for leachate treatment [30].

Parameter	Column	Phase I	Phase II	Phase III
Influent (L·d^{-1})	–	650	720	864
Avg. OLR [kg(COD)·(m^3·d)$^{-1}$]	–	1.9	2.15	2.6
EBCT (d)	Aerobic	0.43	0.38	0.32
	Anoxic	0.12	0.11	0.09
HRT (d)	Aerobic	0.89	0.81	0.67
	Anoxic	0.27	0.25	0.21
SRT (d)	Aerobic	26	21	18
	Anoxic	18	17	13

Table 10
Influent and effluent quality of leachate (unit: mg·L^{-1}) [30].

Parameter	Influent	Effluent		
		Phase I	Phase II	Phase III
TCOD	1259	195	197	302
SCOD	1025	149	153	245
TSS	263	56	60	58
VSS	156	38	37	44
NH_4^+-N	360	34.6	35.4	54.7
NO_3^--N	3.1	57.5	59.9	63.9
TP	6.2	1	1	1.2

Table 11
Comparison of leachate treatment methods.

Reactor type	Influent COD (mg·L^{-1})	HRT (h)	COD removal (%)	Source
CFBBR	1259	8	85	[30]
Trickling filter	800–1350	4.5	52	[31]
UASB	1120–3520	24	77	[32]
MBBR	1740–4850	36	60	[33]
FBBR	1100–3800	34	82	[34]

parameters of the reactor.

Although the CFBBR had very high COD and nitrogen removal efficiencies, it was unable to meet sewer discharge requirements because the high influent COD concentration resulted in an effluent COD concentration above 1000 mg·L^{-1} in all phases of the study (whereas typical sewer discharge is 300 mg·L^{-1} [18]). Many of the other parameters were also above their allowable limits for discharge. However, the high removal efficiencies and low solids yields showed the CFBBR's potential for treating rendering. Increasing the residence time of the rendering in the reactor could improve the treatment performance. Also, using a multistage treatment process or chemical polishing could improve the treatment performance and enable the CFBBR to meet discharge standards.

3.2. Anaerobic fluidized-bed platform

The CFBBR-2 has also been tested as an anaerobic platform for the treatment of high-strength and high-solids waste streams such as municipal sludges and corn ethanol thin stillage. A schematic of the AnFBR is shown in Fig. 9 [37]. Like the CFBBR, the anaerobic platform utilizes a biofilm attached to a carrier media to treat the wastewater; however, the microbes in this process are anaerobic. Because this process only requires an anaerobic environment (i.e., it does not require aerobic or anoxic environments), a single column was used and was operated in the conventional fluidization regime [37]. Due to the environmental requirements of the anaerobic microbes, the system had to be maintained at 37 °C and at a pH of 6.8–7.4 for ideal operation [1].

3.2.1. Municipal wastewater sludge

Municipal sludge is a byproduct of the wastewater treatment process. Primary sludge (PS) is generated from the primary clarifiers following the screening and de-gritting of the wastewater, and is mostly composed of organic material. Activated sludge is settled in the secondary clarifier and later thickened to become thickened waste activated sludge (TWAS). TWAS is mostly active biomass, being composed of the bacteria and other microbes present in activated sludge. TWAS typically takes longer to treat because it is largely composed of active biomass [38].

The AnFBR was tested for the digestion of both PS and TWAS. The digestion of each sludge was tested separately at flow rates ranging from 1.8–16 L·d^{-1}, corresponding to HRTs ranging from 8.9–1.0 d. The average influent for TSS was 38 989 mg·L^{-1} for PS and 34 834 mg·L^{-1} for TWAS, while the average total chemical oxygen demand (TCOD) was 37 488 mg·L^{-1} and 34 414 mg·L^{-1}, respectively [37]. Table 14 and Table 15 [37] summarize the results from treating PS and TWAS, respectively.

As expected, the treatment of TWAS was less extensive than the treatment of PS because TWAS is largely composed of active biomass and is thus more difficult to digest, while PS is predominantly composed of inactive organic material. However, the AnFBR was able to effectively treat both PS and TWAS at considerably shorter HRTs than conventional methods while maintaining comparably high SRTs due to the large amount of biomass attachment [37].

The AnFBR achieved much higher VSS and COD removal efficiencies at much shorter HRTs, compared with conventional methods. In addition, it was able to achieve these efficiencies while operating at OLRs that were 5–10 times higher than those of conventional anaerobic digesters. Table 16 [37,39–41] compares the treatment results of the AnFBR with the results from several examples of conventional digestion methods.

3.2.2. Thin stillage

One of the feed stocks used to produce ethanol as a biofuel is corn. The corn is mashed and fermented, producing ethanol. The leftover mash and liquid of unfermented corn is a high-strength waste called stillage, which must be treated before discharge [42]. Although stillage can be repurposed as a food source for livestock,

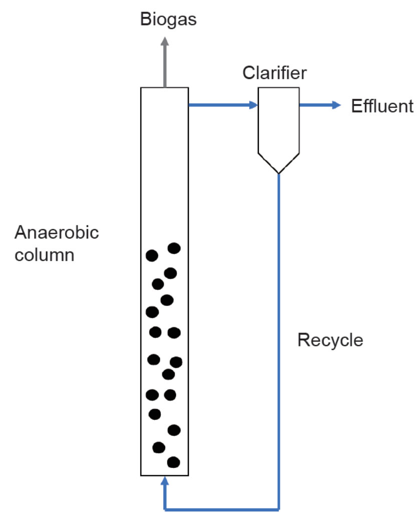

Fig. 9. Diagram of the AnFBR system [37].

Table 12
Summary of rendering treatment operational parameters [36].

Parameter	Column	Phase I	Phase II	Phase III
Influent flow (L·d^{-1})	—	2 ± 0.1	1.5 ± 0.05	1 ± 0.05
OLR [kg(COD)·(m^3·d)$^{-1}$]	—	14.6	11	7.3
HRT (h)	Anoxic	9.36	12.24	18.48
	Aerobic	39.6	52.8	79.2
EBCT (h)	Anoxic	5.52	7.36	11.04
	Aerobic	14.16	18.88	28.32
SRT (d)	Anoxic	2	4.8	20
	Aerobic	3.2	7.1	33

Table 13
Influent and effluent parameters of rendering treatment (unit: mg·L^{-1}) [36].

Parameter	Influent	Effluent		
		Phase I	Phase II	Phase III
TCOD	29 509 ± 678	3 151 ± 586	2 263 ± 220	1 305 ± 85
SCOD	28 527 ± 283	1 466 ± 465	1 039 ± 118	853 ± 32
NH_4^+-N	605.3 ± 6.2	121.8 ± 23.1	94.4 ± 9.6	0.9 ± 0.4
NO_3^--N	3.8 ± 4.4	8.9 ± 2.9	5.5 ± 1.3	3.1 ± 0.7
TP	44.8 ± 5.4	34.6 ± 8.1	27.1 ± 3.3	9.8 ± 2.1
TSS	973 ± 215	2 000 ± 611	1 282 ± 159	460.8 ± 48.2
VSS	676 ±160	1 379 ± 369	908 ± 89	329.9 ± 51.8

the drying process often makes this uneconomical, due to high energy requirements. Instead, anaerobic digestion is a suitable treatment method that can convert stillage into biogas, which can be recovered for energy production via combustion, while simultaneously removing a large portion of the organics present [43].

The treatability of thin stillage using the AnFBR was explored. The AnFBR was fed thin stillage at an OLR of approximately 29 kg(COD)·(m^3·d)$^{-1}$, an anaerobic HRT of 3.5 d, and a solid loading rate of approximately 10–10.8 kg(TSS)·(m^3·d)$^{-1}$. Despite the short retention time, the AnFBR achieved a TCOD removal efficiency of about 88% [37]. Table 17 [37] provides a summary of the major influent and effluent parameters.

When compared with other anaerobic digestion technologies for the treatment of thin stillage, the performance of the AnFBR in treating thin stillage was similar to its performances when treating PS and TWAS. The AnFBR achieved comparable VSS and TCOD removal efficiencies at lower HRTs than conventional methods, thus demonstrating its great capability for treating high-COD and high-solids waste products. Table 18 [37,42,44] provides a comparison of the treatment efficiency of the AnFBR with those of other methods.

4. Modeling

Several models have been developed for the CFBBR using the modeling programs AQUIFAS and BioWin. AQUIFAS combines activated sludge and fixed-film kinetics into a single model. This model utilizes semi-empirical equations and a two-dimensional biofilm model [45–47]. BioWin models the biofilm processes as one-dimensional fully dynamic and steady-state models. AQUIFAS was used for modeling MWW treatment, while BioWin was used for modelling leachate treatment. The models were used to predict the treatment performance of the CFBBR in treating MWW at the lab and pilot scale, and leachate at the pilot scale.

Table 14
Summary of PS treatment [37].

Parameter	Phase I	Phase II	Phase III	Phase IV	Phase V
HRT (d)	8.9	4	1.9	1	1.5
SRT (d)	17.2	6.9	2.9	1.1	1.7
VSS_{Eff} (mg·L^{-1})	3 693	6 326	9 364	21 320	18 069
VSS removal (%)	88	79	70	31	42
COD removal (%)	85	79	68	30	42

Table 15
Summary of TWAS treatment [37].

Parameter	Phase I	Phase II	Phase III	Phase IV
HRT (d)	8.8	4	1.9	2.6
SRT (d)	16.7	7.2	2.7	2.8
VSS_{Eff} (mg·L^{-1})	9 390	13 300	20 400	17 800
VSS removal (%)	69	56	33	42
COD removal (%)	68	55	34	42

Table 16
Comparison of AnFBR treatment capability with those of conventional methods.

Reactor type	Sludge type	OLR [kg(COD)·(m^3·d)$^{-1}$]	COD removal (%)	HRT (d)	Source
AnFBR	PS	4.2	85	8.9	[37]
CSTR	PS	2.1–2.9	33–47	10–15	[39]
AnFBR	TWAS	4.2	68	8.8	[37]
CSTR	TWAS	1	24	20–40	[40]
AnMBR	TWAS	2.4–2.6	48	7–15	[41]

CSTR: continuous stirred-tank reactor.

4.1. Modeling municipal wastewater treatment using AQUIFAS

AQUIFAS unifies activated sludge and fixed-film processes to simulate particulate biofilm operations. It uses semi-empirical equations, incorporating Monod kinetics and the mass transfer kinetics of a biofilm, in order to simulate BNR. By changing the input parameters for the different loadings, the model calculates the theoretical effluent parameters and estimates the biofilm thickness on the particles. The AQUIFAS model has previously been used to successfully model the IFAS and MBBR processes, indicating its potential for modeling the FBBR process [48].

AQUIFAS was used to estimate the effluent parameters based on the pilot-scale data. The error from simulated to actual results varied between 0% and 60%. Most simulated results were close to the actual results, with deviations of 0%–30%, particularly the results for the COD, nitrogen, and phosphorus. However, the suspended solids results differed from the actual results by anywhere from 20% to 67%, although the results remained within the standard deviation [49]. Table 19 [49] provides the results of the simulation compared with the actual pilot study results.

AQUIFAS was also used to model the CFBBR-2 process. This iteration of the model incorporated a predictive fluidization model, including both two and three phases. The fluidization was used to link the dynamics of the fluidized bed to the BNR efficiency more accurately. The model was based on the media type and size, flow rate, and cross-sectional area. It was then used to calculate parameters such as bed expansion, phase hold up, and specific surface area [50].

The simulated effluent data obtained was compared with experimental values from the CFBBR-2 study, and was confirmed with a two-sided *t*-test to be within a 95% confidence interval. The updated model that incorporated fluidization was a significant improvement over the previous AQUIFAS model. Table 20 [50] presents the comparative results.

4.2. Modeling leachate treatment using BioWin

Leachate treatment using the CFBBR was modeled with BioWin. BioWin models the CFBBR systems as one-dimensional, fully dynamic, and steady-state models. It uses data on loading, biomass concentration, and biofilm thickness against experimentally obtained data from large-scale treatment plants. It also incorporates data on the amount of non-biodegradable and non-colloidal solids present (which are easily or readily measurable) [51]. Since the landfill leachate had a high soluble fraction of COD, the influent specifications needed to be altered from those of typical wastewater [52]. Table 21

Table 17
Summary of thin stillage treatment (unit: mg·L^{-1}) [37].

Parameter	Influent	Effluent
TSS	46 400	9 800
VSS	46 200	9 200
TCOD	129 300	14 400
SCOD	62 000	2 700

Table 18
Comparison of AnFBR treatment of thin stillage with those of conventional methods.

Reactor type	OLR [kg(COD)·(m^3·d)$^{-1}$]	HRT (d)	COD removal (%)	Source
AnFBR	28–30	3.5	88	[37]
CSTR	1.6–3.9	24–40	85–86	[42]
ASBR	9.5	10	90	[44]

ASBR: anaerobic sequencing batch reactor.

[52] shows the results of the simulation compared with the actual data from the leachate study.

5. Discussion

5.1. The CFBBR

The CFBBR demonstrated its exceptional ability for treating MWW. It achieved COD removal efficiencies above 90% and the removal of nitrogen and phosphorus (80% and 70%, respectively) at very low HRTs. It was also able to handle higher solid loadings than conventional methods due to its enhanced contact between the substrates and the biofilm. Since the CFBBR was able to treat unclarified primary influent, it is possible for the influent to bypass primary clarification entirely, eliminating the need for primary clarifiers and thereby reducing capital costs. Overall, the CFBBR is capable of treating larger volumes of wastewater at lower retention times than its conventional counterparts.

The longer solid retention time of the CFBBR also leads to reduced solid/sludge yields. Low solids concentrations in the effluent stream could potentially eliminate the need for secondary clarifiers if the concentration meets discharge standards, which in some cases it did. Even in situations where the concentration is unable to meet discharge standards, the low solids concentrations of the CFBBR could at least reduce the size of the clarifiers needed and the amount of sludge produced, thus reducing the capital and operating costs for a plant. Lower overall sludge production from wastewater treatment would also reduce the required sludge treatment capacity. Less sludge to treat would lead to the possibility of using smaller digesters or incinerators for treatment.

5.2. High-strength wastewater treatment

The single-column anaerobic platform for the CFBBR had excellent results when treating high-strength wastewaters. Given the ability of the CFBBR to handle high-solids and COD loadings, it is well-suited for treating wastes such as municipal sludge and thin stillage. Conventional digesters for MWW sludge are often a large capital expenditure and require a large footprint. The significantly lower retention times of the AnFBR would allow the same volume of sludge to be treated in a much smaller reactor. This would reduce the cost and size of the digesters required to treat the sludge produced by the treatment plant. Coupled with the already lower solid/sludge yield of the CFBBR, the use of the AnFBR would significantly

Table 19
Simulated vs. actual data from the pilot study (unit: $mg\cdot L^{-1}$) [49].

Parameter	Phase I		Phase II		Phase III		Phase IV	
	Sim.	Exp.	Sim.	Exp.	Sim.	Exp.	Sim.	Exp.
TCOD	35	26 ± 3	37	39 ± 8	45	41 ± 14	49	45 ± 7
SCOD	13	13 ± 3	9	15 ± 4	17	20 ± 8	18	23 ± 5
NH_4^+	0.8	1.2 ± 0.5	1.1	0.9 ± 0.6	1.4	0.9 ± 0.6	2.4	3.9 ± 0.9
NO_3^-	5	3.6 ± 1.2	5.5	4.7 ± 1.3	7.1	5.4 ± 1.3	9.9	4.8 ± 0.6
TN	7.9	6.2 ± 1.1	9.7	7.6 ± 1.3	11.5	9.4 ± 1.1	15.7	11.5 ± 1.2
PO_4^-	0.42	0.7 ± 0.1	0.34	0.5 ± 0.1	0.6	0.7 ± 0.2	0.51	0.6 ± 0.2
TP	1.12	1 ± 0.1	1.1	1.2 ± 0.2	1.9	1.3 ± 0.4	1.39	1.2 ± 0.4
VSS	20	11 ± 2	25	22 ± 6	25	41 ± 20	25	27 ± 6
TSS	15	9 ± 2	19	16 ± 5	17	21 ± 8	19	21 ± 6

Table 20
Simulated vs. actual data (unit: $mg\cdot L^{-1}$) [50].

Parameter	Feed	Riser exp.	Riser sim.	Downer exp.	Downer sim.
TCOD	398 ± 52	101 ± 40	97.4	50 ± 21	59.6
SCOD	118 ± 24	31 ± 8	36.1	22 ± 5	19.8
NH_4^+	30 ± 4.5	4.1 ± 0.4	4	0.9 ± 0.4	0.72
NO_3^-	0.8 ± 0.3	3.2 ± 1.9	3.3	5.1 ± 1.6	5.8
TP	6.5 ± 1.4	—	—	3.2 ± 0.6	6
TSS	214 ± 41	62 ± 30	51.2	33 ± 14	54
VSS	183 ± 30	50 ± 27	43.8	24 ± 10	37

Table 21
Simulated vs. actual data of leachate treatment in the CFBBR (unit: $mg\cdot L^{-1}$) [52].

Parameter	Feed	Phase I		Phase II	
		Sim.	Exp.	Sim.	Exp.
TCOD	1259 ± 77	236	197 ± 46	235	302 ± 98
SCOD	1025 ± 27	169	153 ± 43	169	245 ± 85
NH_4^+	360 ± 59	33.7	35.4 ± 13.1	54.7	54.7 ± 11.2
NO_3^-	3.1 ± 1.5	61.1	59.9 ± 31.1	58.4	63.9 ± 10.3
TP	6.2 ± 1.3	1.5	1.7 ± 0.3	1.8	2.0 ± 0.6
TSS	263 ± 42	60	60 ± 13	58	58 ± 8
VSS	156 ± 30	45	37 ± 5	44	44 ± 8

cut down on the capital cost of treatment plants.

The AnFBR is also an excellent option for treating high-strength organic wastes from food industries such as dairy processing plants or breweries. The AnFBR can be employed to treat such waste streams and reduce the COD and solids concentrations in order to meet allowable sewer discharge standards.

5.3. Modeling

The results of modeling using both AQUIFAS and BioWin were fairly accurate, with some variation of accuracy between parameters. More work needs to be done to increase the accuracy of modeling the effluent solids concentration. However, the other parameters were estimated accurately. Both the AQUIFAS and BioWin models could serve as viable bases for developing future models for the CFBBR during scale-up work.

6. Future perspectives

The next stage in the development of the CFBBR is scaling up to a full-scale system that can be implemented at municipal treatment plants. Since fluidization is key to the process's enhanced treatment capabilities, maintaining fluidization at a large scale will be the main focus of the scale-up work. The other aspect of the scale-up will be devising methods of retaining the particles within the system, or of recycling the entrained particles back to the reactor. Scaling up can take two possible directions: developing larger fluidized beds based on the same configuration as the lab- and pilot-scale systems, or modifying existing systems to add a fluidization component to enhance treatment performance. Once the scale-up is complete, the implementation of the CFBBR system offers great potential for reducing the capital and operating costs of treatment plants. The CFBBR's compact design also presents an opportunity to establish wastewater treatment systems in a more geographically localized or isolated manner; for example, it could provide on-site treatment for remote resorts or small communities with little to no wastewater piping or treatment plants. Smaller systems could also be installed for individual buildings to avoid the need for wastewater collection and piping entirely. These small local systems would also be excellent for the immediate reclamation and reuse of wastewater, instead of discharging it to the environment, assuming that the wastewater meets reuse standards after treatment. These options show the great potential of the CFBBR for bringing a more effective means of wastewater treatment to places with established treatment systems in need of upgrades, and for introducing wastewater treatment into remote locations that currently lack any adequate treatment.

Compliance with ethics guidelines

Michael J. Nelson, George Nakhla, and Jesse Zhu declare that they have no conflict of interest or financial conflicts to disclose.

Nomenclature

AnFBR	Anaerobic fluidized-bed bioreactor
BNR	Biological nutrient removal
CFBBR	Circulating fluidized-bed bioreactor
COD	Chemical oxygen demand
DO	Dissolved oxygen
EBCT	Empty bed contact time
EBPR	Enhanced biological phosphorus removal
FBBR	Fluidized-bed bioreactor
HRT	Hydraulic retention time
MWW	Municipal wastewater
OLR	Organic loading rate
PAO	Polyphosphate accumulating organism
PS	Primary sludge
SCOD	Soluble chemical oxygen demand
SRT	Solids retention time
TCOD	Total chemical oxygen demand
TN	Total nitrogen
TP	Total phosphorus
TSS	Total suspended solids
TWAS	Thickened waste activated sludge
VSS	Volatile suspended solids

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