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Sustainable enhancement of biogas production from a cold-region municipal wastewater anaerobic digestion process using optimized sludge-derived and commercial biochar additives

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ABSTRACT

Anaerobic digestion (AD) of municipal wastewater sludges produces valuable solid digestate and biogas. Biogas is a source of clean energy and enhancement of its production has been of recent interest for increased electricity generation, among other products. The objective of this study was the development of a novel municipal sludge-derived biochar and its application in a municipal wastewater AD system to increase the biogas production rate. Thickened waste-activated sludge (TWAS) samples were collected from the cold-region municipal wastewater treatment plant and used to synthesize biochar applied in the simulation of AD processes using laboratory-scale reactors. The TWAS-derived biochar was synthesized using the commonly used furnace pyrolysis (sludge-based biochar, SBC), and more novel microwave pyrolysis including phosphoric acid as a microwave activator (activated sludge-based biochar, ASBC). The microwave pyrolysis conditions were optimized using a computational fluid dynamics (CFD) technique. In addition, various commercially available carbon-based additives were assessed for their impacts on the AD process including activated carbon, wood-derived biochar, and forest residue-derived biochar. Results showed that the ASBC increased the cumulative methane production by 50% (333 mL/g VS) versus the control sample (221 mL/g VS) after 30 d. The ASBC showed higher surface area, electrical conductivity, and metal contents versus the other biochars which boosted the AD microbial community growth leading to higher organic matter conversion into biogas. In the ASBC-amended digesters, the bacterial phylum *Bacteroidota*, which contains a major genus of the *dgA-11-gut-group*, exhibited a synergy between organic substrate fermentation and volatile fatty acid production, resulting in enhanced biogas production. The TWAS biochar demonstrated promising performance in enhancing the AD process fostering energy and resource self-sufficiency at municipal wastewater treatment plants. This smart sludge management aligns well with sustainable waste management practices and clean energy production strategies, especially considering that the biochar was sourced from a readily available continuous waste-product stream.

1. Introduction

Municipal wastewater treatment plants (MWTPs) use energy-intensive processes to remove pollutants from wastewater and produce effluents that are safe for receiving environments. A by-product of wastewater treatment processes are organic-rich sludges which are often treated using anaerobic digestion (AD) via hydrolysis, acidogenesis, acetogenesis, and methanogenesis processes (Appels et al., 2011). Hydrolysis breaks down complex-structured proteins, carbohydrates, and lipids into simpler compounds. Acidogenesis further degrades these compounds into volatile fatty acids (VFAs) that, via acetogenesis, are converted into acetate, carbon dioxide, and reduced electron carriers

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such as hydrogen and formate (HCO^-) (Harirchi et al., 2022). The final AD process is methanogenesis, including the interspecies electron transfer (IET) between H_2 and HCO^--producing fermentative bacteria and methanogenic archaea (Westerholm et al., 2022), which leads to the creation of digestate and biogas. Digestate is nutrient-rich and can be used as a fertilizer for sustainable agriculture, while biogas is valuable for use in heating, electricity generation, and transportation. It is important to note that the digestate may contain contaminants such as pharmaceuticals, per- and poly-fluoroalkyl substances (PFAS), heavy metals, and microplastics. Therefore, there is a need to carefully manage the AD feedstock selection and the digestate applications (Nizzetto et al., 2016; Golovko et al., 2022; O Connor et al., 2022).

Biogas is a renewable and clean source of energy composed primarily of methane (CH_4), and carbon dioxide (CO_2), along with trace amounts of hydrogen sulphide (H_2S), hydrogen, and nitrogen. The beneficial use of MWTP biogas can reduce fossil fuel use, greenhouse gas (GHG) emissions, and plant energy costs. Additionally, biogas can be a revenue stream for MWTPs to help compensate for sludge management costs estimated at 25–65% of the total MWTP operating costs (Andreoli et al., 2007; Vlyssides et al., 2004). Further, the appropriate utilization of MWTP biogas is consistent with several United Nations Sustainable Development Goals (SDG) including affordable and clean energy (SDG 7); industry, innovation, and infrastructure (SDG 9); and sustainable cities and communities (SDG 11). Thus, the enhancement of MWTP AD biogas production is of interest to a wide variety of Canadian stakeholders and policymakers given that Canada has been committed to the Paris Agreement in 2015 and net-zero emissions by 2050 in alignment with the SDGs.

Biogas production enhancement is challenging given that AD processes rely on microorganisms that are highly susceptible to environmental conditions including temperature. For example, the Saskatoon Wastewater Treatment Plant (SWTP) in Saskatoon, SK, experiences ambient seasonal temperatures ranging from −45 °C to 35 °C that affect both wastewater and sludge quality and quantity. These marked ambient temperature variations impact wastewater temperatures in open-to-air treatment bioreactors and clarifiers, with average temperatures reaching 17 °C in summer and dropping to 13 °C in winter (Asadi et al., 2021a). These variations can influence the sludge characteristics entering the SWTP AD system. Previous studies have shown that the microbial resilience of municipal sludges in the AD process can be improved, along with increased biogas production, through the addition of carbon-based additives such as biochar (Dang et al., 2016; Mumme et al., 2014).

Biochar has electron-accepting redox-active components and porous structures that facilitate nutrient retention, allow for improved AD buffering capacity and decreased VFA accumulation, and create an environment that promotes increased biogas production due to better microbial growth (Shanmugam et al., 2018; Cheng et al., 2018). For example, Zhou et al. (2020) reported that the AD of municipal sludge and raw corn stover-derived biochar increased CH_4 production by up to 26% because of increased buffering capacity and decreased ammonia inhibition that boosted microbial growth during the AD processes. Similarly, previous studies have shown that wood-derived biochar led to higher microbial growth during municipal sludge AD and improved process stability which increased CH_4 production between 48% and 92% (Shen et al., 2016; Wang et al., 2020a). Overall, the optimization of MWTP processes through the addition of biochar can lead to increased biogas production.

Biochar is obtained by pyrolysis of biomass from sources such as agricultural residues, forestry waste, animal manure, municipal solid wastes, and wastewater sludges. However, pyrolysis conditions are unique to each biomass which warrants the need for the optimization of pyrolysis conditions. Biochar production can take place in a conventional furnace or microwave pyrolysis process with the main operational parameters being furnace temperature (including heating rate), microwave power (surrogate for temperature), and residence times. For example, conventional furnace pyrolysis process temperatures between 400 and 700 °C (Leng et al., 2018; Zhao et al., 2013; Hossain et al., 2017) and reaction residence times of 1–5 h have been shown to produce effective biochars (Zornoza et al., 2016). While it is possible to use a thermometer inside a furnace to monitor temperatures, it is impossible to monitor microwave pyrolysis temperatures accurately due to wave absorption by the thermometer which disrupts the temperature measurement. In the absence of a thermometer, computational fluid dynamics (CFD) has been used to optimize microwave operating parameters during the pyrolysis process. For instance, Benis et al. (2022) used a CFD method to adjust the microwave time and power level for pyrolysis of an optimized canola-based biochar for use in adsorption. Typically, pyrolysis follows momentum conservation, energy balance, and electromagnetic wave equations that can be solved theoretically thereby limiting the need for time-intensive and costly experimental approaches. Recently, Kulkarni et al. (2023) reported that the reaction kinetics and electromagnetic wave interactions for microwave-assisted pyrolysis for biomass valorization can be optimized using a CFD method to increase process efficiency. Similarly, Motasemi and Gerber (Motasemi et al., 2018) used a CFD method to investigate the heating behavior of biomass materials during microwave pyrolysis to determine maximum temperature and temperature profiles during the process. Overall, CFD is a well-established mathematical methodology that has been widely used to solve these equations and simulate various complex phenomena including pyrolysis (Zeynali et al., 2023; Akbari et al., 2023; Jabbari et al., 2021).

Biochar production from municipal sludge requires energy for pretreatment (e.g., dewatering, drying, and acidic activation) and pyrolysis which could potentially offset the process's environmental impacts (Mayer et al., 2021; Gievers et al., 2021; Hosseinian et al., 2024). Therefore, there is a need to use life cycle assessment (LCA) to environmentally evaluate sludge-derived biochar production and its addition to an MWTP AD process. Previous studies have shown promising results for sludge drying and pyrolysis to produce biochar which can be a resource-efficient, cost-effective, and environment-friendly approach for sustainable biogas production as part of SDGs (Liu et al., 2022). For example, Huang et al. (2022) reported that biochar production using municipal sludge with 99% moisture content was environment-friendly with a global warming potential of −0.05 kg CO_2-eq/kg. Additionally, Norberto et al. (2023) produced phosphoric acid-activated biochar from canola straw using microwave-based pyrolysis (7 min at 1000 W). Their LCA results showed a global warming potential of −0.30 kg CO_2 eq/kg for activated biochar production making it an environment-friendly product.

Thus, this study investigated the enhancement of AD biogas production, both mathematically and experimentally, using sludge-based biochars including a novel comparison to commercially available carbon-based additives. A conventional furnace was used to create a sludge-derived biochar (SBC), while a CFD-optimized microwave pyrolysis process was used to produce an activated sludge-derived biochar (ASBC). The produced biochar physicochemical characteristics were compared to commercially available carbon-based additives including activated carbon (AC), wood-based biochar (WBC), and forest residue-based biochar (FBC). The sludge-based biochars and other additives were added to laboratory-scale AD reactors with biogas production monitored for 30 d. After 30 d, removals of volatile solids (VS), ammonia-nitrogen (NH_3-N), volatile fatty acids (VFA), and chemical oxygen demand (COD) were also determined. In addition, bacterial community analysis was conducted for all samples to determine microbial compositions.

The novelty of this study includes: (1) the development of an approach within a typical MWTP in which parts of the on-site sludge were processed to produce biochar, which was then used as another feedstock for the facility's AD. This smart sludge management optimizes wastewater sludge treatment, promotes energy and resource independence for MWTPs and reduces GHG emissions; (2) the microwave-based pyrolysis configuration was designed and optimized by CFD simulation; (3) a comprehensive set of physical, chemical, and biological tests were applied to analyze the AD system with sludge biochar and other additives. This research follows our previous studies in which data-driven models were used to estimate the SWTP AD biogas production using monitored quality and quantity parameters in real conditions without the consideration of additives and microbial analysis (Asadi et al., 2020, 2021b, 2024). The SWTP is a typical Canadian cold-region treatment plant while also being similar to municipal wastewater treatment plants worldwide; thus, results presented herein can be beneficial for informing biogas enhancement in a wide variety of locations.

2. Materials and methods

Fig. 1 shows a schematic flowchart of the study methodology to determine the biogas production rate from the AD of thickened waste-activated sludge (TWAS) amended with the synthesized (SBC and ASBC) and commercial (AC, WBC, FBC) carbon-based additives. The following section includes an overview of the SWTP (Section 2.1), carbon-based additive preparation (Section 2.2), AD experimental setup (Section 2.3), and microbial community analysis (Section 2.4).

2.1. Overview of the Saskatoon Wastewater Treatment Plant (SWTP)

The cold-region SWTP is an advanced municipal wastewater treatment facility located in Saskatoon, SK, Canada, that employs a multitude of treatment processes including a grit and screen facility, primary clarifiers, fermenters, a dissolved air flotation (DAF) thickener, a waste-activated sludge stripping to recover internal phosphate (WASSTRIP) unit, biological nutrient removal systems, an ultraviolet disinfection unit, a nutrient recovery facility, and anaerobic digesters. SWTP treats about 80–120 million liters per day (MLD) of municipal wastewater with effluents from the process being discharged directly into the South Saskatchewan River. The SWTP anaerobic digesters typically receive varying mixtures of primary clarifier sludge, fermenter sludge, and TWAS from the DAF unit (Fig. S1) and produce biogas composed of ~65% CH_4, ~35% CO_2, and an insignificant amount of H_2S (Gbangbo et al., 2023). Based on previous studies, the TWAS samples were used herein given its marked contribution to the SWTP biogas production as compared to the other sludge streams (Asadi et al., 2024).

2.2. Carbon-based additives

The synthesized SBC and ASBC were created using TWAS collected from the SWTP in February 2023 using furnace and microwave-based pyrolysis processes, respectively (Fig. 2). For both synthesized biochars, the TWAS sample was dried at 100 °C for 24 h, ground, and sieved using a 10-mesh to have uniform particle sizes before further processing. The commercial additives were AC from Sigma Aldrich (United States), FBC from Seneca Farms Biochar (United States), and WBC from TITAN (Saskatchewan, Canada). The synthesized and commercial carbon-based additives were stored in air-tight containers at room temperature (~22 °C) before use in experiments.

For SBC (Fig. 2a), TWAS powder was poured into a crucible covered by perforated tin foil and subjected to furnace pyrolysis. The furnace system (Lindberg, USA) was purged using N_2 gas (10 mL/min) to create an oxygen-free environment while its temperature increased at a rate of 5 °C/min to reach a temperature of 500 °C after ~ 90 min (Panwar et al., 2019). The furnace was then held at 500 °C for 1 h before being allowed to return to room temperature (~22 °C) after ~90 min prior to storage.

For ASBC (Fig. 2b), 20 g of TWAS powder was treated with 300 mL of 3.0 mol/L phosphoric acid (H_3PO_4; Sigma-Aldrich, Canada) which serves as an activator and microwave absorber allowing for the biomass pyrolysis in the microwave. This mixture was stirred for 1 h at room temperature and filtered using a 0.45 μm filter (Basix, China). The solids retained on the filter were placed in a quartz reaction vessel within a 2.45 GHz microwave (MRAI, USA) purged with N_2 to create an oxygen-free environment (Fig. 2b). The microwave pyrolysis duration and output power were determined using CFD COMSOL analysis. The defined microwave geometry is included in Fig. 3a, while the detailed description of the CFD modelling is included in the Supporting Information (SI). After the pyrolysis, the produced biochar was washed with deionized water three times, dried at 105 °C for 24 h, and allowed to return to room temperature prior to storage.

The synthesized and commercial carbon-based additives were characterized to determine pH (Model PHS-3C, Lei-Ci, Shanghai, China); electrical conductivity, EC (Model PHS-3C, Lei-Ci, Shanghai, China); metal composition (Inductively Coupled Plasma Mass Spectroscopy, Thermo ICP-MS model iCAP-RQ, Thermo Fisher Scientific, USA); non-metal composition (Series II CHNS/O Analyzer 2400, PerkinElmer); pore structure (by Field-Emission Scanning Electron Microscope (FE-SEM), Hitachi SU8010, Japan); specific surface (by the Brunauer-Emmett-Teller (BET) method, Quantachrome Instruments NOVA Touch lx4 Model, Austria), and biochar yield (Abd El Aziz et al., 2017).

To determine if the biomass feedstock is feasible to use for the production of biochar the biochar yield was calculated (Eq. (1)):

Biochar Yield (%) = (Mass of produced Biochar / Initial mass of feedstock) × 100 (1)

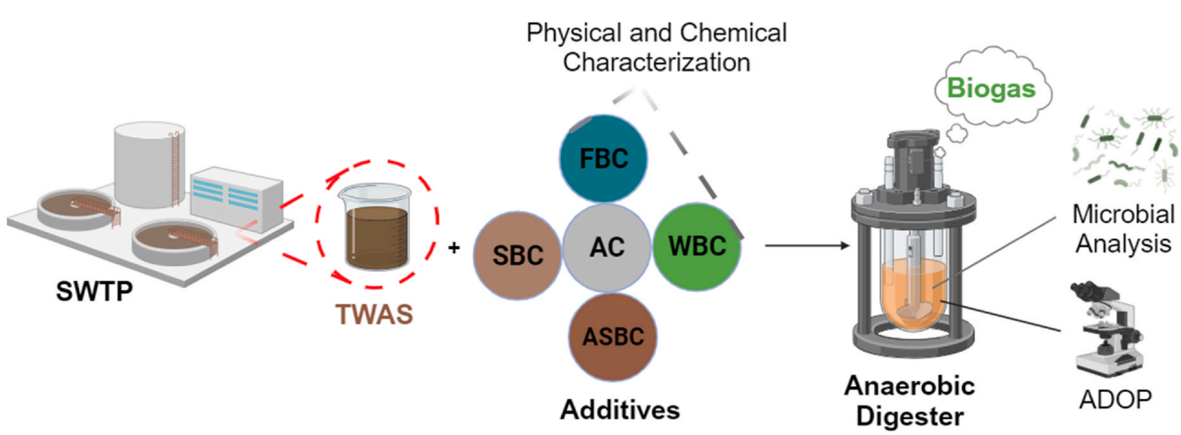

Fig. 1. Schematic flow chart for the determination of biogas production rate from anaerobic co-digestion of thickened waste-activated sludge (TWAS) and carbon-based additives including activated carbon (AC), wood-based biochar (WBC), forest residue-based biochar (FBC), sludge-based biochar (SBC) and activated sludge-based biochar (ASBC). Note: SWTP = Saskatoon Wastewater Treatment Plant, ADOP = Anaerobic Digestion Operational Parameters.

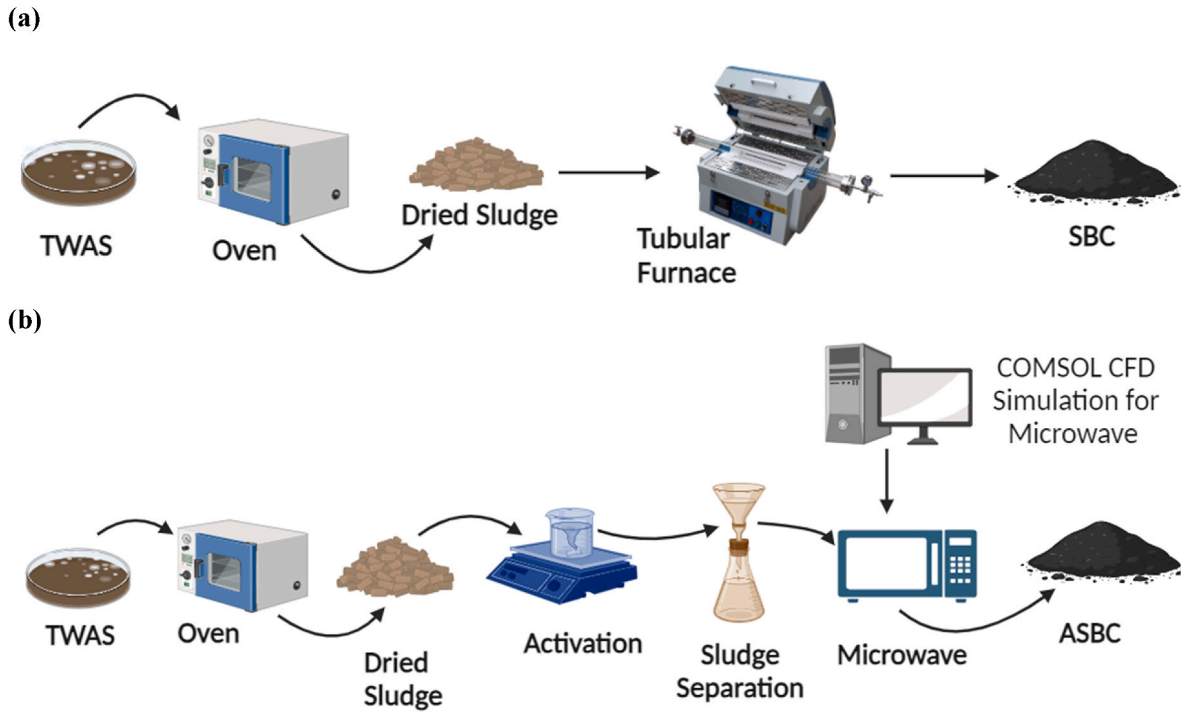

Fig. 2. Schematic flow chart for the synthesis of (a) Sludge-based biochar (SBC) using tubular furnace pyrolysis and (b) Activated sludge-based biochar (ASBC) using phosphoric acid activation and COMSOL-optimized microwave pyrolysis.

2.3. Experimental setup

The TWAS samples used for the AD process experiments were collected on two occasions in early 2023, placed in containers with ice, and transported to the Environmental Engineering laboratories at the University of Saskatchewan. Both samples were used directly in experiments without storage. The experimental setup included 500 mL glass batch reactors filled to a working volume of 400 mL of TWAS, an electrothermostatic water bath to maintain the temperature at 37 °C, and gas bags to collect produced biogas (Fig. S2). Each reactor was purged with N_2 before starting each experimental run and equipped with an automatic stirring system. The additive dosage was 10 g/L which is a typical concentration for biochar used in the literature.

Treatment reactors included: (1) TWAS with mixing and WBC, **TWBC**; (2) TWAS with mixing and FBC, **TFBC**; (3) TWAS with mixing and AC, **TAC**; (4) TWAS without mixing, **TWAS-N;** (5) TWAS with mixing (control sample), **TWAS-C** (6) TWAS with mixing and ASBC, **TASBC;** and (7) TWAS with mixing and SBC, **TSBC**. Experiments were 30 d in duration and were replicated three times. No external inoculum was included as it was deemed unnecessary based on previous experiments conducted by our research group. Before and after each experiment, the sludge was sampled to determine its physiochemical and microbial characteristics (see section 2.4 for detailed microbial analysis methodology). Physiochemical characteristics were determined using the Standard Methods for the Examination of Water and Wastewater (American Public Health Association (APHA) Water Environment Federation (WEF), 2005) (Eaton, 2005) and included chemical oxygen demand, COD (HACH DR/4000U Spectrophotometer, USA); VFA (HACH DR/4000U Spectrophotometer, USA); volatile solids, VS; EC and pH (Model PHS-3C, Lei-Ci, Shanghai, China). The as collected TWAS characterization analysis results included: VS = 24,000 ± 800 mg/L, ammonia nitrogen (NH_3-N) = 224 ± 20 mg/L, VFAs = 1653 ± 110 mg/L, C/N = 7.1 and COD = 35,200 ± 1500 mg/L. The reasonable standard deviation ranges indicate that the collected TWAS sample quality parameters were consistent over the sample collections.

The produced gas from each reactor was collected every three days using Tedlar Bags to determine the biogas production rate via water column displacement. Additionally, gas samples were taken by gas-tight syringes placed downstream of the reactor and transferred to pre-evacuated 12 mL Exetainer vials (LabCo Inc., High Wycombe, UK) before being analyzed by gas chromatography (Scion 456-GC, Bruker Daltonics Inc., USA) coupled with a thermal conductivity detector (TCD) for CO_2 concentration determination, and flame ionizer detector (FID) for CH_4 concentration determination.

The experimental CH_4 production kinetic parameters were calculated using a modified Gompertz model (Eq. (2)) to determine each treatment lag phase, the highest peak rate of CH_4 production, and the maximum potential CH_4 yield.

$$H = H_m \times \exp\left\{ -\exp\left[\frac{R_m \times e}{H_m}(\lambda - t) + 1\right]\right\} \tag{2}$$

Where, H = the cumulative CH_4 yield (mL/g VS) at a specific time, H_m = the maximum potential CH_4 yield (mL/g VS); R_m = the peak rate of CH_4 production (mL/g VS/day), λ = the lag phase duration (days), t = the time throughout the fermentation process (days), and $e \sim 2.718282$.

2.4. Microbial community analysis

Total DNA was isolated from sludges taken from each experiment including technical replicates of each treatment using the DNeasy PowerSoil Pro Kit (Qiagen, Germany) based on the manufacturer's standard protocol and inclusion of extraction blanks for quality control (QC) purposes. The polymerase chain reaction (PCR) amplification process was applied to normalized DNA samples, employing a uniquely dual-tagged primer set targeting the V3-V4 hypervariable regions of the 16S rRNA gene. The primers used were forward primer Prok341F (5′-CCTACGGGNBGCASCAG-3′) and reverse primer Prok805R (5′-GAC-TACNVGGGTATCTAATCC-3′) (Takahashi et al., 2014). The resulting PCR products were examined using agarose gel electrophoresis and subsequently purified using the QIAquick PCR Purification kit (Qiagen, Germany). The sequencing library was constructed, and next-generation sequencing was performed on the Illumina MiSeq platform using the

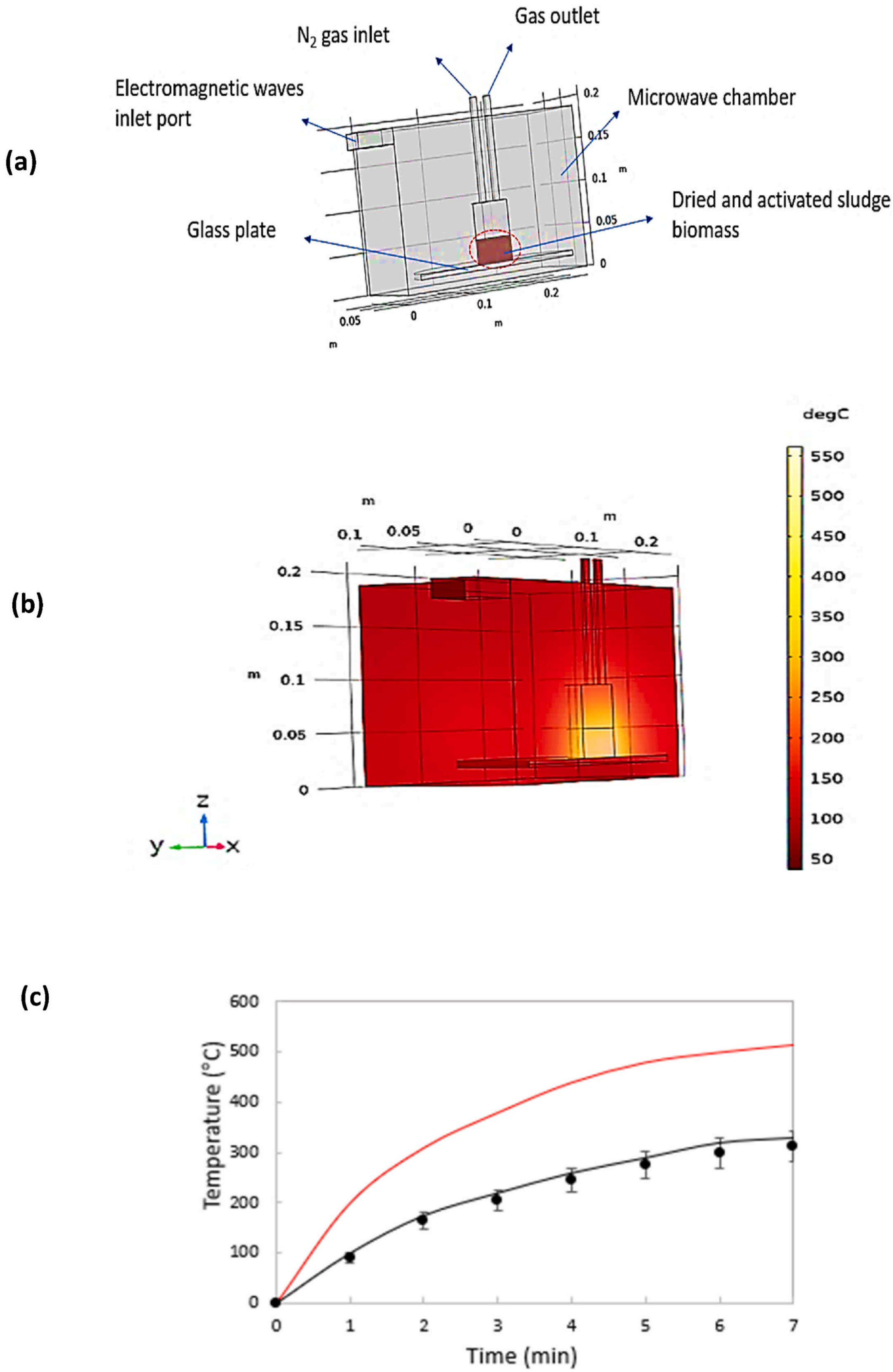

Fig. 3. (a) Scheme of microwave modeling's geometry to determine temperature distribution during pyrolysis for activated sludge-based biochar (ASBC) production; (b) Simulated temperature distribution within the microwave setup following 7 min of pyrolysis; (c) Predicted temperature patterns for the sample core (red solid line) and reactor exterior surface (black solid line), as compared to measured reactor exterior surface (black solid circles). Note: Error bars indicate standard deviations for n = 3.

600-cycle MiSeq Reagent Kit v3. This process included extraction and PCR blanks (Ankley et al., 2022).

For bioinformatics analysis, raw sequence reads were demultiplexed using fastq-multx (Version 1.3.1) according to dual tags of the forward and reverse primers for each sample. VSEARCH (Version 2.14.2) was used to merge the paired-end sequences, which involved the removal of both primers and the filtering of sequences based on quality (excluding those with expected error >1.0), chimera, and length (<400 bp) criteria (Rognes et al., 2016). Zero-radius operational taxonomic units (ZOTUs) were determined using unoise3 with a minimum frequency threshold of 5 (Edgar, 2016). Taxonomic classification was carried out using the q2-feature-classifier against the Silva database (Version 138) (Bolyen et al., 2019). Only ZOTUs classified as Bacteria or Archaea were retained, and those assigned to Mitochondria were excluded. A rarefaction depth of 51,723 per sample was implemented to minimize bias from uneven sequencing depths.

Statistical and graphical analyses were conducted using the R software environment (Version 4.0.3: RStudio Team, 2021). The ggplot2 package (Version 3.3.3) was employed for data visualization (Wickham, 2011). Choa1 diversity index for microbiome was determined using the package iNEXT (version 2.0.20) function ChaoRichness (Hsieh et al., 2016) and Shannon diversity index was determined using package vegan (version 2.6–4). For beta-diversity analysis, Principal Coordinate Analysis (PCoA) utilizing the weighted UniFrac distance metric was implemented to illustrate feature-level compositional distinctions between reactors with various conditions (Lozupone et al., 2011).

3. Results and discussion

This section is divided into four sub-sections including: (1) CFD microwave pyrolysis simulation for ASBC synthesis from thickened wastewater sludge (TWAS); (2) Physiochemical characterizations for TWAS and carbon-based additives; (3) AD of TWAS and carbon-based additives; and (4) Microbial community analysis.

3.1. CFD microwave pyrolysis simulation for ASBC synthesis from thickened wastewater sludge (TWAS)

CFD simulation results using the microwave geometry (Fig. 3a) indicated that the temperature distribution across the TWAS sample was non-uniform with a temperature gradient varying between 350 °C at the sample surface to 500 °C at the sample core for a 7 min pyrolysis time (Fig. 3b). This marked temperature variation indicates that microwave energy was directly absorbed by the TWAS (Wang et al., 2018; Wan et al., 2009). In contrast, for conventional furnace pyrolysis heat transfers from the sample's outer layer into its core by conduction and convection (Wang et al., 2018). The predicted sample core temperature increased over time reaching a maximum of 500 °C after 7 min (Fig. 3c) using 700 W input power. Additionally, the model indicated that the microwave pyrolysis reactor surface temperature increased to 312 °C after 7 min which was consistent with the temperature measurements collected using an optical thermometer outside of the microwave which supports the CFD model's performance.

It is important to note that temperature control during the pyrolysis process using a conventional furnace is not challenging because a thermocouple inside the furnace helps to monitor and achieve the targeted temperature. In contrast, temperature control during the pyrolysis process using the microwave method presents a challenge due to wave absorption by the thermometer. It is necessary to know the input power and time required to reach the actual temperature inside the biomass core. In this regard, a novel CFD study such as that included in this research can help solve this problem and pave the way for scaling up the pyrolysis process using microwaves on a larger scale.

3.2. Physicochemical characterizations for TWAS and carbon-based additives

The physicochemical characterization results for WBC, FBC, AC, ASBC, and SBC are presented in Table 1. The biochar yield represents the percentage of biomass converted into biochar during pyrolysis highlighting the efficiency of transforming feedstock into a carbon-rich material. Biochar yield values for the commercial-based additives (available from the manufacturer) varied between 43 and 48%, while measured ASBC and SBC samples created in this study were marginally lower at 42% and 41%, respectively (Table 1). Nansubuga et al. (2015) and Chen et al. (Wang et al., 2020b) found similar yields using various biomass types while showing the use of produced biochars for soil improvement, carbon sequestration, and wastewater treatment processes. Thus, the TWAS used in this study can be a suitable biomass for biochar production given the reasonable yield values.

BET surface area values were determined by average pore radius and pore volume (Table 1). The commercial biochars had a wide range of surface areas with AC being the highest at 1000 m^2/g, while also having the highest average pore volume (0.690 cm^3/g) and lowest average pore radius (0.015 Å). The ASBC had similar characteristics to AC including a surface area of 900 m^2/g, while the SBC surface area was the lowest recorded at 110 m^2/g. A higher BET surface area typically suggests a higher capacity for beneficial microbial growth (Wang et al., 2020a). Thus, the AC and ASBC samples would be anticipated to perform best in the experiments herein given microbial growth is an important parameter for biogas production.

The CHNS results indicated the highest carbon content for AC (71%), followed by FBC (65%), WBC (63%), SBC (35.6%), and ASBC (23.8%). The TWAS-derived biochar's lower carbon content indicates that they might be less stable in the environment if stored for long durations (Gurwick et al., 2013). The hydrogen content variation was between 0.95 and 2.3%, while the highest nitrogen content was observed in SBC (4.8%), followed by WBC (2.56%), ASBC (2.1%), AC (0.47%), and FBC (0.31%). Lastly, the AC sulfur content was markedly higher (0.97%) than the other biochars with the next highest being WBC at 0.30%. Given the lower carbon content of the synthesized biochars, these samples also had the lowest C/N ratios and highest H/C ratios with 11.3 and 0.080 for ASBC, and 7.41 and 0.050 for SBC, respectively (Table 1).

Overall, the CHNS analysis showed that SBC and ASBC with $H/C < 0.2$ were chemically stable with very low volatile contents (Zhou et al., 2020) which is associated with higher electron exchange capacity and redox properties of biochar which led to higher CH_4 production from AD (Devi et al., 2024). Additionally, its hydrogen and sulfur contents impact methane production and hydrogen sulphide emissions management, respectively (Chojnacka et al., 2015). The hydrogen helps CH_4 production by linking the degradation of complex organics to the production of

Table 1
Physicochemical characterization results for wood-based biochar (WBC), forest residue-based biochar (FBC), activated carbon (AC), activated sludge-based biochar (ASBC), and sludge-based biochar (SBC). DW: dry weight.

Physicochemical parameters	Commercial			Study	
	WBC	FBC	AC	ASBC	SBC
Biochar yield (% DW)	45	43	48	42	41
Average pore radius (Å)	0.019	0.021	0.015	0.016	0.024
Average pore volume (cm^3/g)	0.200	0.240	0.690	0.310	0.197
Surface area (m^2/g)	265	304	1000	900	110
C (% DW)	62.0	65.0	71.0	23.8	35.6
H (% DW)	1.00	2.30	0.95	1.89	1.82
N (% DW)	2.56	0.31	0.47	2.10	4.80
S (% DW)	0.30	0.07	0.97	0.09	0.29
C/N ratio	24.2	209	151	11.3	7.41
H/C ratio	0.016	0.035	0.013	0.080	0.050
EC (μS/cm)	150	232	198	500	410
pH	8.1	8.6	7.3	5.6	6.8

methane through syntrophic associations and specific metabolic pathways of methanogenic archaea (Chojnacka et al., 2015). The sulfur content in AD substrates directly impacts the production of hydrogen sulfide (H_2S) through biological reduction processes. Sulfate-reducing bacteria (SRB) in the digester convert sulfur-containing compounds, such as sulfates (SO_4^{2-}), present in the feedstock into H_2S as part of their metabolic activities (Mumme et al., 2014; Wang et al., 2019).

The commercial biochar EC values ranged from 150 to 232 μS/cm, while both synthesized biochars had markedly higher values with 500 μS/cm and 410 μS/cm for ASBC, and SBC, respectively (Table 1). For

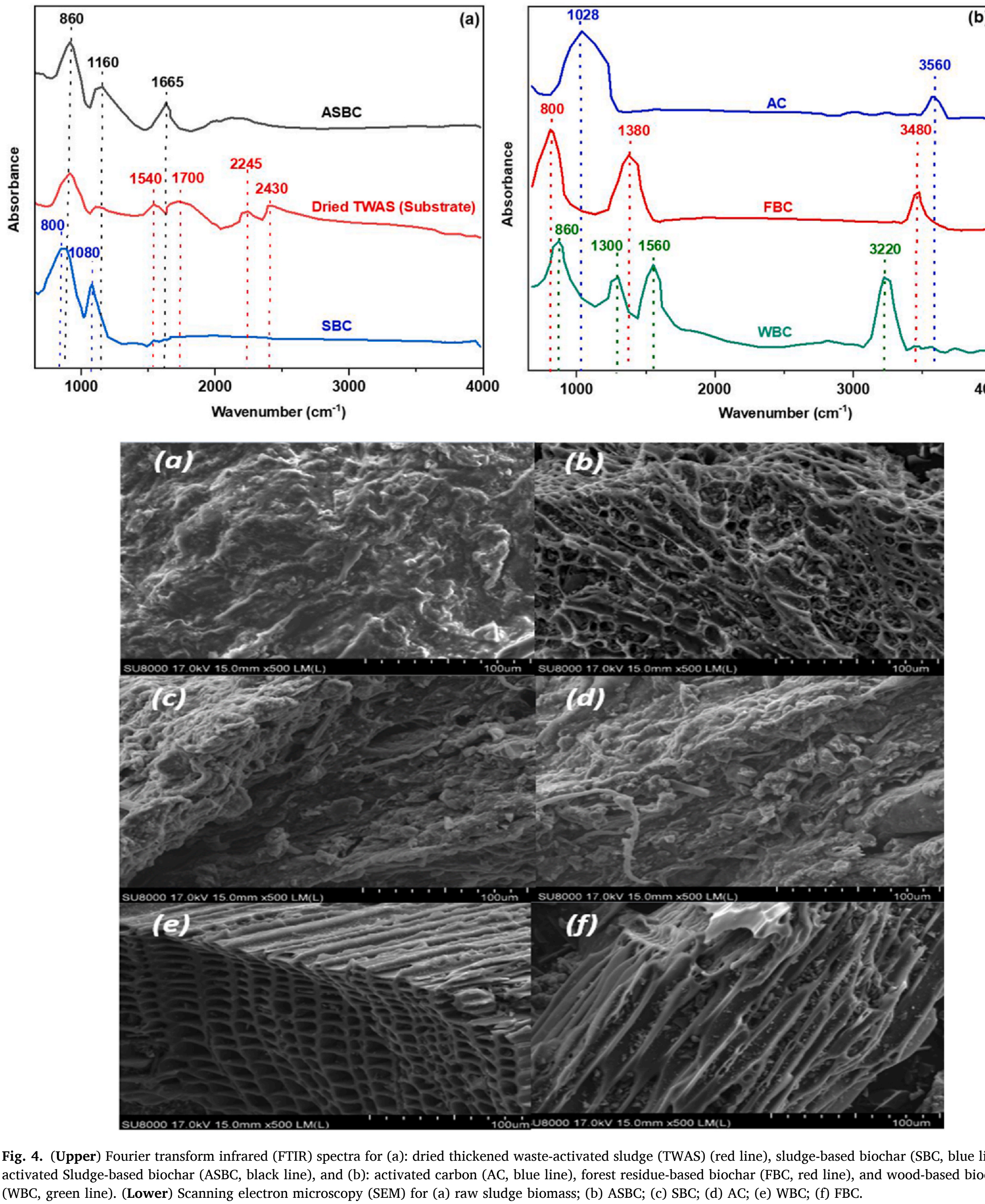

Fig. 4. (**Upper**) Fourier transform infrared (FTIR) spectra for (a): dried thickened waste-activated sludge (TWAS) (red line), sludge-based biochar (SBC, blue line), activated Sludge-based biochar (ASBC, black line), and (b): activated carbon (AC, blue line), forest residue-based biochar (FBC, red line), and wood-based biochar (WBC, green line). (**Lower**) Scanning electron microscopy (SEM) for (a) raw sludge biomass; (b) ASBC; (c) SBC; (d) AC; (e) WBC; (f) FBC.

AD, EC serves as an indirect measure of the ionic strength and salinity in the digester that can influence microbial activity and the overall digestion process (Bhuiyan et al., 2009). The addition of carbon-based additives is known to enhance the AD process by improving substrate-microbe interactions, possibly through mechanisms like direct interspecies electron transfer (DIET) (Lin et al., 2017). Previous studies have explored the role of conductive materials, including carbon-based additives, in promoting DIET, thereby enhancing methane production and process stability. The effectiveness of these additives is often evaluated through changes in EC, among other parameters, indicating shifts in the digester's biochemical environment were conducive to improved biogas yield (Baek et al., 2018; Yamada et al., 2015). Thus, the high EC for ASBC indicates that it may have a high potential to support microbial growth as compared with the commercially available biochars.

The pH values for the commercial biochars (pH 7.3 to 8.6) were markedly higher than the more acidic ASBS (pH 5.6) and SBC (pH 6.8). Except for ASBC, other additive pH values fell within the neutral and alkaline ranges which could be relevant to changes in functional groups due to pyrolysis at 500 °C (Dey et al., 2023). pH is critical in AD as it can significantly affect microbial activity and biogas production efficiency (Zhou et al., 2016). Optimal pH levels are essential for the growth of methanogenic archaea and bacteria responsible for organic matter breakdown. Too acidic or alkaline conditions can inhibit methanogens, reducing methane output or disrupting digestion. Regular monitoring and adjustment of pH are necessary to maintain a suitable environment for effective biogas generation, highlighting the role of pH control in the successful conversion of organic waste to renewable energy (Zhou et al., 2016; Han et al., 2020).

The metal(loid) concentrations for TWAS used in reactors and all carbon-based additives are shown in Table S1 along with maximum acceptable concentrations (MAC) for soils shown for comparative purposes. Overall, the concentrations for arsenic (As), cadmium (Cd), chromium (Cr), copper (Cu), lead (Pb), and iron (Fe) values were below their respective MAC for all samples. Generally, the ASBC and SBC samples had the lowest values for As, Cd, Cr, Co, and Pb as compared with WBC, FBC, and AC. In contrast, the Cu and Fe values were highest in the ASBC and SBC samples. The zinc (Zn) and manganese (Mn) concentrations both had values exceeding MACs (both at 50 mg/kg) with ASBC and SBC marginally above at 56 and 53 mg/kg, respectively. Interestingly, all additives other than WBC had exceedances for Mn (MAC = 15 mg/kg) with ASBC and SBC having the highest values of 73 and 78 mg/kg. Generally, Cu, Fe, and Zn are nutritional supplements and facilitate IET leading to higher biogas production from AD of wastewater sludges (Basiliko et al., 2001).

Overall, the FTIR spectra for TWAS biomass and carbon-based additives were in the range of 650–4000 cm^{-1} (Fig. 4). For all samples, peaks from 3000 to 3600 cm^{-1} indicate the -OH and -NH vibrations (Coates, 2000). Additionally, peaks identified in the 650–900 cm^{-1} span could be relevant to the out-of-plane bending vibrations of = C-H bonds, which are typical for substituted benzene compounds (Nunziante Cesaro et al., 2001). The TWAS FTIR results showed a peak at 1540 cm^{-1} representing N-H bending in amines, and 1700 cm^{-1} representing C=O stretching in carbonyl groups. Both carbonyl groups and amines can be from larger acidic functional groups (Lasch et al., 2002). The FTIR results are consistent with the pH results given that ASBC (pH 5.6) has acidic functional groups via activation by H_3PO_4 with a peak at ~1665 cm^{-1} indicating C=O stretching from carbonyl groups. Expectedly, the SBC FTIR results did not show acidic functional groups given the lack of activation with a peak at 1080 cm^{-1} representing C-O-C stretching vibrations in the presence of pyranose ring-containing groups (Asadi et al., 2021b). Additionally, a peak at 1160 cm^{-1} could be associated with S-O stretching or bending vibrations (González et al., 2017).

The FTIR for AC, FBC, and WBC detected basic functional groups known to be effective in creating neutral and alkaline environments. For example, for AC a peak at 1028 cm^{-1} indicates C-O and C-N single bonds. The C-O is associated with various acidic-oriented functional groups while the C-N bond is associated with basic amines (Ray et al., 2020). FBC a peak at 1380 cm^{-1} could be relevant to either C-H and N-H bending of amines with basic characteristics or C=C stretching in aromatic rings (Asadi et al., 2024). C-H and C=C can be found in both acidic and basic functional groups. Similarly, the WBC peaks were ~1300 cm^{-1} representing C-N stretching of amines, and 1560 cm^{-1} representing C-H bending and N-H bending of amines or C=C stretching in aromatic rings (Kayed et al., 2022).

Overall, the use of FTIR analysis can be useful in characterizing carbon-based additives in AD systems by identifying the specific functional groups and molecular bonds these additives introduce which can potentially affect biogas yield and quality (Zhuang et al., 2020). For instance, FTIR of carbon-based additives has demonstrated how these materials influence the chemical environment within the digester thereby enhancing microbial activity, stabilizing the digestion process, and/or improving the gas production efficiency (Zhuang et al., 2020).

The SEM results for raw sludge TWAS biomass, ASBC, SBC, AC, WBC, and FBC are presented in Fig. 4. Expectedly, the TWAS SEM results indicated a smooth, agglomerated structure with low porosity. Overall, all biochars show a porous, layered structure characterized by flaky particles on their surfaces due to the emission of combustion gases during the high-temperature pyrolysis process (Ingole et al., 2016). Porosity, surface area, and overall morphology of biochar are key determinants of its suitability for various applications including soil amendment, water retention, and contaminant adsorption (Lehmann et al., 2015; Singh et al., 2012). SEM analysis is a foundational tool in biochar research, offering insights into the material's microscopic structure, which directly impacts its environmental and agricultural applications.

3.3. AD of TWAS and carbon-based additives

3.3.1. Biogas production

The cumulative CH_4, CO_2, and total biogas production are presented in Fig. 5. After 30 d, the TAC treatment had the highest CH_4 yield with 352 mL CH_4/g VS, followed by TASBC at 332 mL CH_4/g VS, TFBC at 308 mL CH_4/g VS, TWBC at 272 mL CH_4/g VS, and TSBC at 252 mL CH_4/g VS (Fig. 5a). As expected, the control TWAS (TWAS-C) and non-mixed TWAS (TWAS-N) treatments had the lowest CH_4, and total biogas, yields at 221 and 194 mL CH_4/g VS, and 360 and 289 mL CH_4/gVS, respectively. Trends leading up to 30 d were reasonably consistent for all treatments overall. For example, the slightly increasing yields until the 10th day (Fig. 5a) can be attributed to the initiation and completion of hydrolysis and acidogenesis processes (Rocha-Meneses et al., 2022; Bajpai et al., 2017).

The CH_4 yield results were in agreement with the CH_4 production kinetic parameters determined by the modified Gompertz equation ($R^2 > 0.99$; Table 2). For all TWAS and carbon-based additives, the model predicted CH_4 yield (H) were in agreement with the experimental results, while the TAC treatment achieved the highest peak biomethane production rate (R_m) of 18.1 mL/g VS/d. However, this was closely followed by TASBC and TFBC with R_m values of 17.0 mL/g VS/d and 16.02 mL/g VS/d, respectively. Interestingly, the lag phase duration (λ) range was 3.4–8.9 d with the order of this duration being TAC < TWBC < TASBC < TFBC < TWAS-C < TSBC < TWAS-N (Table 2). This result indicates that the carbon-based additives markedly increased biogas production earlier in the digestion process. Similarly, Wang et al. (2020) reported that fir-based biochar additive increased the cumulative CH_4 production of an AD process by 11% attributing the increase to biochar redox-active components and conductivity (Wang et al., 2020a). Chiappero et al. (2021) investigated the effects of adding a sewage sludge-based biochar (10 g/L) to an AD system using waste-activated sludge (WAS) as the substrate and found a 17% CH_4 production enhancement (Chiappero et al., 2021). Lastly, Zhang et al. (2019) reported a 65% cumulative CH_4 production rate enhancement by using a corn straw-based biochar additive (15.9 g/L) in an AD system using

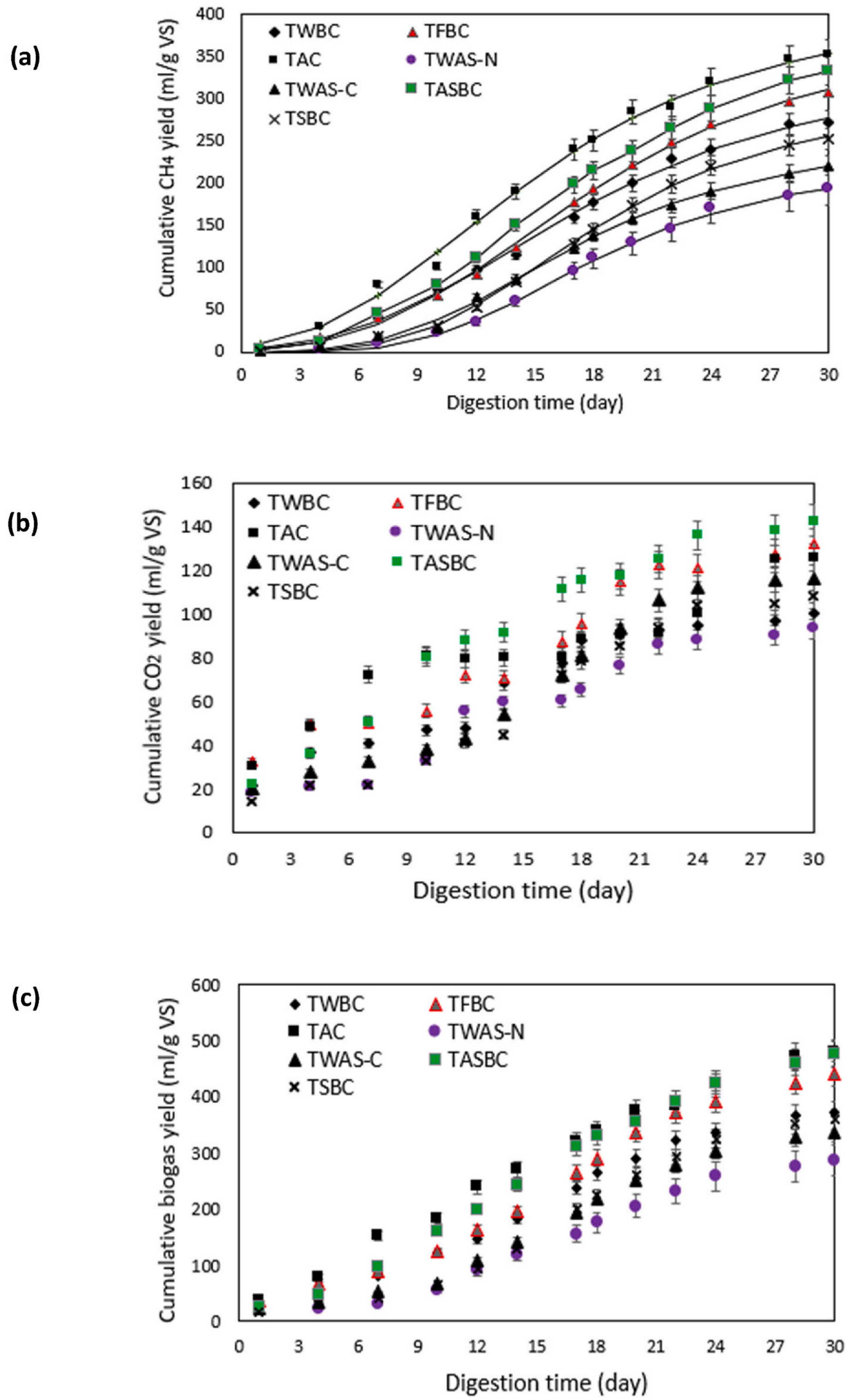

Fig. 5. Gas yields for the anaerobic digestion of thickened waste-activated sludge (TWAS) and carbon-based additives for (a) cumulative CH_4 yield from experiments (markers) and the Gompertz model (black solid lines); (b) cumulative CO_2 yield from experiments (markers) and (c) cumulative biogas yield from experiments. Note: Error bars indicate standard deviations for n = 3. See Section 2.3 for sample acronym information.

sewage sludge as the substrate.

The focus of biogas production is on the CH_4 yield; however, it is also important to determine the CO_2 and total biogas production yields. Overall, the range in CO_2 yields was markedly smaller than the CH_4 at 93–143 mL CO_2/g VS with the lowest value for TWAS-N and the highest for TASBC (Fig. 5b). Interestingly, given TAC had the highest CH_4 yield, while the TASBC at the highest CO_2 yield, the highest cumulative biogas yield was the same for these two treatments at 480 mL biogas/g VS (Fig. 5c). However, the overall total biogas yields followed a similar order as the CH_4 (Fig. 5b) and CO_2 (Fig. 5c) results.

3.3.2. Impacts of additives on removal efficiencies

To further assess the impacts of the addition of carbon-based additives on the AD process, the removal efficiencies of key parameters were

Table 2
Kinetic parameters for methane production at different reaction conditions. Note: H_0 = the actual biomethane yield; H = predicted biomethane yield (mL/g VS); R_m = is the peak biomethane production rate (mL/g VS/day); λ = the duration of lag phase (day). See Section 2.3 for sample acronym information.

Reactor	H_0	Kinetic model Parameters			
		H	R_m	λ	R^2
TWBC	272	278	13.84	5.0	0.99
TFBC	308	312	16.02	5.9	0.99
TAC	352	354	18.09	3.4	0.99
TWAS-N	196	195	12.03	8.9	0.99
TWAS-C	221	220	12.93	7.3	0.99
TASBC	333	334	17.01	5.3	0.99
TSBC	252	256	15.36	8.5	0.99

assessed including VS, NH_3-N, VFAs, and COD (Fig. 6). Note that for each of these parameters, a higher removal is considered to be beneficial to the AD process given higher removals result in higher biogas production rates. Overall, the TAC treatment had the highest removal efficiency for all four parameters at 70% for VS, 35% for NH_3-N, 83% for VFAs, and 53% for COD. The TASBC treatment had similar removals for these parameters, while the TFBC and TWBC treatments were marginally lower overall. As expected, the TWAS-N and TWAS-C treatments had the lowest removal efficiencies, while the TSBC treatment was marginally better than these two treatments having no biochar amendment.

Wang et al. (Wang et al., 2020a; Pan et al., 2019) and Pan et al. (Wang et al., 2020a; Pan et al., 2019) showed removal efficiencies can be attributed to high EC values and redox-active moieties of carbon-based additives that facilitated electron transfer and accelerated the mesophilic-type AD startup phase via minimizing intermediate product accumulations, specifically inhibitors. Biochar addition to AD processes could impact the balance among NH_3/NH_4^+, $CO_2/H_2CO_3/HCO_3^-$, and $C_xH_yCOOH/C_xH_yCOO^-$ thereby leading to NH_3-N reduction, VFA ionization, and neutralized pH levels (Baqer et al., 2022; Milojevic et al., 2021). Therefore, the system's buffer capacity would be more resilient to changes. Similarly, Zhang et al. (2019) reported 41% of VFA removal by using corn straw-based biochar during AD of sewage sludge which was 100% higher than without biochar. Covali et al. (2021) reported the NH_3-N removal enhancement (50%) versus control samples for a digestate using a wood pellet-based biochar. The noted beneficial properties of biochar include the degradation of similar parameters in conjunction with the enhancement in CH_4 production thereby suggesting that supplementation of carbon-based additives in AD systems is a valid strategy overall for efficient organic waste treatment with benefits to the environment and renewable energy production (Chen et al., 2008; Liu et al., 2021).

3.4. Microbial community analysis

Microbial community diversity and composition analysis for TWAS and carbon-based additive treatments were based on 3802 ZOTUs (Fig. 7 and S3 – S5). PCoA results indicated similar microbial community abundances for TAC, TSBC, and relatively TWAS-C; and TASBC, TWBC, and relatively TFBC given their positions in proximity to Axis 1, which captured ~66% of variations (Fig. S3). The Chao 1 diversity index was ~2850 for TWAS and increased for AD treatment groups that varied between 2900 and 3100 for TWAS-C, TFBC, and TASBC; and further increased to 3100–3300 for TWAS-N, TAC, TSBC, and TWBC. This indicates that the AD processes can culture new molecular species (Fig. S4a). The Shannon diversity index was ~5.9 for TWAS indicating a high level of biodiversity. After the AD processes, this index decreased to

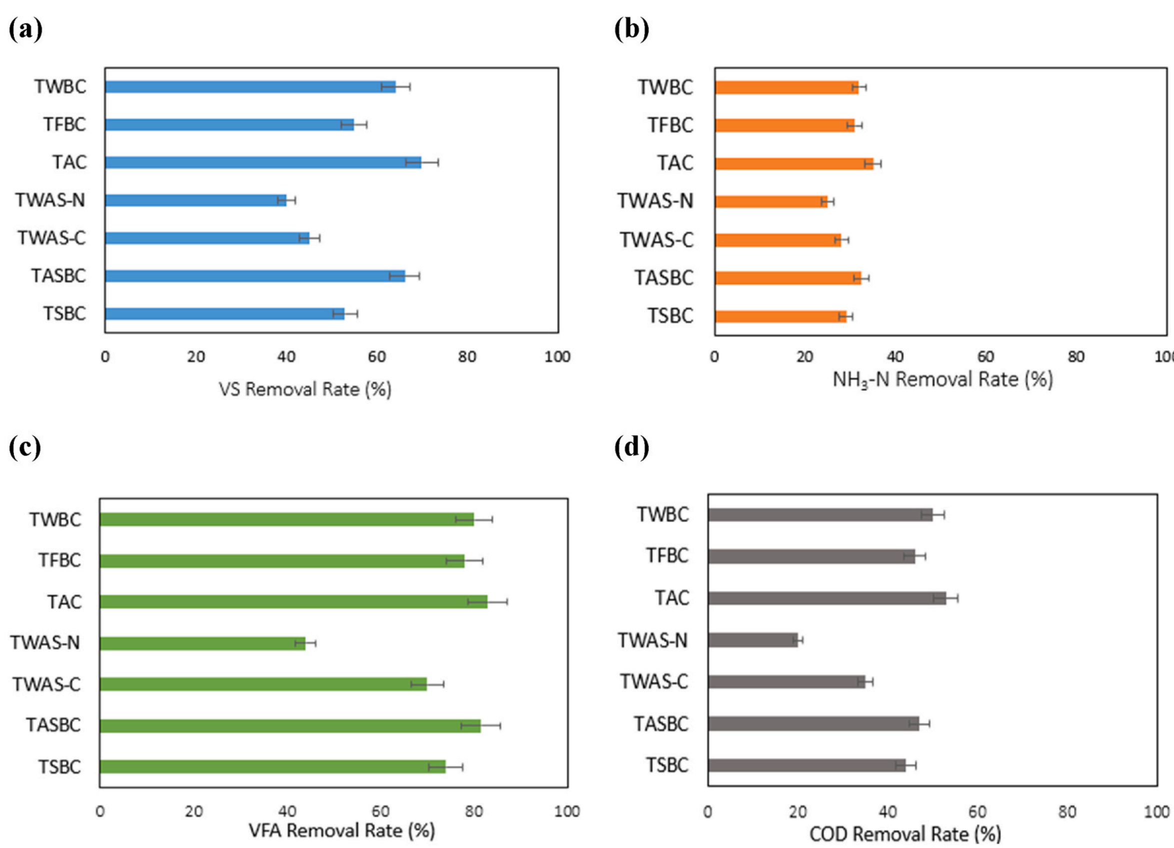

Fig. 6. Results for anaerobic digestion of thickened waste-activated sludge (TWAS) and carbon-based additive removal rates for (a) volatile solids, VS; (b) ammonia-nitrogen, NH_3-N; (c) volatile fatty acids, VFA; and (d) chemical oxygen demand, COD. Note: Error bars indicate standard deviations for n = 3. See Section 2.3 for sample acronym information.

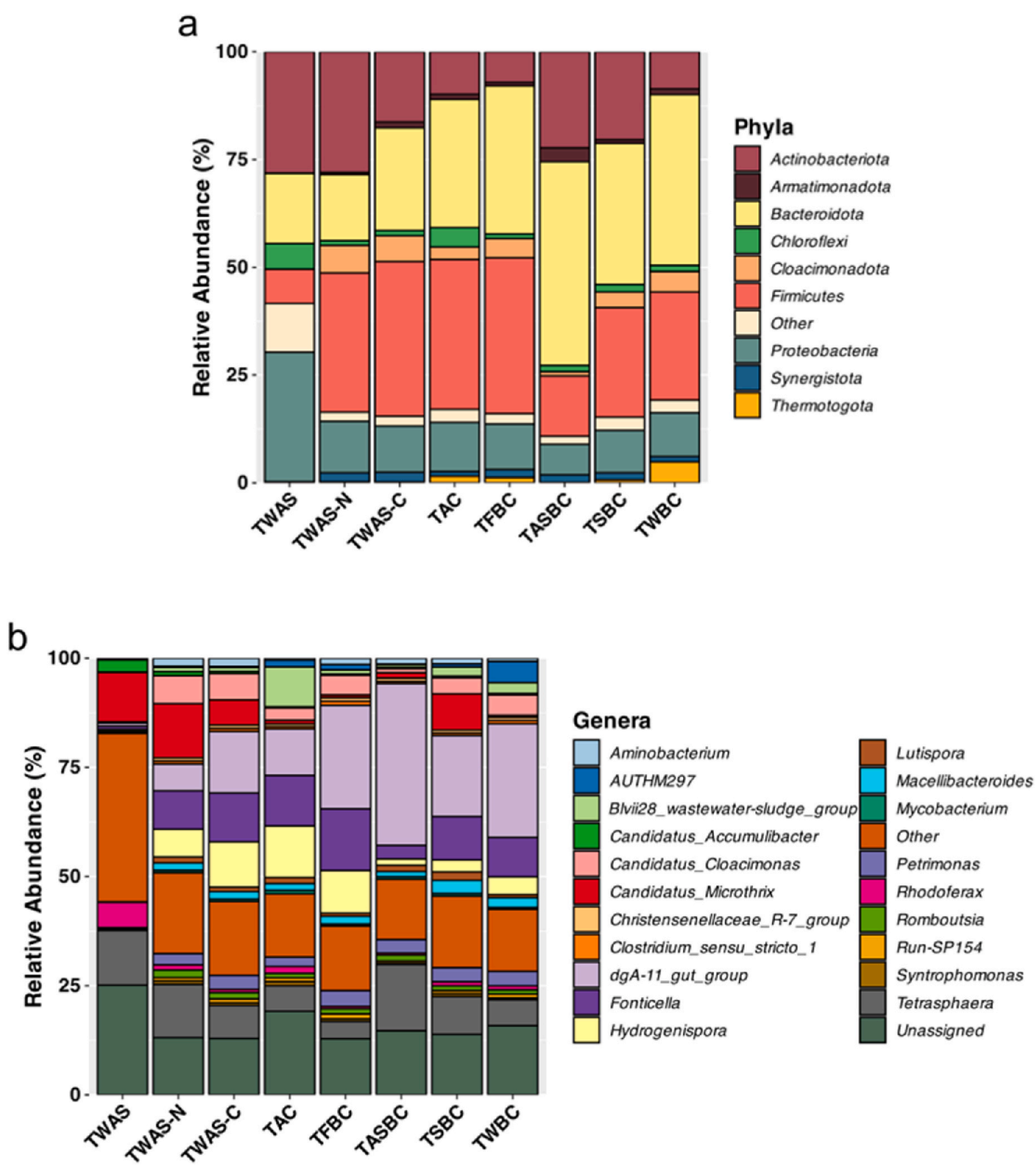

Fig. 7. Anaerobic digestion of thickened waste-activated sludge (TWAS) and carbon-based additive microbial community relative abundance (%) including: (a) bacteria phyla, (b) and bacteria genera. Phyla with relative abundance <1% and genera with relative abundance <0.5% were clustered into the 'Other' category in each instance.

4.6 for TASBC and varied between 5 and 5.5 for all other treatment groups. This indicates that the AD processes create environmental conditions for specific microbes to dominate (Fig. S4b).

Despite the lack of external inoculum for experiments herein, microbiome community results indicated the microbial community for anaerobic conversion was established. As shown in Fig. 7a, the developed microbial phyla were dominated by *Bacteroidota*, *Firmicutes,* and *Proteobacteria* which were consistent with previous studies that analyzed nine different AD microbial communities (Nguyen et al., 2019). Additionally, Stroot et al. (2001) developed mesophilic (37 °C) laboratory-scale digesters with retention times of 20 d to co-digest the organic fraction of municipal solid waste, primary sludge, and waste-activated sludge. Stroot et al. (2001) used three digesters including a control (no inoculum), cattle manure inoculum, and anaerobic digester sludge inoculum to evaluate the role of the inoculum in the digester's startup and performance. Interestingly, the control digester exhibited lower volatile fatty acid (VFA) accumulation, as compared to the inoculated digesters. Additionally, after 14 d, the ratio of the VFA concentration to the bicarbonate alkalinity reached below 1 (α) indicating the digestion stability (Poggi Varaldo et al., 1992).

Overall, microbiome community results indicated that the carbon-based additives could boost microbial groups involved in biogas production. For instance, *Bacteroidota* and *Firmicutes* were two types of phyla whose total relative abundances increased from 24% in TWAS to 50% for TWAS-N and 62–66% for other treatment groups. *Bacteroidota* and *Firmicutes* perform functions for the hydrolysis and acidogenesis stages, where they ferment the organic substrates and produce VFAs, alcohols, H_2, and CO_2 (Nguyen et al., 2019). Additionally, their spore-forming capacity increased resilience against environmental stress and led to more stable and efficient AD processes (Luiz et al., 2023). The genera analysis showed that the *DgA-11 gut group* prevailed in all the cultures with a relative abundance from ~1% for TWAS to 5.5% for TWAS-N, 14% for TWAS-C, and 11–37% for other treatment groups with carbon-based additives. The next abundant genera were *Tetrasphaera* and its relative abundance did not have a distinctive pattern. Overall, *Tetrasphaera* can enhance the hydrolysis of complex organic matter and increase the VFA availability (Baek et al., 2016). Additionally, the *Blvii28 wastewater-sludge group* was found to increase in TAC (9%) compared to other treatments (<2%). This group of bacteria is known to biotransform environmental contaminants (e.g., tetrabromobisphenol A, TBBPA) in anaerobic sludge and is positively related to methane recovery and production (Lefevre et al., 2019; Maza-Márquez et al., 2023).

The family *Rikenellaceae* appeared as a significant microbial group for all treatment groups increasing from <1% for TWAS to 7–38% for treatment groups (Fig. S5). Interestingly, biochar-added treatment groups had a marked relative abundance of >20%. This family plays a pivotal role in the biodegradation of complex substrates and contributes to VFA and biogas productions (Sparagon et al., 2022; Schwan et al., 2020). The exact mechanism by which *Rikenellaceae* influences biogas yield, whether through direct metabolic activities or synergistic interactions within the microbial community, warrants further investigation to elucidate their role in augmenting methane production within AD systems (Castellano-Hinojosa et al., 2018).

4. Conclusions

This study presents the potential of carbon-derived additives, specifically biochar TWAS, in increasing the production of biogas in an AD process. The CFD simulation study helped to obtain the optimal pyrolysis temperature (500 °C) by adjusting microwave operational factors (time and input power) during the ASBC production. Overall, novel furnace-type biochar (SBC) and microwave-type biochar (ASBC) were equally effective as compared to the commercial additives (AC, WBC, and FBC) with remarkable quality and quantity improvement in produced biogas. Specifically, the biogas composition showed an increase in CH_4 content to over 70% and a reduction in CO_2 levels to below 30%, compared to the control sample (TWAS-C) which produced biogas with 65% CH_4 and 35% CO_2. The increased CH_4 contribution in SBC- and ASBC-amended AD systems can indicate the potential of sludge-derived biochar to purify biogas and enhance CH_4 yield. The 30th-day cumulative biogas production rates were 475 mL biogas/g VS for the ASBC-amended AD system and 360 mL biogas/g VS for the SBC-amended AD system, showing increases of 41% and 7% respectively, compared to TWAS-C with 337 mL/g VS. Additionally, in comparison to TWAS-C, their relative key parameter removal efficiencies increased 8–25% for VS, 1–7% for NH_3-N, 4–13% for VFA, and 9–22% for COD. The increased removal rates indicated that the SBC- and ASBC-amended AD could produce biosolids with a lower residual organic content and inhibitory compounds. It would lead to a more stabilized biosolids product, which is desirable for further handling, processing, and use in agriculture.

Moreover, the results indicated that the carbon-derived additive utilization positively affected the AD bacterial communities. This not only points to a way to clean the biogas and digestate but also points out how it is possible to smartly manage municipal sludge and promote energy and resource independence for MWTPs and GHG emission reduction. Additionally, it holds the potential to reduce the operational costs and the processing times in sludge treatment plants. Strategic approaches involved in the optimization of sludge treatment practices, greenhouse gas emissions, and operational costs all provide critical input toward the development of MWTP management in Canada and worldwide. Based on these promising results from ASBC and SBC, our future work will focus on determining the optimal sludge-derived biochar particle sizes and dosages to investigate their impact on AD performance and biogas purification. Additionally, our future work will include the assessment of LCA and life cycle cost assessment (LCCA) to quantify the environmental and economic impacts of biochar production from MWTPs using furnace- and microwave pyrolysis technologies. Further, we will explore the potential of sludge-derived biochar for removing emerging pollutants, such as PFAS and microplastics, from the digestate.

CRediT authorship contribution statement

Rahman Zeynali: Writing – original draft, Methodology, Formal analysis. **Mohsen Asadi:** Writing – original draft, Methodology, Investigation. **Phillip Ankley:** Writing – original draft, Formal analysis. **Milena Esser:** Writing – review & editing, Investigation, Formal analysis. **Markus Brinkmann:** Writing – review & editing, Supervision. **Jafar Soltan:** Writing – review & editing, Supervision, Funding acquisition. **Kerry McPhedran:** Writing – review & editing, Funding acquisition, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jclepro.2024.143948.

Data availability

Data will be made available on request.

References

Abd El Aziz, M., et al., 2017. Prediction of biochar yield using adaptive neuro-fuzzy inference system with particle swarm optimization. In: 2017 IEEE PES PowerAfrica. IEEE.
Akbari, M., Salimi, H., Zeynali, R., Akbari, S., 2023. Enhancing an industrial feedwell design and operation using computational fluid dynamics. Computational Particle Mechanics 1–13.
Andreoli, C.V., Von Sperling, M., Fernandes, F., 2007. Sludge Treatment and Disposal. IWA publishing.
Ankley, P.J., et al., 2022. Effects of in situ experimental selenium exposure on finescale dace (Phoxinus neogaeus) gut microbiome. Environ. Res. 212, 113151.
Appels, L., et al., 2011. Anaerobic digestion in global bio-energy production: potential and research challenges. Renew. Sustain. Energy Rev. 15 (9), 4295–4301.
Asadi, M., Guo, H., McPhedran, K., 2020. Biogas production estimation using data-driven approaches for cold region municipal wastewater anaerobic digestion. J. Environ. Manag. 253, 109708.
Asadi, M., McPhedran, K., 2021a. Estimation of greenhouse gas and odour emissions from a cold region municipal biological nutrient removal wastewater treatment plant. J. Environ. Manag. 281, 111864.
Asadi, M., McPhedran, K., 2021b. Biogas maximization using data-driven modelling with uncertainty analysis and genetic algorithm for municipal wastewater anaerobic digestion. J. Environ. Manag. 293, 112875.
Asadi, M., Zaynali, R., Soltan, J., McPhedran, K.N., 2024. Biogas production enhancement from a cold region municipal wastewater anaerobic digestion. Canadian Society of Civil Engineering Annual Conference (in press).
Baek, G., Kim, J., Shin, S.G., Lee, C., 2016. Bioaugmentation of anaerobic sludge digestion with iron-reducing bacteria: process and microbial responses to variations in hydraulic retention time. Appl. Microbiol. Biotechnol. 100, 927–937.

Baek, G., Kim, J., Kim, J., Lee, C., 2018. Role and potential of direct interspecies electron transfer in anaerobic digestion. Energies 11 (1), 107.

Bajpai, P., Bajpai, P., 2017. Basics of Anaerobic Digestion Process. Anaerobic technology in pulp and paper industry, pp. 7–12.

Baqer, Y., Chen, X., 2022. A review on reactive transport model and porosity evolution in the porous media. Environ. Sci. Pollut. Control Ser. 29 (32), 47873–47901.

Basiliko, N., Yavitt, J.B., 2001. Influence of Ni, Co, Fe, and Na additions on methane production in Sphagnum-dominated Northern American peatlands. Biogeochemistry 52, 133–153.

Benis, K.Z., et al., 2022. A binary oxide-biochar composite for adsorption of arsenic from aqueous solutions: combined microwave pyrolysis and electrochemical modification. Chem. Eng. J. 446, 137024.

Bhuiyan, M.I.H., Mavinic, D.S., Beckie, R.D., 2009. Determination of temperature dependence of electrical conductivity and its relationship with ionic strength of anaerobic digester supernatant, for struvite formation. J. Environ. Eng. 135 (11), 1221–1226.

Bolyen, E., et al., 2019. Reproducible, interactive, scalable and extensible microbiome data science using QIIME 2. Nat. Biotechnol. 37 (8), 852–857.

Castellano-Hinojosa, A., et al., 2018. New concepts in anaerobic digestion processes: recent advances and biological aspects. Appl. Microbiol. Biotechnol. 102, 5065–5076.

Chen, Y., Cheng, J.J., Creamer, K.S., 2008. Inhibition of anaerobic digestion process: a review. Bioresour. Technol. 99 (10), 4044–4064.

Cheng, Q., de los Reyes, F.L., Call, D.F., 2018. Amending anaerobic bioreactors with pyrogenic carbonaceous materials: the influence of material properties on methane generation. Environmental Science: Water Research & Technology 4 (11), 1794–1806.

Chiappero, M., Berruti, F., Mašek, O., Fiore, S., 2021. Analysis of the influence of activated biochar properties on methane production from anaerobic digestion of waste activated sludge. Biomass Bioenergy 150, 106129.

Chojnacka, A., et al., 2015. Noteworthy facts about a methane-producing microbial community processing acidic effluent from sugar beet molasses fermentation. PLoS One 10 (5), e0128008.

Coates, J., 2000. Interpretation of Infrared Spectra, a Practical Approach.

Covali, P., et al., 2021. The effect of untreated and acidified biochar on NH3-N emissions from slurry digestate. Sustainability 13 (2), 837.

Dang, Y., et al., 2016. Enhancing anaerobic digestion of complex organic waste with carbon-based conductive materials. Bioresour. Technol. 220, 516–522.

Devi, P., Eskicioglu, C., 2024. Effects of biochar on anaerobic digestion: a review. Environ. Chem. Lett. 1–42.

Dey, D., Sarangi, D., Mondal, P., 2023. Biochar: porous carbon material, its role to maintain sustainable environment. In: Handbook of Porous Carbon Materials. Springer, pp. 595–621.

Eaton, A., 2005. American Public Health association (APHA), American water works association (AWWA), & water environment federation (WEF). Method 3030D "digestion for metal"(total element analysis & leaching test). Standard Methods for the Examination of Water & Wastewater, twenty-first ed. APHA-AWWA-WEF, Washington, DC.

Edgar, R.C., 2016. UNOISE2: improved error-correction for Illumina 16S and ITS amplicon sequencing. bioRxiv, 081257.

Gbangbo, K.R., et al., 2023. Influence of water content on hydrogen sulfide adsorption in biogas purification with Musa paradisiaca biochar. Chemistry Africa 6 (2), 657–665.

Gievers, F., Loewen, A., Nelles, M., 2021. Life cycle assessment of sewage sludge pyrolysis: environmental impacts of biochar as carbon sequestrator and nutrient recycler. Detritus 16 (94.10), 31025.

Golovko, O., et al., 2022. Organic micropollutants, heavy metals and pathogens in anaerobic digestate based on food waste. J. Environ. Manag. 313, 114997.

González, M., et al., 2017. Functionalization of biochar derived from lignocellulosic biomass using microwave technology for catalytic application in biodiesel production. Energy Convers. Manag. 137, 165–173.

Gurwick, N.P., Moore, L.A., Kelly, C., Elias, P., 2013. A systematic review of biochar research, with a focus on its stability in situ and its promise as a climate mitigation strategy. PLoS One 8 (9), e75932.

Han, Y., Green, H., Tao, W., 2020. Reversibility of propionic acid inhibition to anaerobic digestion: inhibition kinetics and microbial mechanism. Chemosphere 255, 126840.

Harirchi, S., et al., 2022. Microbiological insights into anaerobic digestion for biogas, hydrogen or volatile fatty acids (VFAs): a review. Bioengineered 13 (3), 6521–6557.

Hossain, M.A., Ganesan, P., Jewaratnam, J., Chinna, K., 2017. Optimization of process parameters for microwave pyrolysis of oil palm fiber (OPF) for hydrogen and biochar production. Energy Convers. Manag. 133, 349–362.

Hosseinian, A., et al., 2024. Life cycle assessment of sewage sludge treatment: comparison of pyrolysis with traditional methods in two Swedish municipalities. J. Clean. Prod. 455, 142375.

Hsieh, T., Ma, K., Chao, A., 2016. iNEXT: an R package for rarefaction and extrapolation of species diversity (H ill numbers). Methods Ecol. Evol. 7 (12), 1451–1456.

Huang, C., Mohamed, B.A., Li, L.Y., 2022. Comparative life-cycle assessment of pyrolysis processes for producing bio-oil, biochar, and activated carbon from sewage sludge. Resour. Conserv. Recycl. 181, 106273.

Ingole, R., Lokhande, B., 2016. Effect of pyrolysis temperature on structural, morphological and electrochemical properties of vanadium oxide thin films. J. Anal. Appl. Pyrol. 120, 434–440.

Jabbari, B., Jalilnejad, E., Ghasemzadeh, K., Iulianelli, A., 2021. Modeling and optimization of a membrane gas separation based bioreactor plant for biohydrogen production by CFD–RSM combined method. Journal of Water Process Engineering 43, 102288.

Kayed, K., Kurd, D.B., 2022. The effect of oxidation temperature on the (1000–1300) Cm− 1 band in FT-IR spectra of silicon oxide synthesized by thermal oxidation of silicon wafers. Silicon 14 (15), 10081–10086.

Kulkarni, A., et al., 2023. Advances in computational fluid dynamics modeling for biomass pyrolysis: a review. Energies 16 (23), 7839.

Lasch, P., Boese, M., Pacifico, A., Diem, M., 2002. FT-IR spectroscopic investigations of single cells on the subcellular level. Vib. Spectrosc. 28 (1), 147–157.

Lefevre, E., et al., 2019. Acetate promotes microbial reductive debromination of tetrabromobisphenol A during the startup phase of anaerobic wastewater sludge bioreactors. Sci. Total Environ. 656, 959–968.

Lehmann, J., Joseph, S., 2015. Biochar for Environmental Management: Science, Technology and Implementation. Routledge.

Leng, L., Huang, H., 2018. An overview of the effect of pyrolysis process parameters on biochar stability. Bioresour. Technol. 270, 627–642.

Lin, R., et al., 2017. Boosting biomethane yield and production rate with graphene: the potential of direct interspecies electron transfer in anaerobic digestion. Bioresour. Technol. 239, 345–352.

Liu, M., Wei, Y., Leng, X., 2021. Improving biogas production using additives in anaerobic digestion: a review. J. Clean. Prod. 297, 126666.

Liu, X., et al., 2022. Enhanced biogas production in anaerobic digestion of sludge medicated by biochar prepared from excess sludge: role of persistent free radicals and electron mediators. Bioresour. Technol. 347, 126422.

Lozupone, C., et al., 2011. UniFrac: an effective distance metric for microbial community comparison. The ISME journal 5 (2), 169–172.

Luiz, F.N., et al., 2023. Metataxonomic characterization of the microbial community involved in the production of biogas with microcrystalline cellulose in pilot and laboratory scale. World J. Microbiol. Biotechnol. 39 (7), 184.

Mayer, F., Bhandari, R., Gäth, S.A., 2021. Life cycle assessment of prospective sewage sludge treatment paths in Germany. J. Environ. Manag. 290, 112557.

Maza-Márquez, P., et al., 2023. Microbial indicators of efficient performance in an anaerobic/anoxic/aerobic integrated fixed-film activated sludge (A2O-IFAS) and a two-stage mesophilic anaerobic digestion process. Chemosphere, 139164.

Milojevic, N., Cydzik-Kwiatkowska, A., 2021. Agricultural use of sewage sludge as a threat of microplastic (Mp) spread in the environment and the role of governance. Energies 14 (19), 6293.

Motasemi, F., Gerber, A., 2018. Multicomponent conjugate heat and mass transfer in biomass materials during microwave pyrolysis for biofuel production. Fuel 211, 649–660.

Mumme, J., Srocke, F., Heeg, K., Werner, M., 2014. Use of biochars in anaerobic digestion. Bioresour. Technol. 164, 189–197.

Nansubuga, I., et al., 2015. Digestion of high rate activated sludge coupled to biochar formation for soil improvement in the tropics. Water Res. 81, 216–222.

Nguyen, L.N., Nguyen, A.Q., Nghiem, L.D., 2019. Microbial Community in Anaerobic Digestion System: Progression in Microbial Ecology. Water and wastewater treatment technologies, pp. 331–355.

Nizzetto, L., Futter, M., Langaas, S., 2016. Are Agricultural Soils Dumps for Microplastics of Urban Origin? ACS Publications.

Norberto, J., Benis, K.Z., McPhedran, K.N., Soltan, J., 2023. Microwave activated and iron engineered biochar for arsenic adsorption: life cycle assessment and cost analysis. J. Environ. Chem. Eng. 11 (3), 109904.

Nunziante Cesaro, S., et al., 2001. Mass and FTIR spectroscopic investigations of gaseous manganese tetrafluoride. Inorg. Chem. 40 (1), 179–181.

O Connor, J., et al., 2022. Physical, chemical, and microbial contaminants in food waste management for soil application: a review. Environmental Pollution 300, 118860.

Pan, J., et al., 2019. Achievements of biochar application for enhanced anaerobic digestion: a review. Bioresour. Technol. 292, 122058.

Panwar, N., Pawar, A., Salvi, B., 2019. Comprehensive review on production and utilization of biochar. SN Appl. Sci. 1, 1–19.

Poggi Varaldo, H.M., Oleszkiewicz, J.A., 1992. Anaerobic co-composting of municipal solid waste and waste sludge at hlgh total solids levels. Environmental technology 13 (5), 409–421.

Ray, A., Banerjee, A., Dubey, A., 2020. Characterization of biochars from various agricultural by-products using FTIR spectroscopy, SEM focused with image processing. Int. J. Agric. Environ. Biotechnol. 13 (4), 423–430.

Rocha-Meneses, L., et al., 2022. Current progress in anaerobic digestion reactors and parameters optimization. Biomass Conversion and Biorefinery 1–24.

Rognes, T., et al., 2016. VSEARCH: a versatile open source tool for metagenomics. PeerJ 4, e2584.

Schwan, B., et al., 2020. Chemically stressed bacterial communities in anaerobic digesters exhibit resilience and ecological flexibility. Front. Microbiol. 11, 867.

Shanmugam, S.R., Adhikari, S., Nam, H., Sajib, S.K., 2018. Effect of bio-char on methane generation from glucose and aqueous phase of algae liquefaction using mixed anaerobic cultures. Biomass Bioenergy 108, 479–486.

Shen, Y., et al., 2016. Towards a sustainable paradigm of waste-to-energy process: enhanced anaerobic digestion of sludge with woody biochar. J. Clean. Prod. 135, 1054–1064.

Singh, B.P., Cowie, A.L., Smernik, R.J., 2012. Biochar carbon stability in a clayey soil as a function of feedstock and pyrolysis temperature. Environmental science & technology 46 (21), 11770–11778.

Sparagon, W.J., et al., 2022. Fine scale transitions of the microbiota and metabolome along the gastrointestinal tract of herbivorous fishes. Animal Microbiome 4 (1), 1–21.

Stroot, P.G., McMahon, K.D., Mackie, R.I., Raskin, L., 2001. Anaerobic codigestion of municipal solid waste and biosolids under various mixing conditions—I. Digester performance. Water Res. 35 (7), 1804–1816.

Takahashi, S., et al., 2014. Development of a prokaryotic universal primer for simultaneous analysis of Bacteria and Archaea using next-generation sequencing. PLoS One 9 (8), e105592.
Vlyssides, A., Karlis, P., 2004. Thermal-alkaline solubilization of waste activated sludge as a pre-treatment stage for anaerobic digestion. Bioresour. Technol. 91 (2), 201–206.
Wan, Y., et al., 2009. Microwave-assisted pyrolysis of biomass: catalysts to improve product selectivity. J. Anal. Appl. Pyrol. 86 (1), 161–167.
Wang, J., et al., 2018. Impact of filled materials on the heating uniformity and safety of microwave heating solid stack materials. Processes 6 (11), 220.
Wang, H., Larson, R., Runge, T., 2019. Impacts to hydrogen sulfide concentrations in biogas when poplar wood chips, steam treated wood chips, and biochar are added to manure-based anaerobic digestion systems. Bioresour. Technol. Rep. 7, 100232.
Wang, P., et al., 2020a. Enhancement of biogas production from wastewater sludge via anaerobic digestion assisted with biochar amendment. Bioresour. Technol. 309, 123368.
Wang, R., et al., 2020b. Production, properties, and catalytic applications of sludge derived biochar for environmental remediation. Water Res. 187, 116390.
Westerholm, M., Calusinska, M., Dolfing, J., 2022. Syntrophic propionate-oxidizing bacteria in methanogenic systems. FEMS (Fed. Eur. Microbiol. Soc.) Microbiol. Rev. 46 (2), fuab057.
Wickham, H., 2011. *ggplot2.* Wiley interdisciplinary reviews. Comput. Stat. 3 (2), 180–185.
Yamada, C., et al., 2015. Conductive iron oxides accelerate thermophilic methanogenesis from acetate and propionate. J. Biosci. Bioeng. 119 (6), 678–682.
Zeynali, R., et al., 2023. Investigating the effects of operating parameters on the performance of sorption-enhanced membrane reactor for ethanol steam reforming reaction using computational fluid dynamics method. Phys. Fluids 35 (10).
Zhang, M., Li, J., Wang, Y., Yang, C., 2019. Impacts of different biochar types on the anaerobic digestion of sewage sludge. RSC advances 9 (72), 42375–42386.
Zhao, L., Cao, X., Mašek, O., Zimmerman, A., 2013. Heterogeneity of biochar properties as a function of feedstock sources and production temperatures. J. Hazard Mater. 256, 1–9.
Zhou, J., et al., 2016. Biogas production and microbial community shift through neutral pH control during the anaerobic digestion of pig manure. Bioresour. Technol. 217, 44–49.
Zhou, H., Brown, R.C., Wen, Z., 2020. Biochar as an additive in anaerobic digestion of municipal sludge: biochar properties and their effects on the digestion performance. ACS Sustain. Chem. Eng. 8 (16), 6391–6401.
Zhuang, J., et al., 2020. Observation of potential contaminants in processed biomass using fourier transform infrared spectroscopy. Appl. Sci. 10 (12), 4345.
Zornoza, R., et al., 2016. Stability, nutrient availability and hydrophobicity of biochars derived from manure, crop residues, and municipal solid waste for their use as soil amendments. Chemosphere 144, 122–130.