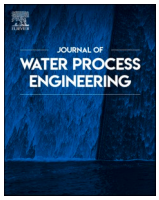
Journal of Water Process Engineering 59 (2024) 105024

Contents lists available at ScienceDirect

Journal of Water Process Engineering

journal homepage: www.elsevier.com/locate/jwpe



Degradation and removal mechanisms of mixed veterinary antibiotics in swine manure during anaerobic and storage treatments: Validation and characterization

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ARTICLE INFO

Editor: Wenshan Guo

Keywords:
Mixed antibiotics
Removal mechanisms
Anaerobic treatment
Competitive adsorption
Modified bentonite

ABSTRACT

The presence of mixed veterinary antibiotics (MVAs) in swine farms raises environmental concerns, necessitating a thorough examination of its degradation and removal mechanisms. In this study, a series of batch experiments were conducted to explore the degradation, biodegradation, and adsorption of mixed tetracycline (TC), oxytetracycline (OTC), norfloxacin (Norf), and sulfadiazine (SDZ) under anaerobic and storage swine manure treatments. Results revealed a varied antibiotics removal pathway despite their mixture, with dominant degradation mechanism for TC, biodegradation for OTC and SDZ, and adsorption for Norf. Anaerobic conditions exhibited higher MVAs removal (average of 44.65 ± 14.9 %) compared to storage conditions (average of 36.22 ± 11.2 %). MVAs' removal pathways showed different significant responses to contact time, temperature, and initial concentration. On the other hand, Degradation half-lives of MVAs were prolonged, particularly for TC and Norf, in contrast to OTC and SDZ. These half-lives were overall shorter under anaerobic conditions than in storage conditions. Storage condition was favourable for MVAs adsorption onto swine sludge than anaerobic condition. Furthermore, MVAs adsorption behaviour into modified bentonite (MB) showed a high capacity except for SDZ, higher adsorption under storage conditions compared to anaerobic conditions. MVAs' adsorption onto MB fitted well with the pseudo-second kinetic model. The outcomes of this research would provide a more in-depth understanding of MVAs removal mechanisms during the anaerobic and storage treatments of animal wastes.

1. Introduction

Demand for animal protein has led to rapid, intensive swine farming industrialization worldwide [1,2]. Veterinary antibiotics (VAs) are widely consumed to keep swine healthy and increase growth productivity. In this regard, the most administrated antibiotics used in the swine husbandry are tetracycline (TC), oxytetracycline (OTC), norfloxacin (Norf), and sulfadiazine (SDZ) according to [3]. However, based on the chemical composition of the employed antibiotics in livestock industry, 25–90 % of administered VAs are excreted in the urine and feces of the animal [4,5]. These excretions not only serve as important carriers for the growth of antibiotic resistance in livestock and human, but they also gradually reduce their effectiveness over time [6]. Consequently, OTC and TC were observed in swine manure at concentration levels of up to 183.5 mg/kg and 2.08 mg/kg, respectively [7,8]. The contents of SDZ and Norf in swine manures were reported to be 235.1 and 5.5 mg/kg, respectively [9,10]. Therefore, swine manure is a substantial source for the spread of TC, OTC, SDZ, and Norf into the

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https://doi.org/10.1016/j.jwpe.2024.105024
Received 20 December 2023; Received in revised form 14 February 2024; Accepted 17 February 2024
Available online 22 February 2024

environment. For animal wastes management, the anaerobic treatment process (i.e., anaerobic digestion or anaerobic lagoons) is a broadly applied technology due to its high outcome performance and low required cost [11]. Likewise, the storage process (i.e., open lagoons or tanks) is also commonly used in intensive swine farming systems as a simple waste management method before manure's land application [12,13]. Although anaerobic and storage treatment practices showed their potentially to remove VAs depending on their types and influent concentrations as well as the process itself's operating conditions, most previous studies mainly reflected the removal of individual VAs and figuring out their impacts [14–16]. However, antibiotics are not present as single compound in the environment [1,17]. Antibiotics may exist as a mixtures containing several antibiotics, these mixtures content may vary depending on the industry from which it originates. Meanwhile, antibiotics have a different characteristic that may generate obstacles on their removal performances when they exist in mixtures. It has been observed that the antibiotics in mixtures caused a more significant inhibition on the anaerobic process than a single antibiotic [18,19]. Furthermore, studies about single antibiotics removal in anaerobic and storage processes focused on their removal efficiencies and mechanisms. However, information about the removal mechanisms of mixed veterinary antibiotics (MVAs) belonging to different classes is still limited.

Additionally, study of antibiotics degradation pathways can help in the design and optimization of treatment technologies. Biodegradation, natural degradation, and adsorption have been suggested as the primary mechanisms for removing VAs during anaerobic and storage treatment processes [5,20]. Biodegradation is considered a substantial removal pathway for VAs in biological processes. VAs' biodegradation in animal manures was found in averages of 64.4, 73.6, and 55.3 %, respectively, for tetracyclines (TCs) (i.e., TC and OTC), sulfonamides (SAs) (i.e., SDZ), and fluoroquinolones (FQs) (i.e., Norf) [5]. Natural degradation of antibiotics represents the factors a part from microorganisms that may breakdown antibiotics such as abiotic mechanisms and temperature. These mechanisms can influence the persistence, effectiveness, and potential environmental impacts of antibiotics mixture, including their potential to contribute to antibiotic resistance in bacteria [21]. However, literature lacks of sufficient research in this point, that needs a considerable attention. Additionally, the biodegradation kinetics of antibiotics in light of their mixing also require considerable consideration. On the other hand, the adsorption pathway was found to be of the foremost pathway for TCs into sludge ranging by an average of 72.1 %. While was humble for FQs (i.e., Norf) by 55.1 % and low for SAs (i.e., SDZ) by 32.2 % during the biological processes [5]. At the same time, the adsorption of VAs onto clays material such as bentonite was found to be extreme for TC and Ciprofloxacin [22,23].

Additionally, the previously conducted studies mostly explored single antibiotic's degradation and removal mechanisms; however, little is known about the behaviour of MVAs during the anaerobic and storage treatment of swine manure. Several phenomena such as adsorption competition, degradation competition antagonism or synergistic effects, and higher inhibition effects on the treatment process stability and performance [19,24–27]. These phenomena may impact the removal pathways of MVAs during treatment processes that needs an extensive research focus. Thus, understanding the removal mechanisms and fate of MVAs during anaerobic and storage treatments of swine manure would assist their environmental risk management in the environment and would help improving wastewater treatment abilities.

Therefore, to fill this information gap, the current study investigated the fate of mixed TC, OTC, SDZ, and Norf during anaerobic and storage treatment processes. Moreover, mixed antibiotics' degradation, biodegradation, and adsorption mechanisms were explored. The potential adsorption ability of modified bentonite for mixed antibiotics and their kinetics were likewise examined. Moreover, the first-order kinetic model was used to assess the biodegradation data of MVAs in swine manure, as well as adsorption kinetics of MVs onto MB using Pseudo-first order and Pseudo-second order models were investigated according to [4].

2. Material and methods

2.1. Materials

The swine manure substrate used in the current study was collected from a swine farm in Beijing, China without medical treatment during last three months prior the study. The characteristics of swine manure are given in Table 1. VAs were obtained from Sigma-Aldrich to prepare the VAs stock solution, TC (CAS no. 60–54-8, >90 %), OTC (CAS no. 6153-64-6, >98 %), Norf (CAS no. 70458–96-7, >98 %), and SDZ (CAS no. 68–35-9, >98 %). Methanol (CH_3OH, Sigma-Aldrich) and acetonitrile (C_2H_3N, Sigma-Aldrich) of High-performance liquid chromatography (HPLC) grade, disodium ethylene–diamine tetraacetate (Na_2EDTA), Oxalic acid, sodium dihydrogen phosphate (NaH_2PO_4), and disodium hydrogen phosphate (Na_2HPO_4) of analytical-reagent grade were obtained from Beijing Chemicals Company (Beijing, China).

2.2. Experimental setup

The experimental setup of this work consisted of three sets of batch experiments. According to our previous study [28], storage of swine manure that is similarly to the open lagoons with no aeration at 30 °C was maximized the removal of MVAs and decreased their consequences. Also, anaerobic conditions of swine treatment at 37 °C was employed to remove MVAs in our previous study [26]. Thus, exploring the removal pathways of MVAs under both conditions was undertaken in this study as follows:

In the first experiments (degradation), three levels of MVAs, i.e., 12.5, 25, and 50 mg VAs/l were chosen based on their detection levels in the literature [5]. The mixed antibiotics was in equal ratios to clarify their removal pathways, ensuring no influence from each other due to their higher dosage. In this experiment, deionized water (DI) without substrate was utilized to evaluate the stability of MVAs under abiotic and natural degradation followed the study by [29]. Around 100 ml of DI was taken in 120 ml glass bottles in triplicates under the following operation conditions, anaerobic (37 °C) and storage (30 °C) and incubated in dark place. T1, T2, and T3 represented the levels of MVAs concentration, while R1 and R2 represented the anaerobic and storage conditions, respectively (Table 2).

In the second experiment (biodegradation and adsorption into sludge), biodegradation removal pathway of MVAs and their adsorption into sludge pathway were investigated in this experiment. Thus, batch studies were conducted using swine manure as substrate (total solids 5 %). Whereas, around 100 ml of swine manure (TS 5 %) was taken in 120 ml glass bottles using different spikes of MVAs (12.5, 25, and 50 mg/l, represent T1, T2, and T3), and the bottles were placed under anaerobic (37 °C) and storage (30 °C) conditions, represented by R3 and R4, respectively.

In the third experiment (adsorption into MB), the adsorption affinity of MVAs into modified bentonite (MB) was investigated in this study (MB characteristics was illustrated in Table S1). Thus, sodium bentonite was used as raw material for the preparation of MB. Raw bentonite was

Table 1
Characteristics of swine manure that used in the current study.

	Unit	Raw swine manure
pH		7.56 ± 0.2
TS	%	5.13 ± 0.2
VS	% of TS	78.0 ± 0.4
Ash	%	21.9 ± 0.6
COD	g/l	39.7 ± 2.5
VFAs	g/l	1.06 ± 0.3
TN	g/l	4.30 ± 0.7
TAN	g/l	1.50 ± 0.2

TS: total solid; VS: volatile solid; COD: chemical oxygen demand; VFAs: total volatile fatty acids; TN: total nitrogen; TAN: total ammonium nitrogen.

Table 2
Experimental setup of the batch tests.

Reactor	Anaerobic condition	Storage condition	Substrate	Antibiotics (mg/l)			Antibiotics removal pathway
				12.5	25	50	
R1 (T1, T2, T3)	+	−	DI	+	+	+	Degradation
R2 (T1, T2, T3)	−	+		+	+	+	
R3 (T1, T2, T3)	+	−	Swine liquid manure	+	+	+	Biodegradation/sludge adsorption
R4 (T1, T2, T3)	−	+		+	+	+	
R5 (T1, T2, T3)	+	−	DI + MB (1, 5, and 10 g/L)	−	−	+	Adsorption
R6 (T1, T2, T3)	−	+		−	−	+	

"R (1, 2, 3, 4, 5, 6)" represented the reactors. "+" indicated "with", "-"indicated "without". DI indicated distilled water. MB indicated Modified Bentonite.

thermally treated at 300 °C for 2 h, as described by [30]. MB characterization was depicted in (section S1.2) in supplementary material. The adsorption studies were conducted using three levels of MB, i.e., 1, 5, and 10 g/L, under a MVAs concentration of 50 mg/l, represent T1, T2, and T3, respectively. The treatment set of studies conducted under storage was represented by R5 (T1, T2, and T3) while under anaerobic conditions was represented by R6 (T1, T2, and T3). pH of the substrate was neutral. Details about the experimental setup are given in Table 2.

In order to prepare the standard stock solutions of each VAs, (1 mg-VA/mL) of each VAs was dissolved in 10 % methanol, then completed with distilled water. For each experiment, individual VAs solutions were mixed right before spiking to test assays to the concentrations, i.e., 12.5, 25, and 50 mg/l. Following this, with 99.9 % N_2, all glass bottles were flushed and immediately covered with rubber sealant, and closed with aluminum caps to guarantee the anaerobic conditions and airproof to displace any oxygen present, representing anaerobic digestion conditions (R1, R3, and R5) at a temperature level of 37 °C. While opened bottles were used in the case of storage conditions (R2, R4, and R6) under a temperature level of 30 °C. All batch experiments were carried out under dark conditions to avoid antibiotics photo-degradation. All experiments were conducted in triplicate to get statistically reliable results.

2.3. Antibiotics detection analysis

The samples for each experimental set were collected periodically and pre-treated following the method developed by [31]. Antibiotics detection from liquid and solid fractions analysis along with instrumental running conditions were explained in details in supplementary material (section S1.1).

2.4. Data analysis

2.4.1. Antibiotics adsorption, biodegradation and degradation calculations

The proportion of MVAs removed from DI was used to quantify the antibiotics' degradation, while MVAs' biodegradation in swine manure was calculated according to Eq. (1):

$$\text{Biodegradation (\%)} = \text{Total removal (\%)}–\text{Degradation (\%)} \quad (1)$$

The removal rate of MVAs is used to calculate the removal percentage of antibiotics that whether not adsorbed into digested sludge or not remained in the liquid fraction. The removal efficiency was calculated according to the following equation: (2):

$$R = \frac{(Ci - Cf)}{Ci} * 100 \quad (2)$$

where R is the removal rate (%), C_i is the MVAs influent concentration, and C_f is the MVAs effluent concentration.

2.4.2. Biodegradation kinetics

The first-order kinetic model was used to assess the biodegradation data of MVAs in swine manure and estimating rate of pollutant degradation according to [4]. Following equation was used for analysing kinetics of MVAs (3):

$$C_t = C_o * e^{-k1*t} \quad (3)$$

where C_0 is the initial concentration of VAs, C_t represents the concentration of the VAs at time t, and k1 represents the degradation rate constant. Half-lives, or t1/2 may be estimated using the formula (DT50 = ln 2/k).

2.4.3. Antibiotics' adsorption kinetics

The amount of MVAs sorbed onto MB was calculated using the following equation:

$$Q_e = \frac{C_o - C_e}{m} \quad (4)$$

where Q_e (mg/g) is the equilibrium sorption capacity, C_o and C_e (mg/l) are initial and equilibrium aqueous phase MVAs concentrations, respectively, and m (g/l) is the dosage concentration of MB.

The adsorption kinetics of MVs onto MB were fitted using Pseudo-first order and Pseudo-second order models to investigate the kinetics characteristics during the adsorption process according to [4] as follows:

Pseudo-first-order kinetics

$$ln(q_e - q_t) = lnq_e - k_1 t \quad (5)$$

Pseudo-second-order kinetics

$$\frac{t}{q_t} = \frac{1}{k_2 q_e^2} + \frac{t}{q_e} \quad (6)$$

where, k_1 is the rate constant of Pseudo-first-order kinetics (/h), k_2 is the pseudo-second-order rate constant (g/mg·h), qe and qt are the amounts of MVAs adsorbed on MB at equilibrium and at time t (h).

2.4.4. Statistical analysis

The collected data are presented as the average of duplicated samples ± standard deviation (SD). One-way analysis of variance (ANOVA) combined with Tukey's honestly significant differences (HSD) post-hoc tests were used to determine significant differences, with *p*-values <0.05 considered significant. For analyses, SPSS statistics (v.25, IBM Corporation, United States, 2017), and the linear regression model (lm function) were utilized using the open-access software R (v. 4(2023)) and RStudio (v. 4.3.1©, 2022 RStudio, Inc.).

3. Results and discussion

3.1. MVAs removal pathways

In this study, MVAs removal pathways including degradation, biodegradation and adsorption into sludge, and adsorption into MB during storage and anaerobic treatment of swine manure were studied. The overall removal of MVAs from swine manure revealed that incubation time had a significant ($p < 0.05$) influence on the removal process. The temperature difference between anaerobic and storage conditions showed a significant ($p < 0.05$) impact on TC, OTC, and SDZ,

while no significance appeared for Norf. Additionally, the initial concentration had a significant impact ($p < 0.05$) only on TC removal. A detailed description of the MVAs removal pathways is provided in the following sections.

3.1.1. MVAs degradation pathway

VAs used to keep swine healthy may undergo a series of reactions that would cause its attenuation naturally [20]. The outcomes from degradation experiment in the current study depicted that all MVAs degradation were significantly ($p < 0.05$) affected by the incubation time. Temperature difference in both treatment processes showed a significant impact on Norf's degradation, while no difference was observed on the degradation of TC, OTC, and SDZ. (Fig. 1). Moreover, it was perceived that TC had the highest degradation rate among co-antibiotics under anaerobic conditions, and its degradation efficiency increased with incremented concentration.

As shown in Fig. 1a, the average degradation percentage of TC under anaerobic conditions was 20.7 % (Fig. 1a) compared to 16.5 % with storage conditions (Fig. 1b). This difference in degradation was non-significant ($p < 0.05$), which means the difference in temperature

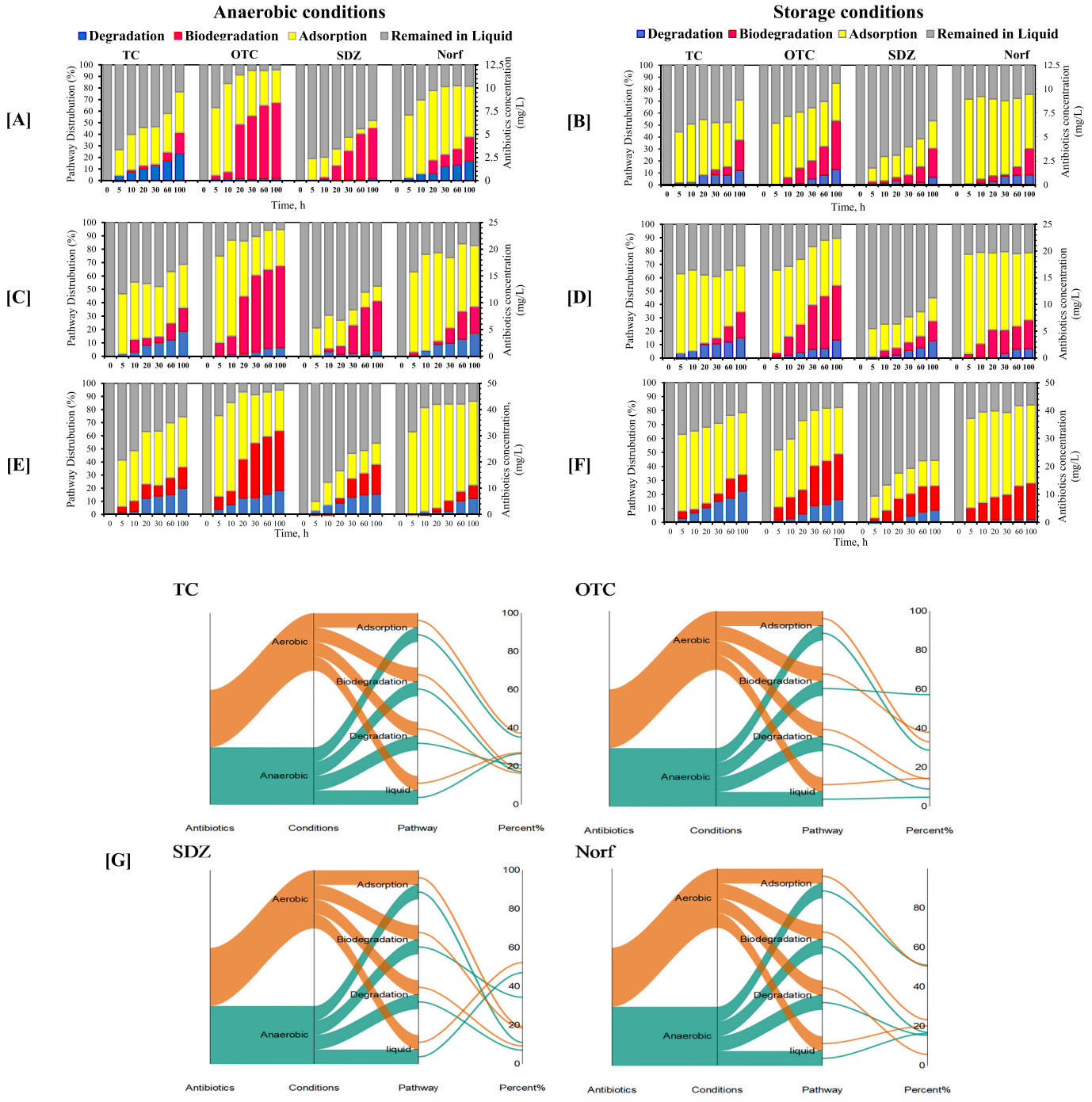

Fig. 1. The removal pathways of the combined antibiotics (TC, OTC, SDZ, and Norf) at different antibiotics concentrations 12.5 mg/l [A], 25 mg/l [C], and 50 mg/l [E] for anaerobic conditions, while 12.5 mg/l [B], 25 mg/l [D], 50 mg/l [F] for storage conditions from DI and swine manure experiments (R1, R2, and R3), and MVAs pathways at the end of running period under storage and anaerobic conditions, and [G] proportion of each pathway (i.e., degradation, biodegradation, and adsorption) in antibiotics fate.

between anaerobic and storage conditions has no influence on TC degradation. Since TCs antibiotics (i.e., TC and OTC) are unstable because of their unique chemical structure, they may undergo abiotic degradation depending on pH, temperature, redox, and light to generate degradation products via epimerization, dehydration or other pathways [35,36]. The variation of those factors may result in this small variation on TCs degradation between anaerobic and storage conditions. Also, it is reported previously by [14] that TC has high sorption potential to reactor walls that may contribute on TC degradation compared to co-antibiotics. Furthermore, TC's degradation was not affected ($p < 0.05$) by its initial concentration. Also, it was found that TC's degradation remained unaffected at incremented initial concentration under anaerobic conditions, while degradation increases under storage conditions at higher concentrations. TC can be degraded considerably in an anaerobic environment such as deep water and anaerobic systems, while it's degradation lowered under storage conditions (i.e., surface waters). Similar results were depicted by [14]. As previously reported by [37], TC could be degraded to different intermediates such as ETC (4-epitetracycline), ATC (anhydrotetracycline), and EATC (4-epianhydrotetracycline). Also, OTC can be degraded into less toxic intermediate such as 2-acetyl-2-decarboxamido-oxytetracycline.

Additionally, it was found that the degradation percentage of OTC under anaerobic conditions increased from 2.4 % to 18.2 % with an increase in initial antibiotic concentration from 12.5 mg/l to 50 mg/l (Fig. 1e), shows a significant ($p < 0.05$) impact on its degradation. Likewise, it was observed that OTC could be degraded by 14.3 % under storage conditions compared to 9.0 % under anaerobic conditions within 100 h (Fig. 1d). However, there is no significance on OTC's degradation is existed induced the temperature difference. Thus, this degradation variance between both processes could be due to instability of OTC structure to undergo several abiotic degradation pathways [36]. A similar degradation trend was observed for SDZ with an increase in degradation percentage from 1.5 % to 15.5 % when VAs concentrations raised from 12.5 mg/l to 50 mg/l under anaerobic conditions. According to the statistical analysis, initial concentration of SDZ showed a significant ($p < 0.05$) impact on SDZ's degradation. [38] reported that sulfonamides antibiotics (i.e., SDZ) did not breakdown via hydrolysis (degradation) under normal environmental conditions, whereas [20] observed that <5 % of sulfamethoxazole was degraded within 56 days. [39] reported that abiotic and biotic degradation of sulfonamides antibiotics were negligible. However, under temperature conditions of the current study, 15.5 % and 8.8 % of SDZ degraded within 100 h under anaerobic and storage conditions, respectively at 50 mg/l (Fig. 1g). This difference is considered relatively low and indicates no influence of temperature difference on SDZ degradation. It means that SDZ's functional groups such as piperazine ring (N—H) and amino group (NH_2) associated with other mechanisms such as H-bonding ($-SO^{2-}$, pyrimidine N; (anilinic N, sulfonamidic N) that might be degraded into smaller, less active or inactive compounds forming carboxyl groups or transformed to intermediates such as 2-aminopyrimidine and 4-hydroxy-2-aminopyrimidine under this study conditions [40].

On the other hand, a significant higher degradation of Norf by 15.5 % was observed under anaerobic conditions compared to storage conditions by 5.8 % within 100 h. This could be due to the higher temperature effect, since temperature difference between anaerobic and storage showed a significant ($p < 0.05$) impact on Norf's degradation. Similarly, initial concentration affects Norf degradation significantly ($p < 0.05$) under anaerobic and storage conditions, Fig. 1. Overall, MVAs could be degraded in DI significantly under anaerobic condition by an average of 11.2 to 16.5 % and by 10.1 to 12.4 % under storage condition within 100 h, when initial concentration increased from 12.5 m/l to 50 m/l. These degradation rates were driven the MVAs natural breakdown under regular operation conditions without the microbial activity role. Similar results were observed by [29] who observed that combined chlortetracycline and OTC were degraded much higher in control assays (without solids) in a very short period of time than those determined from assays including inoculum and manure substrate. [41] reported that OTC and TC may be degraded under surface discharge typically have temperatures range from 0 to 35 °C and pH ranges from 6 to 8.5, those conditions commonly used in surface water, animal wastes treatment, wastewaters, and ground water. Moreover, abiotic degradation was observed in swine manure microcosm under anaerobic conditions with half-life of 15 days [42].

3.1.2. MVAs biodegradation in swine sludge

Mixed antibiotics biodegradation induced by microbial activity under anaerobic and storage conditions was illustrated in Fig. 1. The outcomes from experimental runs depicted that biodegradation of MVAs has improved significantly with the increasing the incubation time, temperature difference has significant ($p < 0.05$) impact on OTC, SDZ, and Norf, while initial concentration has significant impact only on TC. Based on the removal averages, anaerobic conditions led to higher biodegradation of MVAs than with storage conditions. Notably, biodegradation of MVAs was stopped partially in the first 10, 5, 10, and 20 h for TC, OTC, SDZ, and Norf, respectively, under anaerobic and for 30, 10, 5, and 10 h, respectively under storage conditions, (Fig. 1a). These delay in biodegradation could be due to lack of abilities from microorganisms to degrade MVAs induced by generated microbial activity inhibition. Also, due to stress on microbial activity generated by MVAs, microorganisms may prioritize the degradation of one antibiotic over another. However, the increase of biodegradation after these delays could be explained as the microbial activity start acclimatizing to the generated inhibition. The overall biodegradation of MVAs declined from an average of 36.9 % to 23.8 % and from 27.9 % to 22.1 % under anaerobic and storage conditions, respectively, when the initial level increased from 12.5 mg/l to 50 mg/l.

Among studied MVAs, the biodegradation of OTC was the highest, with an average of 57.3 % and 38.1 % under anaerobic and storage conditions, respectively (6.34 and 2.7 times more than degradation in DI). OTC is known for its high biodegradation under such conditions, which is consistent with [19,28]. While TC has the lowest biodegradation with 17.3 % within 100 h. This lower biodegradation of TC resulted from being TC hard to be degraded. Similar results were reported by [21], who found that TC degradation in DI was higher than its biodegradation in swine manure, while biodegradation of OTC was higher in swine manure than its degradation in DI. [4,43] observed that no biodegradation of TC occurred under storage conditions. Moreover, both TC and OTC could be degraded through the breakdown of the parent compounds or by hydroxylation, acetylation of the amino group (OTC), and loss of *N*-methyl group by demethylation of the dimethyl amino group at C4 position (i.e., TC) (Gaballah el., 2023a). On the other hand, the biodegradation of SDZ was significantly ($P < 0.05$) higher under anaerobic conditions compared with storage conditions at varying initial concentrations. This probably could be due to higher removal performance of co-antibiotics under anaerobic conditions that made less stress on microorganisms who consuming SDZ as sole carbon source and energy source and/or via co-metabolism. These findings agree with [44], who observed that the removal of SDZ was mainly attributed to the anaerobic microorganisms than storage microorganisms. SDZ could be degraded by an average of 34.5 % and 18.8 % under anaerobic and storage conditions, respectively, within 100 h, higher at 12.5 mg/l and lower at 50 mg/l, respectively. SDZ at low initial concentration was significantly removed while showing more recalcitrant at high initial concentration. SDZ is known for its low degradability during swine manure management practices. This degradability could be governed by acetylation mechanism of the amino group or by the cleavage of the piperazine ring (N—H) forming carboxyl groups to transfer SDZ into small intermediates such as 2-aminopyrimidine and 4-hydroxy-2-aminopyrimidine. Similarly, the biodegrading of Norf under anaerobic conditions ranged from 26.1 % to 21.7 %, with an increment in the concentration ranged from 12.5 mg/l to 50 mg/l. Compared to co-antibiotics, the lower biodegradation of Norf might be due to its

strong adsorption affinity to sludge and low desorption rates [45]. Norf is very resistant to degradation in biological systems because of its unique physicochemical characteristics, electronegativity, and high stability of C—F bonds and hard degradation of piperazine ring (H—N) group that breakdown mainly through a cleavage mechanism to other intermediates such as (*m/z* 318 and m/z 336) and Norf-C ($C_{14}H_{17}FN_3O$) [36].

Furthermore, each antibiotic showed a significant ($P < 0.05$) difference in biodegradation performance under the anaerobic and storage processes. For OTC and SDZ, the biodegradation pathway was predominant while it did less contribution for Norf and TC. This should be accredited to the distinctive functional groups of each antibiotic, which influence its degradation and interactions with microbes [46]. For example, OTC and SDZ are frequently associated with the electron-donating of (hydroxyl and primary amine) functional groups that are easily lost or assimilated by microorganisms, but this is not available in case of TC and Norf. Overall, the biodegradation performance obtained in this study are somewhat different and relatively lower than the previous studies for individual spikes and lower/higher than other antibiotics mixtures as reported in Table 3. The average biodegradation of TCs was reported to achieve 52 % – 100 % and 45.1 % – 99.2 % for SAs as reviewed by [5]. Another study by [47] has observed the complete removal of TC and sulfamethoxy- diazine (0–50 mg/l) from swine manure after 2–3 days of AD. This may indicate that mixed antibiotics adversely affected the biodegradation of each antibiotic. This could be due to degradation competition that may generate when several antibiotics are mixed, since microorganisms may preferentially degrade one antibiotic over another [27]. This degradation competition may result in a decline of the overall removal performance of the system that needs a further research for more confirmation. It's worth noting that a relatively low biodegradation of mixed antibiotics would be due to some other reasons such as the observed antibiotic could be probably an intermediate metabolite of another antibiotic in the same mixture, this may happen with FQs (i.e., Norf). Moreover, the breakdown of function groups of antibiotics may be inhibited due to co-antibiotics exist. In additions, desorption process of swine sludge releases more antibiotics into wastewater that hampered role of microorganisms in removing mixed antibiotics through biodegradation.

Additionally, the biodegradation of MVAs in swine manure fitted well with the first-order kinetic model, with R^2 values ranging from 0.77 to 0.99, as presented in Table 4. Comparatively, the biodegradation of OTC under anaerobic conditions appeared to be much faster (>50 % degraded in <30 h) than co-antibiotics. This quick removal of OTC could be due to its rapid transformation to a less toxic intermediate 2-acetyl-2-carboxamide-oxytetracycline catalysed by microorganisms [36]. In contrast, TC and SDZ degraded slowly under storage conditions with DT50 values of 216.4–899.9 h and 261.4–414.6 h, respectively. A little higher rate of degradation of TC and SDZ was observed under anaerobic conditions with DT50 values of 283.8–510.5 h and 114.4–245.2 h, respectively. The lower removal efficiency and long DT50 of SDZ were also reported previously. For example, [4] observed around 23.9–33.5 % of spiked SDZ in swine manure was degraded with DT50 values of 223.6–533.2 h under anaerobic conditions. Similarly, lower DT50 values (272.9–342.1 h) for Norf were observed under storage conditions contrasted with values (264.8–586.2 h) found under anaerobic conditions. Herein, the biodegradation of MVAs was perceived to be decreased with an increase in their initial concentration from 12.5 mg/l to 50 mg/l.

3.1.3. MVAs adsorption onto swine sludge

The adsorption process of antibiotics in the biological treatments plays an essential role in their removal [36]. In this study, the adsorption performance of MVAs varied with antibiotic type and treatment conditions, i.e., storage and anaerobic. The results showed that the adsorption of MVAs into swine sludge was significantly ($p < 0.05$) affected by incubation time. The adsorption of TC, OTC, and Norf was influenced by temperature differences between storage and anaerobic conditions, while Norf was not. Moreover, the initial concentration exhibited a significant ($p < 0.05$) variance in the adsorption of TC and Norf, while it did not for OTC and SDZ.

TC and OTC were adsorbed onto swine sludge by 34.2 and 61.6 %, respectively, under anaerobic conditions. While under storage conditions, 52.3 % and 51.4 % of TC and OTC were adsorbed in the first 5 h of contact time (Fig. 1). However, with an increase in contact time to 100 h, the adsorption of TC was almost remained to close at 35.3 % against a decline in OTC adsorption to 28.9 % under anaerobic conditions. These results indicated that TC cannot be desorbed under study conditions while OTC was desorbed easily. This behaviour can be explained as being TC was hard degraded as mentioned in previous section, while OTC was degraded sufficiently. It was also corroborated that the adsorption performance for both TC and OTC was optimal at an initial concentration of 25 mg/l, followed by 50 mg/l, while lowered at 12.5 mg/l. This could be due to that both TC and OTC were degraded less at higher initial concentrations compared to 12.5 mg/l. However, TCs compounds are known for their high adsorption affinity to solids, however, TC and OTC adsorption behaviour in this study was not satisfactory. Individual adsorption behaviour of TC and OTC was >90 % in the first 30 min of contact with sludge according to [4,43]. Thus, it is suggested that the adsorption performance of both TC and OTC could probably influenced by co-antibiotics.

Among the MVAs in this study, SDZ showed the lowest adsorption into solids, with reductions of 16.1 % and 18.1 % under anaerobic and

Table 3
Individual and combined antibiotics removal performances in literature.

Antibiotics	Concentration (mg/l)	Removal pathway	Removal	HRT (days)	Substrate	Running conditions	References
TC	1024	Degradation	64	39	Cattle manure	AD	[32]
OTC	1024	Degradation	57	39	Cattle manure	AD	[32]
Norf	100	Adsorption	80	24	sewage sludge	AD	[33]
TC + OTC	100	Degradation	90, 69.1	30	Swine manure	AD	[29]
CIP	5	Degradation	20.9				
SMX	5	Degradation	81.28	90	Cattle manure	AD	[34]
CIP + SMX	5 + 5	Degradation	6.46; 17.42				
OTC	45	Degradation	82.3				
TC		Degradation	79.7	250 h	synthetic swine wastewater	Biodegradability test	[14]
TYL		Degradation	69.6				
OTC + TC + TYL	45 + 45 + 45 + 45	Degradation	51.2;47.9;67.8				
SDZ	200	Degradation	19	70	synthetic wastewater	AD	[16]
RX + SDZ	60,100	Degradation	62.3;24				
TC + OTC + SDZ + Norf	50	Adsorption competition	70.9;87.5;41.4;55.2	40	Swine manure	AD	[19]

Note: Tetracycline (TC); Oxytetracycline (OTC); Sulfamethoxazole (SMX); Ciprofloxacin (CIP); Norfloxacin (Norf); Sulfadiazine (SDZ); Tylosin (TYL); Roxarsone (RX).

Table 4
Degradation rate constants (k1) and half-lives (t1/2) of the CVAs in both anaerobic and aerobic digestion, Spearman corr.

Antibiotic	Initial concentration (mg/l) (C_0)	Equation	$K_1(h^{-1})$		R^2		DT50 (h)	
			Anaerobic	Storage	Anaerobic	Storage	Anaerobic	Storage
	12.5		0.00244	0.00320	0.82	0.77	283.8	216.4
TC	25.0		0.00205	0.00267	0.77	0.94	338.2	260.4
	50.0		0.00136	0.00077	0.94	0.77	510.5	899.9
	12.5		0.01071	0.00550	0.99	0.99	64.74	126.1
OTC	25.0		0.00824	0.00520	0.99	0.99	78.43	132.9
	50.0	$C_t = C_o * e^{-k1*t}$	0.00534	0.00320	0.99	0.99	129.8	222.2
	12.5		0.00606	0.00265	0.99	0.94	114.7	261.4
SDZ	25.0		0.00495	0.00179	0.99	0.99	140.1	387.1
	50.0		0.00283	0.00167	0.94	0.94	245.2	414.6
	12.5		0.00262	0.00254	0.89	0.94	264.8	272.9
Norf	25.0		0.00203	0.00224	0.90	0.77	341.4	309.8
	50.0		0.00118	0.00203	0.86	0.99	586.2	342.1

storage conditions, respectively, during 100 h of contact time. These results may indicate that adsorption was not the dominant pathway for the removal of SDZ in swine manure treatment. Lower adsorption of SDZ might be attributed to its low N-octanol-water distribution coefficients (log K_{ow}: 0.09) with high water solubility [48–50]. Furthermore, electrostatic repulsion by the negative charge surface of sludge might also be responsible for the lower adsorption of SDZ [4]. In contrast, Norf has low (log K_{ow}: 0.46) and high water solubility, its adsorption onto sludge was caused by both hydrophobic partitioning and hydrophobicity-independent mechanisms (e.g., electrostatic interactions) [36]. These mechanisms are closely linked to the sludge properties that maximized when pH ranged between 6.0 and 8.0, which made the adsorption as the primary removal pathway of Norf during biological wastewater treatment. As shown in Fig. 1, about 58.9 % and 79.3 % of spiked Norf were adsorbed under anaerobic and storage conditions within the first 5 h, respectively. This adsorption performance is much higher than it was for other mixed antibiotics in the current study. While within 100 h, Norf's adsorption was declined to 51.1 % and 53.4 %; respectively under anaerobic and storage conditions. This declining of Norf's adsorption might be due to improvement in other removal pathways, which were higher under storage conditions compared to anaerobic conditions.

Additionally, the adsorption pathway participated in removing around 28.4 to 37.3 % and 33.3 to 37.9 % of MVAs within 100 h of contact with swine manure sludge, under anaerobic and storage conditions; respectively, when initial concentration increased from 12.5 m/l to 50 m/l, Fig. 1g. Moreover, it was found that storage condition was favourable for MVAs adsorption onto swine sludge than anaerobic condition. This difference could be attributed to higher degradation and biodegradation performances under anaerobic condition compared to storage condition. Interestingly, MVAs showed low adsorption compared to Norf in the current study. Norf is known for its high affinity for solids adsorption that may occupy most of the solids' active sites generating an adsorption competition between co-existing compounds. Due to limited active sites for adsorption (TS: 5 %), it is suggested that an adsorption competition was exhibited between MVAs that led to low adsorption performance and then negatively affected MVAs' overall removal under both running conditions. This finding was in agreement with [19,26–28], where they also observed an adsorption competition when antibiotics from different groups were mixed.

3.2. Adsorption of MVAs onto modified bentonite (MB) material

3.2.1. MVAs' adsorption onto MB

MB is an effective adsorbent for removing toxic compounds like antibiotics [51,52]. Bentonite is found to attracting much attention because of its constructional cations, such as Fe^{3+}, K^+, Na^+, and Al^{3+}, which could enrich the microbial activity against toxic compounds during animal residues management practices. Bentonite has distinguished features towards antibiotics adsorption. Recently, bentonite was used as an additives during animal manure treatment practices to enhance the process's performance towards energy production [53], and to improve antibiotics removal during anaerobic digestion [26].

In this study, the adsorption of MVAs onto MB under anaerobic and storage conditions was examined. The kinetics of MVAs adsorption onto MB was likewise studied using 50 mg/l of MVAs with different weights of MB (1, 5, and 10 g/l), as presented in Fig. 2. The results showed that the adsorption of MVAs onto MB was significantly ($p < 0.05$) affected by the contact time and also influenced by the initial MB doses, except for Norf. The temperature difference between the anaerobic and storage processes exhibited a significant variance in the case of OTC and SDZ but showed no variance in the case of TC and Norf. Overall, under both running conditions, Norf had the highest adsorption rates onto MB, while SDZ has the lowest adsorption rates. These results are matched with the adsorption of MVAs onto swine sludge in previous sections. The same set of experiments was conducted without the addition of MB as a control Moreover, the adsorption of TC onto MB increased from 75.9 % to 90.6 % and 75.5 % to 88.6 %, with an increase in the amount of MB from 1 g/L to 10 g/L after 5–10 h under anaerobic and storage conditions; respectively (Fig. 2a and b). After 10 h, TC adsorption onto 1 g/L MB increased slightly, while reached the maximum at 5 and 10 g/L MB. The maximum adsorption of OTC onto MB under anaerobic and storage conditions was around 71 % and 95 %, respectively, at 5 g/L MB. While the lowest adsorption of OTC was observed at 1 g/l MB under anaerobic than storage conditions. Notably, TC and OTC showed low desorption behaviour, which means MB's active sites could retain TCs ions for a long time. Both TC and OTC showed higher adsorption under storage conditions than anaerobic conditions.

The adsorption of SDZ into MB increased from 17.5 % to 24.8 % and 25 % to 28.8 %, with an increase in the amount of MB from 1 g/L to 10 g/L under anaerobic and storage conditions, respectively, Fig. 2. SDZ's adsorption showed a slight increase with time increases at 1 g/l while no significance on its adsorption with MB amount increases. The low adsorption of SDZ could be due to its low water solubility [48,50]. In contrast, Norf showed the highest adsorption to MB among co-antibiotics, averaging from 89.0 % to 95.5 % and 97.7 % to 98 % under anaerobic and storage conditions (Fig. 2a and b). Likewise, the results outcomes depicted that Norf has less desorption behaviour under both running conditions. The amount of MB showed a varied adsorption rates, higher with 5 and 10 g/l and lower at 1 g/l. MVAs adsorption into MB was found to be higher under storage condition compared to anaerobic conditions. This finding is consistent with the adsorption behaviour of MVAs into sludge. However, no significant difference between MB performance at 5 and 10 g/L. A possible explanation is that increasing initial MB may result in unsaturated adsorption surfaces and make an agglomeration phenomenon. Thus, it is suggested that using MB around to 5 g/l is sufficient for attaining optimal adsorption of MVAs. Furthermore, the relatively low desorption performance of MVAs with MB, MB could be recommended for maintaining swine manure

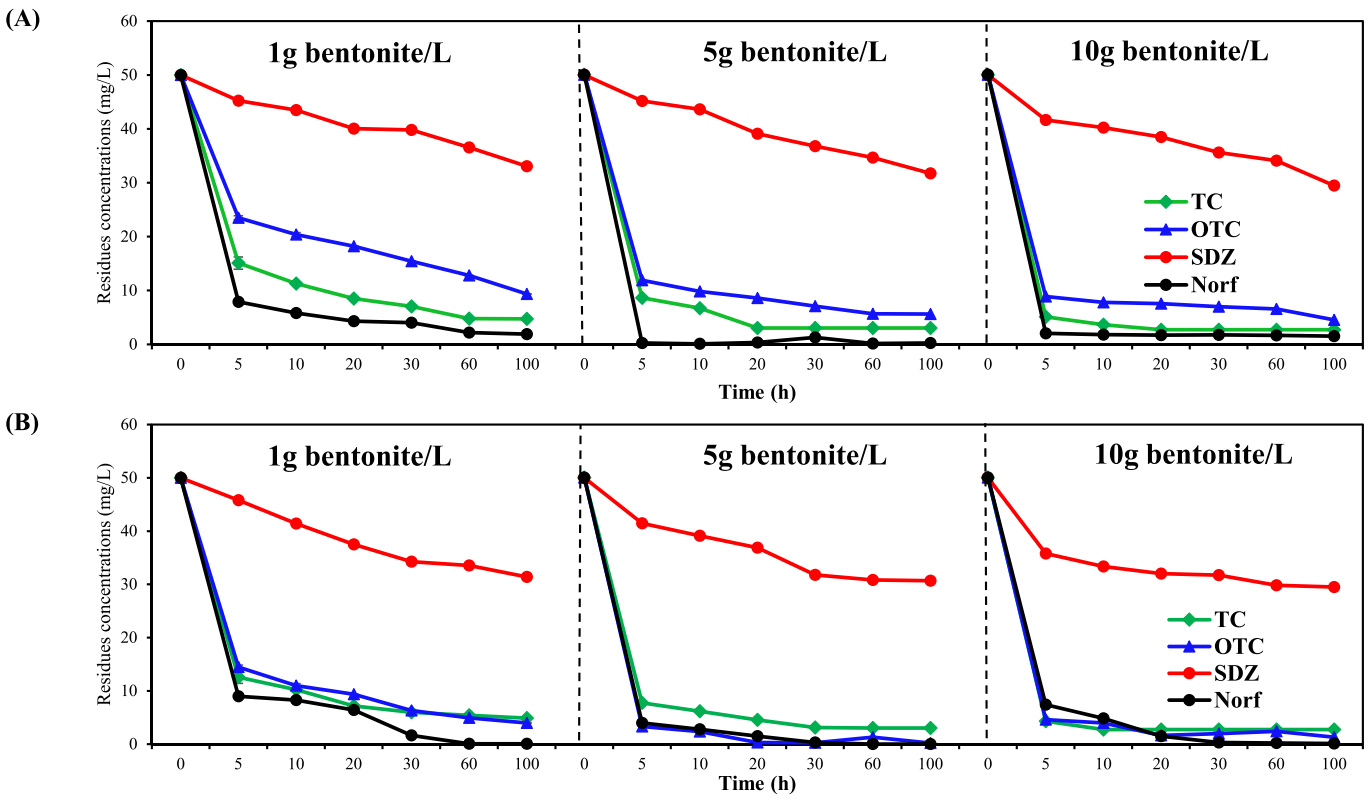

Fig. 2. The residues concentrations of CVAs in the reactors caused by adsorption onto MB with change in amount of modified bentonite under anaerobic [a] and storage conditions [b].

treatment processes under the MVAs exist.

In the current study, pseudo-second-order kinetic model and pseudo-first-order kinetic model were separately applied for the regression of the adsorption process of MVAs onto MB. The experimental results are illustrated in Fig. 3 and Table 5, which fit well to the pseudo-second-order kinetic model with relatively higher correlation coefficients compared to pseudo-first-order kinetic model. Moreover, the theoretical values of q_e calculated from the pseudo-second-order equation model fitted well with experimental q_e values than with pseudo-first-order equation model. Thus, the pseudo-second-order model is more suitable to describe the behaviour of the adsorption process of MVAs onto MB than the pseudo-first-order kinetic model. The outcomes of the current study are in agreement with the results of previous studies [4,43]. These results indicated that the chemisorption would be rate-limited, and the sorption capacity of MB was proportional to the number of active sites on the sorbent. Therefore, the increase in the pseudo-second-order rate constant (k_2) was observed with an increase in the amount of MB. This might be due to the increased availability of adsorption active sites with an increasing amount of adsorbent under both running conditions.

3.2.2. *Microstructure and chemical elements of bentonite*

Raw natural bentonite, modified bentonite (blank), and modified bentonite with spiked antibiotics after 100 h of running conditions (anaerobic and storage) were characterized using SEM and EDS (Fig. 4). The structure of raw bentonite was found to be porous with a rough micro-surface having irregular layer slices (Fig. 4a). This surface structure could potentially increase the contact between adsorbed compounds and bentonite [54]. Thermal modification of raw bentonite was observed to significantly improve the workability of bentonite by immobilizing microorganisms as can be seen clearly in (Fig. 4b). Whereas, the major elements observed in the natural bentonite before modification were oxygen, silicon, aluminum, and sodium (Fig. 4a). Theses contents like oxygen and silicon were slightly changed after thermal modification of raw bentonite (Fig. 4b). This change might probably cause an enchantment in the adsorption behaviour of MB.

After the adsorption of MVAs under storage condition (100*h*), a remarkable change in the surface area of MB was seen (Fig. 4c). It was found that the structure of bentonite in R5T1 was more crumbled and crashed with larger pores and more furrows than the structure of bentonite in raw and MB (Fig. 4a and b). It also observed that these features were increased with R5T2 and R5T3 (Fig. 4e and g). This might be attributed to the antibiotics attachment on the surface of the MB, which is increased with increasing the amount of bentonite. Moreover, the content of oxygen element dropped significantly from 49.6 % with blank to 45.2 %, 44.4 %, and 38.6 % for R5T1, R5T2, and R5T3, respectively. Likewise, silica content was increased from 33.5 % in MB to 38.1 %, 53.8 %, and 44.9 % with R5T1, R5T2, and R5T3, respectively.

Furthermore, it was found that the structure of R6T1, R6T1, and R6T1 were less crumbled and crashed under anaerobic (Fig. 4d, f and h), compared to those under storage condition. This might indicate that MB adsorbed fewer antibiotics under anaerobic conditions than with under storage conditions. This was consistent with MVAs adsorption illustrated in Fig. 2, that showed storage conditions offer higher MVAs' adsorption than under anaerobic conditions.

4. Conclusions

This research sheds light on the diverse removal mechanisms of MVAs during animal manure treatment, emphasizing the importance of understanding their distinct behaviours to develop more environmentally friendly and sustainable treatment practices. Each antibiotic showed a different removal pathway; degradation for TC, biodegradation for OTC and SDZ, and adsorption for Norf. MVAs could be removed by 44.65 ± 14.9 % under anaerobic conditions compared to 36.22 ± 11.2 % under storage conditions. A relatively high and fast adsorption of Norf affected co-antibiotics' adsorption, which stressed the

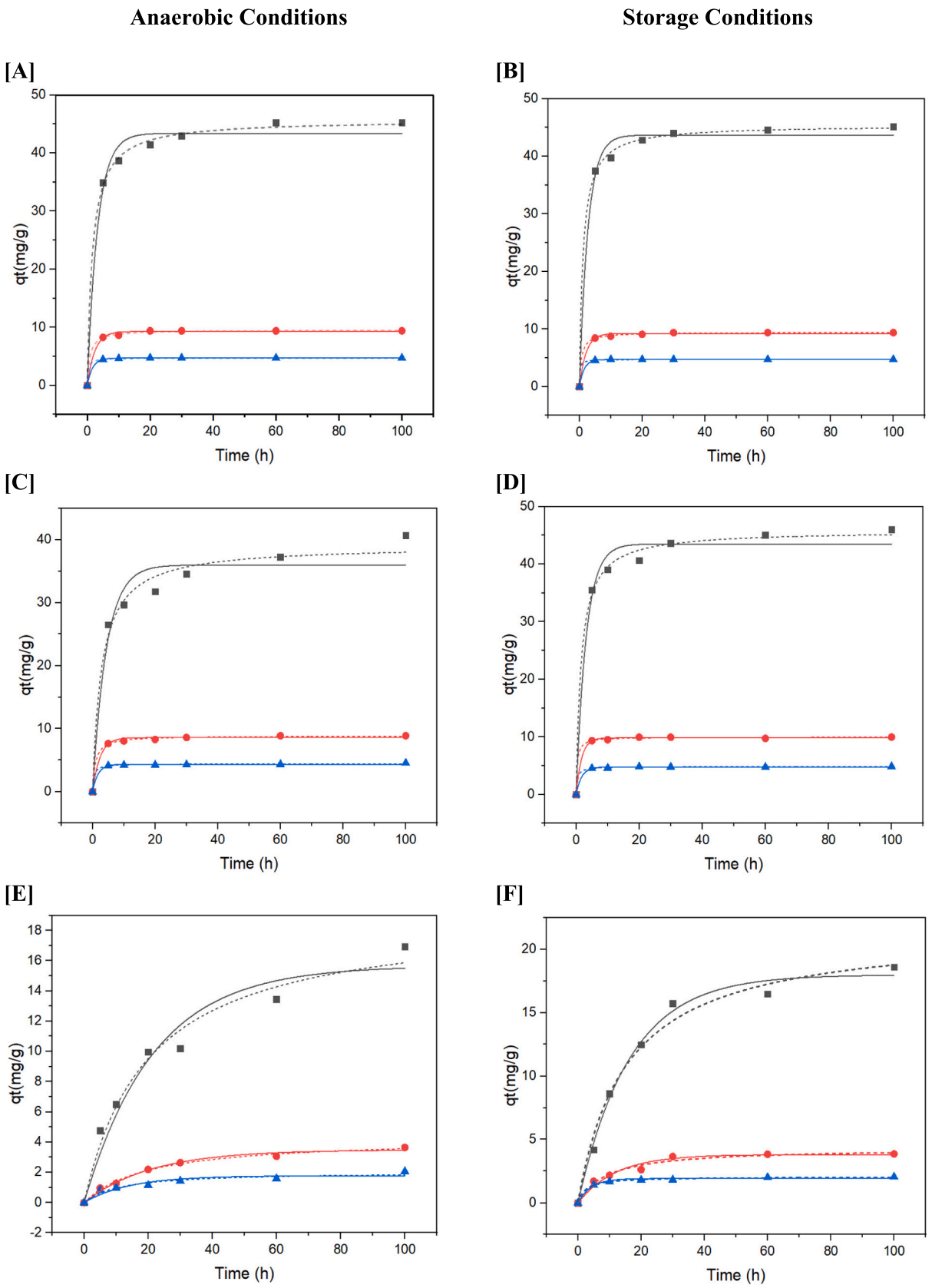

Fig. 3. Adsorption kinetics data and fitted modes of TC [A, B], OTC [C, D], SDZ [E, F], Norf [G, H] under anaerobic and storage conditions, respectively onto different concentrations of MB.

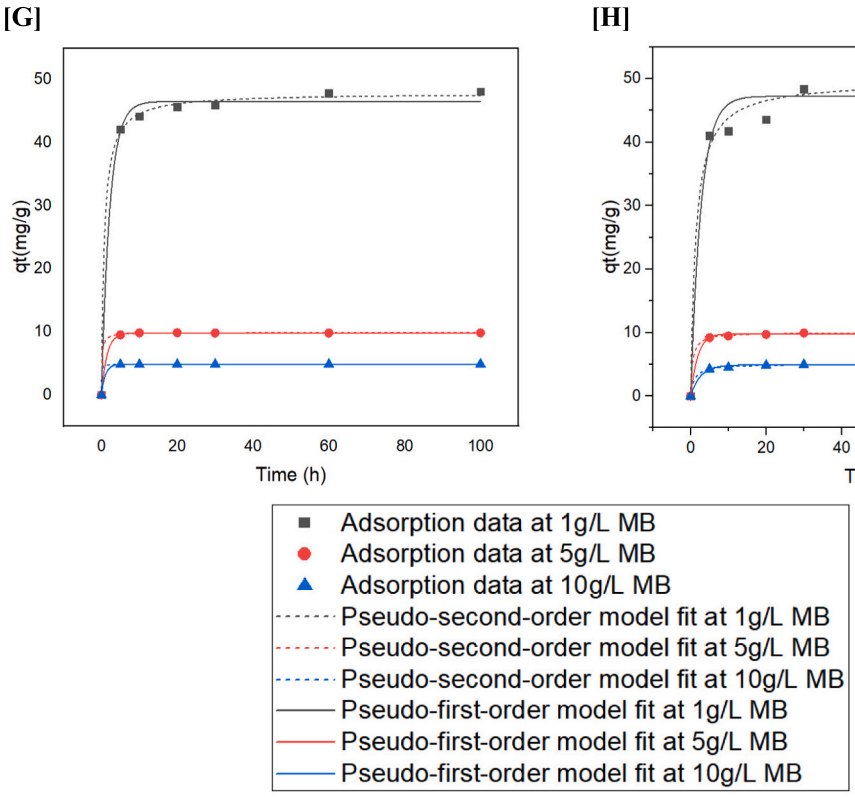

Fig. 3. (*continued*).

Table 5
Kinetic model for the adsorption of combined antibiotics onto modified bentonite.

Kinetic Equation	Equation	Parameter	Reactor	Anaerobic condition				Storage condition			
				TC	OTC	SDZ	Norf	TC	OTC	SDZ	Norf
Pseudo-first-order kinetics	$ln(q_e - q_t) = lnq_e - k_1 t$	q_e	T1	43.39	35.971	15.656	46.529	43.670	43.497	17.983	47.228
			T2	9.298	8.5718	3.4903	9.8237	9.2320	9.8346	3.8100	9.8329
			T3	1.713	4.3346	1.7756	4.8297	4.7270	4.7806	1.9183	4.8978
		K_1	T1	0.299	0.2251	0.0462	0.4538	0.3626	0.3121	0.0613	0.3645
			T2	0.418	0.4196	0.0479	0.6884	0.4776	0.5861	0.0836	0.5351
			T3	0.602	0.5826	0.0704	0.9838	0.6797	0.5821	0.2458	0.3834
		R^2	T1	0.988	0.9528	0.9524	0.9951	0.9923	0.9844	0.9915	0.9738
			T2	0.996	0.9929	0.9859	0.9999	0.9965	0.9984	0.9651	0.9957
			T3	0.999	0.9959	0.8983	0.9999	1.0	0.9979	0.9809	0.9946
Pseudo-second-order kinetics	$\frac{t}{q_t} = \frac{1}{k_2 q_e^2} + \frac{t}{q_e}$	q_e	T1	45.71	39.086	19.032	47.852	45.412	45.829	21.595	49.634
			T2	9.563	8.8623	4.2956	9.8753	9.4594	9.9679	4.3197	10.028
			T3	4.765	4.4117	2.0476	4.8391	4.7545	4.8356	2.0534	5.0670
		K_2	T1	0.013	0.0089	0.0026	0.0283	0.0191	0.0136	0.0030	0.0150
			T2	0.129	0.1259	0.0115	0.6336	0.1654	0.2902	0.0255	0.2033
			T3	0.709	0.5474	0.0436	4.4538	1.2205	0.5493	0.2067	0.198
		R^2	T1	0.999	0.9838	0.9766	0.9990	0.9991	0.9967	0.9862	0.9894
			T2	0.999	0.9986	0.9930	0.9997	0.9994	0.9991	0.9758	0.9995
			T3	0.999	0.9978	0.9469	0.9999	0.9997	0.9991	0.9962	0.9990

biodegradation process and decreased the overall removal performance. This stress caused a longer half-life for antibiotics with relative inhibition impacts. MB showed a high capacity for MVAs adsorption except for SDZ, higher under anaerobic conditions than under storage conditions. Further studies in this direction will help improve our understanding and management of MVAs residues in the environment.

CRediT authorship contribution statement

Mohamed S. Gaballah: Writing – review & editing, Writing – original draft, Validation, Software, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Jianbin Guo:** Writing – review & editing, Visualization, Supervision, Software, Project administration, Funding acquisition. **Atif Muhmood:** Writing – review & editing, Resources, Data curation. **Mostafa Sobhi:** Writing – review & editing, Investigation, Formal analysis, Conceptualization. **Mohamed Ateia:** Writing – review & editing, Resources, Investigation, Formal analysis, Data curation. **Mohamed A. Ghorab:** Writing – review & editing. **Yonghui Zheng:** Visualization, Validation, Methodology, Investigation. **Renjie Dong:** Writing – review & editing, Supervision.

Declaration of competing interest

The authors declare that they have no known competing financial

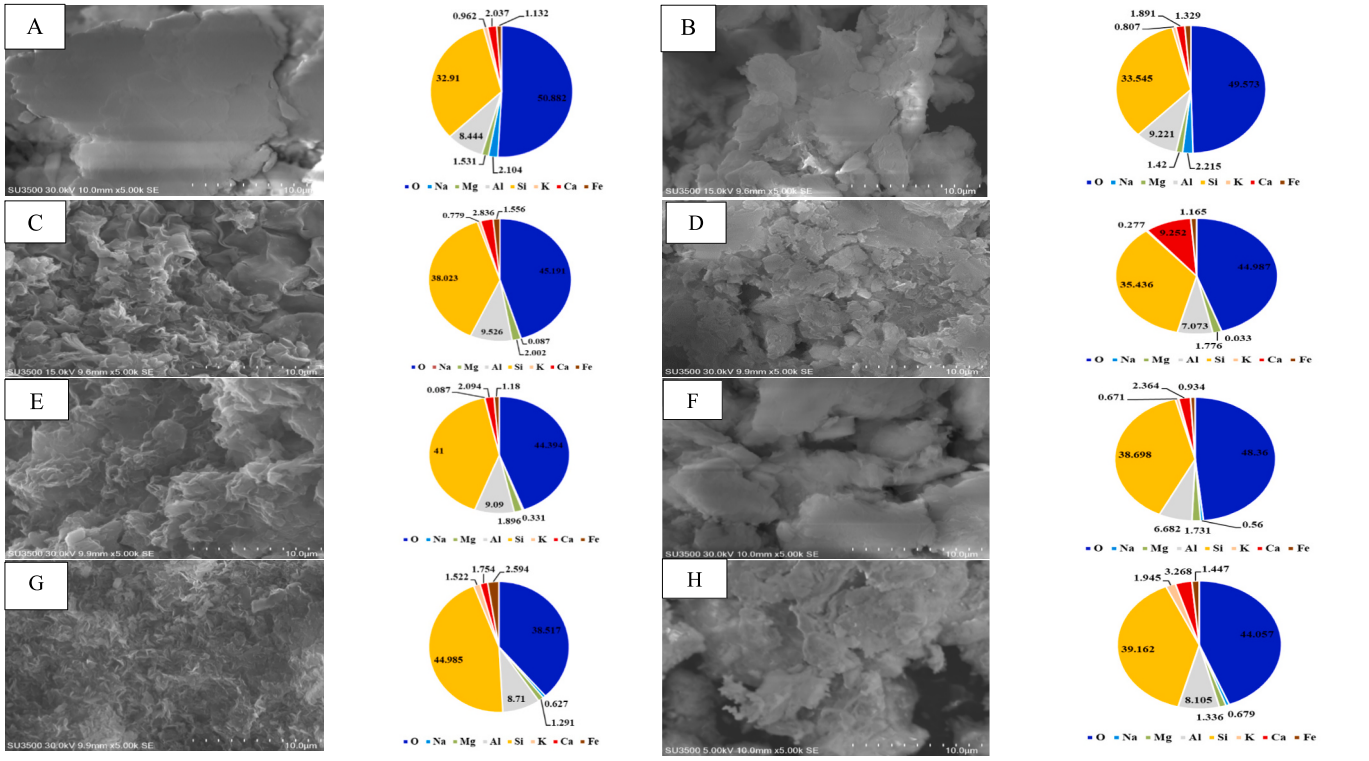

Fig. 4. Surface morphology and elemental contents at 2500* magnification in the micro-area of natural bentonite and bentonite thermally modified [A, B] respectively using SEM and EDS. [C, E, and G] refer to SEM images and EDs spectra of R5 (T1, T2, T3), while [D, F, and H] refer to SEM images and EDs spectra of R6 (T1, T2, T3).

interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

Acknowledgements

The authors thank the National Natural Science Foundation of China for funding their work, and the 2115 Talent Development Program of China Agricultural University for supporting their research.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jwpe.2024.105024.

References

[1] D. Gu, Q. Feng, C. Guo, S. Hou, J. Lv, Y. Zhang, S. Yuan, X. Zhao, Occurrence and risk assessment of antibiotics in manure, soil, wastewater, groundwater from livestock and poultry farms in Xuzhou, China, Bull. Environ. Contam. Toxicol. (2019), https://doi.org/10.1007/s00128-019-02692-0.

[2] L. Feng, M.E. Casas, L.D.M. Ottosen, H.B. Møller, K. Bester, Removal of antibiotics during the anaerobic digestion of pig manure, Sci. Total Environ. 603–604 (2017) 219–225, https://doi.org/10.1016/j.scitotenv.2017.05.280.

[3] FDA, 2018 Summary Report on Antimicrobial Sold or Distributed for Use in Food-Producing Animals. United States Food and Drug Administration (FDA)/Center for Veterinary Medicine (CVM) (Accessed Aug. 16, 2019) 102(4), 2019, pp. 24–25, https://doi.org/10.1051/matecconf/201712107005.

[4] D. Cheng, H.H. Ngo, W. Guo, S.W. Chang, D.D. Nguyen, Y. Liu, X. Shan, L. D. Nghiem, L.N. Nguyen, Removal process of antibiotics during anaerobic treatment of swine wastewater, Bioresour. Technol. 300 (2020) 122707, https://doi.org/10.1016/j.biortech.2019.122707.

[5] M.S. Gaballah, J. Guo, H. Sun, D. Aboagye, M. Sobhi, A. Muhmood, R. Dong, A review targeting veterinary antibiotics removal from livestock manure management systems and future outlook, Bioresour. Technol. 333 (2021) 125069, https://doi.org/10.1016/j.biortech.2021.125069.

[6] F.J. Andriamanohiarisoamanana, I. Ihara, G. Yoshida, K. Umetsu, Kinetic study of oxytetracycline and chlortetracycline inhibition in the anaerobic digestion of dairy manure, Bioresour. Technol. 315 (2020) 123810, https://doi.org/10.1016/j.biortech.2020.123810.

[7] E. Marti, M. Gros, M. Boy-roura, J. Ovejero, A.M. Busquets, J. Colón, M. Petrovic, S. Ponsá, Pharmaceuticals removal in an on-farm pig slurry treatment plant based on solid-liquid separation and nitrification-denitrification systems, Waste Manag. 102 (2020) 412–419, https://doi.org/10.1016/j.wasman.2019.11.001.

[8] F. Yin, H. Dong, W. Zhang, Z. Zhu, B. Shang, Additional function of pasteurisation pretreatment in combination with anaerobic digestion on antibiotic removal, Bioresour. Technol. 122414 (2019), https://doi.org/10.1016/j.biortech.2019.122414.

[9] M. Wohde, S. Berkner, T. Junker, S. Konradi, L. Schwarz, R.A. Düring, Occurrence and transformation of veterinary pharmaceuticals and biocides in manure: a literature review, Environ. Sci. Eur. 28 (2016), https://doi.org/10.1186/s12302-016-0091-8.

[10] L. Zhao, Y.H. Dong, H. Wang, Residues of veterinary antibiotics in manures from feedlot livestock in eight provinces of China, Sci. Total Environ. 408 (2010) 1069–1075, https://doi.org/10.1016/j.scitotenv.2009.11.014.

[11] C. Lee, S. Jeong, M. Ju, J. Young, Fate of chlortetracycline antibiotics during anaerobic degradation of cattle manure, J. Hazard. Mater. 386 (2020) 121894, https://doi.org/10.1016/j.jhazmat.2019.121894.

[12] M. Sobhi, J. Guo, M.S. Gaballah, B. Li, J. Zheng, X. Cui, H. Sun, R. Dong, Selecting the optimal nutrients recovery application for a biogas slurry based on its characteristics and the local environmental conditions: a critical review, Sci. Total Environ. 814 (2022) 152700, https://doi.org/10.1016/j.scitotenv.2021.152700.

[13] M. Sobhi, J. Zheng, B. Li, M.S. Gaballah, D. Aboagye, J. Guo, R. Dong, Carbon footprint of dairy manure management chains in response to nutrient recovery by aerobic pre-treatment, J. Environ. Manage. 328 (2023) 116975, https://doi.org/10.1016/j.jenvman.2022.116975.

[14] L. Delgadillo-Mirquez, P. Gonzalez-Tineo, D. Serrano, U. Durán, Effects of short-term inhibition of Tetracyclines and macrolides on specific methanogenic activity and swine biodegradability, Environ. Technol. Innov. 27 (2022) 1025 74, https://doi.org/10.1016/j.eti.2022.102574.

[15] L. Xiao, Y. Wang, E. Lichtfouse, Z. Li, S. Kumar, J. Liu, D. Feng, Q. Yang, F. Liu, Effect of antibiotics on the microbial efficiency of anerobic digestion of wastewater: a review, Fornt. Microbiol. 11 (2021), https://doi.org/10.3389/fmicb.2020.611613.

[16] R. Tang, S. Yuan, F. Chen, X. Zhan, W. Wang, Z. Hu, Effects of roxarsone and sulfadiazine on biogas production and their degradation during anaerobic digestion, Int. Biodeterior. Biodegrad. 140 (2019) 113–118, https://doi.org/10.1016/j. ibiod.2019.04.001.

[17] X. Zhou, J. Wang, C. Lu, Q. Liao, F.O. Gudda, W. Ling, Antibiotics in animal manure and manure-based fertilizers: occurrence and ecological risk assess- ment, Chemosphere 255 (2020) 127006, https://doi.org/10.1016/j.chemosphere.2020.127006.
[18] L. Xiao, Y. Wang, E. Lichtfouse, Z. Li, P.S. Kumar, J. Liu, D. Feng, Q. Yang, F. Liu, Effect of antibiotics on the microbial efficiency of anaerobic digestion of wastewater: a review, Front. Microbiol. 11 (2021) 1–11, https://doi.org/10.3389/fmicb.2020.611613.
[19] M.S. Gaballah, J. Guo, A. Hassanein, M. Sobhi, Y. Zheng, M. Philbert, B. Li, R. Hui Sun, E. Dong, Removal performance and inhibitory effects of combined tetracycline, Oxytetracycline, sulfadiazine, and Norfloxacin on anaerobic digestion process treating swine manure, Sci. Total Environ. 857 (2023) 159536, https://doi.org/10.1016/j.scitotenv.2022.159536.
[20] J. Li, M. Cui, Kinetic study on the sorption and degradation of antibiotics in the estuarine water: an evaluation based on single and multiple reactions, Environ. Sci. Pollut. Res. 27 (2020) 42104–42114, https://doi.org/10.1007/s11356-020-10194-4.
[21] J. Kasumba, K. Appala, G.E. Agga, J.H. Loughrin, E.D. Conte, Anaerobic digestion oflivestock and poultry manures spiked with tetracycline antibiotics, J. Environ. Sci. Heal. Part B. 55 (2019) 135–147.
[22] A.P. Fagundes, A. Felipe Viana da Silva, B. Bueno de Morais, D. Lusitâneo Pier Macuvele, J. Nones, H. Gracher Riella, N. Padoin, C. Soares, A novel application of bentonite modified with copper ions in the tetracycline adsorption: an experimental design study, Mater. Lett. 291 (2021) 129552, https://doi.org/10.1016/j.matlet.2021.129552.
[23] N. Genç, E.C. Dogan, M. Yurtsever, Bentonite for ciprofloxacin removal from aqueous solution, Water Sci. Technol. 68 (2013) 848–855, https://doi.org/10.2166/wst. 2013.313.
[24] E.G. Ozbayram, O. Arikan, B. Ince, Z. Cetecioglu, S. Aydin, O. Ince, Acute effects of various antibiotic combinations on acetoclastic methanogenic activity, Environ. Sci. Pollut. Res. 22 (2015) 6230–6235, https://doi.org/10.1007/s11356-014-3841-4.
[25] F. Yin, H. Dong, W. Zhang, Z. Zhu, B. Shang, Y. Wang, Removal of combined antibiotic (florfenicol, tylosin and tilmicosin) during anaerobic digestion and their relative effect, Renew. Energy 139 (2019) 895–903, https://doi.org/10.1016/j.renene.2019.03.001.
[26] M.S. Gaballah, J. Guo, A. Hassanein, M. Sobhi, M. Philbert, Y. Zheng, R. Dong, Improving removal of combined veterinary antibiotics and mitigating their negative impacts during anaerobic digestion of swine manure using modified bentonite, J. Water Process Eng. 53 (2023) 103877, https://doi.org/10.1016/j.jwpe.2023.103877.
[27] M.S. Gaballah, H. Chand, J. Guo, C. Zhang, Mixed veterinary antibiotics removal and effects on anaerobic digestion of animal wastes: current practices and future perspectives, Chem. Eng. J. 483 (2024) 149313.
[28] M.S. Gaballah, M. Sobhi, J. Guo, M. Philbert, L. Min, Y. Zheng, M.A. Ghorab, R. Dong, Evaluating the removal and impacts of combined tetracycline, oxytetracycline, sulfadiazine, and norfloxacin during swine manure storage process at ambient temperatures, J. Environ. Chem. Eng. 11 (2023) 109765, https://doi.org/10.1016/j.jece.2023.109765.
[29] J.A. Álvarez, L. Otero, J.M. Lema, F. Omil, The effect and fate of antibiotics during the anaerobic digestion of pig manure, Bioresour. Technol. 101 (2010) 8581–8586, https://doi.org/10.1016/j.biortech.2010.06.075.
[30] J. Ma, M. Amjad Bashir, J. Pan, L. Qiu, H. Liu, L. Zhai, A. Rehim, Enhancing performance and stability of anaerobic digestion of chicken manure using thermally modified bentonite, J. Clean. Prod. 183 (2018) 11–19, https://doi.org/10.1016/j. jclepro.2018.02.121.
[31] M.S. Gaballah, X. Li, Z. Zhang, A. Al-Anazi, H. Sun, M. Sobhi, M. Philbert, M. A. Ghorab, J. Guo, R. Dong, Determination of tetracycline, oxytetracycline, sulfadiazine, norfloxacin, and enrofloxacin in swine manure using a coupled method of on-line solid-phase extraction with the UHPLC-DAD, Antibiotics 10 (2021), https://doi.org/10.3390/antibiotics10111397.
[32] I. Koniuszewska, M. Harnisz, E. Korzeniewska, M. Czatzkowska, J.P. Jastrzębski, L. Paukszto, S. Bajkacz, E. Felis, P. Rusanowska, The effect of antibiotics on mesophilic anaerobic digestion process of cattle manure, Energies 14 (2021) 1–19, https://doi.org/10.3390/en14041125.
[33] N. Li, H. Liu, Y. Xue, H. Wang, X. Dai, Partition and fate analysis of fluoroquinolones in sewage sludge during anaerobic digestion with thermal hydrolysis pretreatment, Sci. Total Environ. 581–582 (2017) 715–721, https://doi.org/10.1016/j.scitotenv.2016.12.188.
[34] V.M. Miritana, L. Patrolecco, A. Barra Caracciolo, A. Visca, F. Piccinini, A. Signorini, S. Rosa, P. Grenni, G.L. Garbini, F. Spataro, J. Rauseo, G. Massini, Effects of ciprofloxacin alone or in mixture with sulfamethoxazole on the efficiency of anaerobic digestion and its microbial community, Antibiotics 11 (2022), https://doi.org/10.3390/antibiotics11081111.
[35] X. Wu, Y. Wei, J. Zheng, X. Zhao, W. Zhong, The behavior of tetracyclines and their degradation products during swine manure composting, Bioresour. Technol. 102 (2011) 5924–5931, https://doi.org/10.1016/j.biortech.2011.03.007.
[36] A.S. Oberoi, Y. Jia, H. Zhang, S.K. Khanal, H. Lu, Insights into the fate and removal of antibiotics in engineered biological treatment systems: a critical review, Environ. Sci. Technol. 53 (2019) 7234–7264, https://doi.org/10.1021/acs.est.9b01131.
[37] T. Alvarino, P. Nastold, S. Suarez, F. Omil, P.F.X. Corvini, H. Bouju, Role of biotransformation, sorption and mineralization of 14C-labelled sulfamethoxazole under different redox conditions, Sci. Total Environ. 542 (2016) 706–715, https://doi.org/10.1016/j.scitotenv.2015.10.140.
[38] Z.N. Norvill, A. Shilton, B. Guieysse, Emerging contaminant degradation and removal in algal wastewater treatment ponds: identifying the research gaps, J. Hazard. Mater. 313 (2016) 291–309, https://doi.org/10.1016/j.jhazmat.2016.03.085.
[39] T. Kaeseberg, J. Zhang, S. Schubert, R. Oertel, P. Krebs, Abiotic, biotic and photolytic degradation affinity of 14 antibiotics and one metabolite – batch experiments and a model framework, Environ. Pollut. 241 (2018) 339–350, https://doi.org/10.1016/j.envpol.2018.05.074.
[40] S.I. Mulla, Q. Sun, A. Hu, Y. Wang, M. Ashfaq, S.A.M.A. Shah Eqani, C.P. Yu, Evaluation of sulfadiazine degradation in three newly isolated pure bacterial cultures, PloS One 11 (2016) 1–14, https://doi.org/10.1371/journal.pone.0165013.
[41] K.A. Loftin, C.D. Adams, M. Rolla, M.T.M. Usgs, Effects of ionic strength, temperature, and pH on degradation of selected antibiotics, J. Environ. Qual. 386 (2008) 378–386, https://doi.org/10.2134/jeq2007.0230.
[42] K.A. Loftin, The Effects and Fate of Selected Veterinary Antibiotics in Two Missouri Anaerobic Swine Lagoons, University of Missouri–Rolla, 2006.
[43] M.H. Huang, Y.D. Yang, D.H. Chen, L. Chen, H.D. Guo, Removal mechanism of trace oxytetracycline by aerobic sludge, Process. Saf. Environ. Prot. 90 (2012) 141–146, https://doi.org/10.1016/j.psep.2011.08.008.
[44] L. Feng, M. Escolà, L. Ditlev, M. Ottosen, H. Bjarne, K. Bester, Removal of antibiotics during the anaerobic digestion of pig manure, Sci. Total Environ. 603–604 (2017) 219–225, https://doi.org/10.1016/j.scitotenv.2017.05.280.
[45] F. Yin, H. Dong, W. Zhang, S. Wang, B. Shang, Z. Zhu, Ability of anaerobic digestion to remove antibiotics contained in swine manure, Biosyst. Eng. 212 (2021) 175–184, https://doi.org/10.1016/j.biosystemseng.2021.10.011.
[46] Y. Tian, J. Li, L. Tang, J. Meng, J. Li, Antibiotics removal from piggery waste- water by a novel aerobic-microaerobic system: efficiency and mechanism, Chem. Eng. J. 454 (2023) 140265, https://doi.org/10.1016/j.cej.2022.140265.
[47] J.C. Shi, X.D. Liao, Y.B. Wu, J.B. Liang, Effect of antibiotics on methane arising from anaerobic digestion of pig manure, Anim. Feed Sci. Technol. 166–167 (2011) 457–463, https://doi.org/10.1016/j.anifeedsci.2011.04.033.
[48] E. Müller, W. Schüssler, H. Horn, H. Lemmer, Aerobic biodegradation of the sulfonamide antibiotic sulfamethoxazole by activated sludge applied as co-substrate and sole carbon and nitrogen source, Chemosphere 92 (2013) 969–978, https://doi.org/10.1016/j.chemosphere.2013.02.070.
[49] J. Cha, K.H. Carlson, Biodegradation of veterinary antibiotics in lagoon waters, Process Saf. Environ. Prot. 127 (2019) 306–313, https://doi.org/10.1016/j.psep.2019.04.009.
[50] S.F. Yang, C.F. Lin, C.J. Wu, K.K. Ng, A. Yu-Chen Lin, P.K. Andy Hong, Fate of sulfonamide antibiotics in contact with activated sludge - sorption and biodegradation, Water Res. 46 (2012) 1301–1308, https://doi.org/10.1016/j.watres.2011.12.035.
[51] M.S. Romero-Güiza, J. Vila, J. Mata-Alvarez, J.M. Chimenos, S. Astals, The role of additives on anaerobic digestion: a review, Renew. Sustain. Energy Rev. 58 (2016) 1486–1499, https://doi.org/10.1016/j.rser.2015.12.094.
[52] Y. Abbas, S. Yun, Z. Wang, Y. Zhang, X. Zhang, K. Wang, Recent advances in bio-based carbon materials for anaerobic digestion: a review, Renew. Sustain. Energy Rev. 135 (2021) 110378, https://doi.org/10.1016/j.rser.2020.110378.
[53] J. Ma, J. Pan, Y. Zhang, Z. Yao, J. Yu, J. Luo, R. Shen, M.K. Awasthi, L. Zhao, Alleviating “inhibited steady-state” in anaerobic digestion of poultry manure by bentonite amendment: performance evaluation and microbial mechanism, Bioresour. Technol. 360 (2022) 127519, https://doi.org/10.1016/j.biortech.2022.127519.
[54] P. Wang, X. Wang, X. Chen, L. Ren, Effects of bentonite on antibiotic resistance genes in biogas slurry and residue from thermophilic and mesophilic anaerobic digestion of food waste, Bioresour. Technol. 336 (2021) 125322, https://doi.org/10.1016/j.biortech.2021.125322.