Original Article

WM&R

Waste Management & Research
1–8

DOI: 10.1177/0734242X19879221
journals.sagepub.com/home/wmr
SAGE

Study of food waste degradation in a simulated septic tank

Hongjian Lin[1,2], Yuchuan Wang[1], Leif van Lierop[1], Carlos Zamalloa[1], Casey Furlong[3], Michael Keleman[3] and Bo Hu[1]

Abstract

Septic systems are typically designed to treat domestic wastewater from households without access to centralized facilities. The installation of a food waste disposer (FWD) may increase the discharge of food waste (FW) into the wastewater; therefore, the installation of a FWD is discouraged in households that have a septic system. This study was conducted to determine how a typical dose of FW from a FWD can affect the performance of a septic system in terms of sewage treatment and solids accumulation. A 20-L control tank was compared with an experiment tank to which FW was added, increasing the amount of total suspended solids (TSS) by 31.3% and total chemical oxygen demands by 46.3% for a period of 110 days. Although the influent water quality changed dramatically, the effluent from the experiment tank had a substantially lower percentage increase in water quality parameters compared with the effluent from the control. It was found that in the experiment tank, 75.8% of FW TSS was degraded, whereas only 36.7% of sewage TSS was degraded, and that 18.8% of FW TSS and 44.9% of sewage TSS accumulated in the experiment tank. The addition of FW increased the scum accumulation, even though the dry matter of the scum layer was much less in quantity than the sludge layer. It also increased the lipid content in the sludge. The increase in the scum layer was mainly due to the increase in protein from the addition of the FW. Overall, compared with sewage TSS, FW TSS tends to be more biodegradable, which indicates that the impact on pumping frequency from adding FW will be insignificant.

Keywords

Food waste, septic tank, household wastewater, food waste disposer, sludge, degradation

Received 4th June 2019, accepted 6th September 2019 by Associate Editor Mario Grosso.

Introduction

Food waste (FW) accounts for approximately 60% of municipal solid waste in developing countries and 30% in developed countries (Maalouf and El-Fadel, 2017). FW is currently handled and treated through landfill or incineration as part of municipal solid waste (Walia and Sanders, 2019). It was estimated that over 50% of US households have installed a food waste disposer (FWD) or a garbage disposer/grinder), A FWD may change the way in which FW is treated because a substantial portion of FW can be diverted to wastewater treatment systems, both centralized systems such as wastewater treatment plants, and decentralized systems such as on-site wastewater treatment systems (Davidsson et al., 2017; Iacovidou et al., 2012; Marashlian and El-Fadel, 2005; Yang et al., 2010). Raw FW is ground by the disposer into smaller particle sizes of mostly between one-quarter to one and a quarter centimeters (about 0.01 to 0.5 inches) as specified by ASSE 1008, the American Society of Sanitary Engineering Standard (American Society of Sanitary Engineering, 1986). The ground FW is then flushed with water and transported through plumbing or sewer pipes to treatment facilities.

Previous studies concluded that the installation of a FWD may increase the strength of nutrients in wastewater and, along with this, compound the problem of their treatment (Marashlian and El-Fadel, 2005). An important feature of FW is that it induces a higher proportion of chemical oxygen demand (COD), and the nutrient concentration of FW total suspended solids (TSS) increases compared with the total nitrogen (TN) and total phosphorus (TP) in sewage TSS: 7.5% to 62% for COD, 2% to 60% for TSS, 1.4% to 19% for TN, and 1.2% to 14% for TP. This increased strength was estimated to increase the treatment cost in centralized treatment systems due to additional aeration and nutrient control processes (NYC Department of Environmental Protection, 1997; Thomas, 2011). However, it was also suggested by some other studies that the higher loading of organic carbon would improve performance with regard to the removal of

[1]Department of Bioproducts and Biosystems Engineering, University of Minnesota, USA
[2]College of Biosystems Engineering and Food Science, Zhejiang University, China
[3]Emerson, USA

Corresponding author:
Bo Hu, Department of Bioproducts and Biosystems Engineering, University of Minnesota, 1390 Eckles Avenue, BAE Building, St. Paul, MN 55108-6005, USA.
Email: bhu@umn.edu

biological nutrients and, therefore, reduce the associated cost of methods employed to remove chemical nutrients (Battistoni et al., 2007). FW may increase COD, TSS, TN, and TP in sewage. This possible negative effect has raised concerns in relation to stratifying FW from solid waste disposal in wastewater treatment systems (Guven et al., 2018), and has affected the use of FWDs and their regulation in European countries. For example, the country with the highest FWD installation rate in Europe is the UK. Here, the penetration rate is around 5% because the UK does not have any legal restrictions with regard to the installation of FWDs, whereas their use is much restricted and discouraged in Germany (Iacovidou et al., 2012).

The Environmental Protection Agency (EPA) has estimated that nearly 25% of US households rely on decentralized systems for sewage treatment. These are commonly known as septic systems either at individual or community cluster scales (EPA, 2017). Property development in suburban areas led to difficulties with access to sewer systems and centralized wastewater treatment plants (WWTPs) for new-build houses, and the percentage of septic systems in use remains stable. There are some studies that evaluated the effect of FW on the effluent quality of septic tanks with alternative configurations, for example, an up-flow anaerobic sludge blanket (UASB) septic tank for treating black water, and it was found that the addition of FW led to an increased total of COD concentration in the effluent, but a similar removal efficiency and a substantially increased soluble COD removal efficiency (Kujawa-Roeleveld et al., 2005; Luostarinen and Rintala 2007). On the contrary, to the best knowledge of the authors, no single peer-reviewed empirical study has assessed the impact of FW on the treatment performance of conventional septic tank effluent or sludge accumulation. Only one observation (Crites and Technobanoglous, 1998) was relevant. This stated that due to the installation of a FWD, the septic tank effluent had a slight or no increase in terms of COD (from 345 to 400 mg/L), TSS (from 80 to 85 mg/L), total ammonium (NH4+) (from 40 to 44 mg/L), organic N (from 24 to 31 mg/L), and TP (no change, 16 mg/L) depending on the installation of filtration systems. This study did not monitor the influent strength and it was not known whether the FW compounds and loadings were typical or not.

To determine how a typical dose from an FWD installation can affect the performance of a septic system in terms of treatment and solids accumulation, a bench-scale simulation was designed and conducted in 1-L tanks for six months. The experiment tank was fed with a mixture of sewage and FW and the control tank was fed with sewage only. It was found that FW was substantially better degraded than simulated sludge at a typical septic tank operating temperature (Lin et al., 2017). At a FW loading of a 34.8% increase in COD compared with sewage, no considerable effect of FW was found on tank performance for TP and TN removal. Proportionally, there was more COD from FW than from sewage, and more suspended solids were degraded. Proportionally fewer FW suspended solids compared with sewage suspended solids accumulated in the experiment tanks as a result of better anaerobic biodegradation. The limitations of the aforementioned study, however, were that the experiment used a small number of 1-L bench-scale tanks, the operating mode was untypical, and the experiment tank was fed with food in suspension rather than typical sizes of ground FW. Considering the minimal septic tank size of 1140 L (300 gal.), and the average particle size of ground food of 0.32 cm to 0.64 cm (an eighth to a quarter of an inch), the experimental results may not be representative of actual conditions. To further explore the question of how a FWD affects the operation and performance of a septic system in a more precise way, a pilot-scale study based on 20-L septic tanks was, therefore, conducted.

Materials and methods

Setup and operation of simulated septic tanks

After pretreatment of coarse debris by screening (primary treatment), the sewage was collected from the St. Paul Metro wastewater treatment plant. Because the influent to an individual household septic system can vary dramatically, the sewage primary effluent was used to represent the raw water in the septic system sewage influent. This sewage wastewater has been flowing in the sewer pipes before it reaches the treatment plant and the effluent after primary screening treatment is, typically, very consistent. This raw water, even though it cannot mimic the fluctuation of the influent parameters of the wastewater flowing into a typical septic system, provides a consistent input for the research study. FW was obtained after grinding representative food waste samples procured from an InSinkErator FWD (model 5-84a, SN 15041100454) and the FW recipe had been used in a previous characterization study (Kim et al., 2015). During grinding, about 1 kg of a heterogeneous mixture of FW was diluted by a factor of 6.17 because of flushing the waste via tap water. The simulated septic tanks were made from cuboid polycarbonate containers and modified to a size with a ratio of length to width over 2:1. An influent tank and effluent tank were used to store water samples temporarily for daily collection. The two septic tanks (Figure 1), one the experiment tank, the other the control tank, were fed at scheduled times with designated influent and FW. Both septic tanks were initially subjected to feeding with sewage only, both to adjust the feeding rate and to assess the similarity between the two in terms of water quality; eventually, starting from Day 12, FW was added manually to the experiment tank according to the designated amount and frequency. Based on the assumption that FW makes up about an additional 25%–30% of total COD compared with sewage in the operation of septic tanks (Abu-Orf et al., 2014; Iacovidou et al., 2012), the proportion of FW addition to the experiment tank was designed so that there would be an average 30% increase in total COD (tCOD) compared with sewage. However, the real tCOD increase in the influent was determined as 46%. The tanks were housed in a temperature-controlled incubating room with a temperature of 15^{o}C and a hydraulic retention time (HRT) of 6.86 days. This was

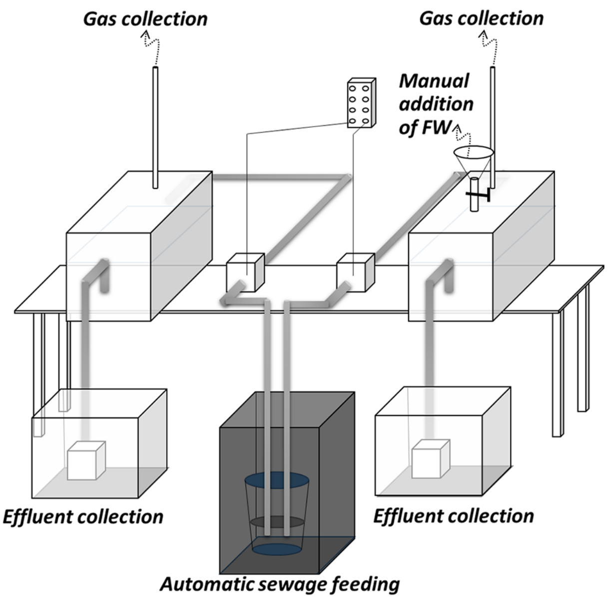

Figure 1. Setup of simulated septic tanks. The control tank was fed with sewage only, whereas the experiment tank was fed with sewage and food waste.
FW: food waste.

much higher than most typical commercial septic tanks, which have a HRT of 1–3 days (Nasr and Mikhaeil, 2013), with smaller tanks typically designed with longer retention times to minimize the effects of turbulence. The operating conditions were the following: HRT, 6.86 days; sewage feeding frequency, 3 times/day; sewage feeding rate, 2.91 L/day; FW feeding frequency, 2 times/week; FW feeding rate, 78 mL/week; temperature of tank operation 15^{o}C; and the ratio of FW addition to the experiment tank to result in a 46% increase in COD. The experiment was conducted over a period of 110 days in 20-L simulated septic tanks.

Water quality analysis of influent, effluent, and mixed liquor, and measurement of sludge accumulation

Influent and effluent samples were collected on a daily basis. FW samples for composition analysis was prepared from the FWD by further blending and dilution at a factor of another 12.75. Water characteristic analyses on tCOD, soluble COD (sCOD), particulate COD (pCOD), TP, TN, and pH were performed using commercial colorimetric methods with a UV-vis spectrophotometer or electrode probes according to American Public Health Association standard methods. TSS were obtained by filtering 20 mL of water sample through 0.45 μm filter paper that was dried overnight at 105°C. After 110 days' operation, the mixed liquor was obtained from the tanks for solids analysis to determine the composition of TSS fed to septic tanks. The height of sludge accumulated in each tank was measured indirectly after the mixed liquor was transferred to volumetric flasks and settled overnight before reading the sludge depth. The scum layer was measured for depth directly, and was collected using a 2 mm hole size (mesh size 10) screen for volume and dry mass quantification. Protein content was obtained by multiplying the difference between the total Kjeldahl nitrogen and total ammoniacal nitrogen content of a dry sample by a factor of 6.25. For crude lipid content determination, dried and ground solids (sludge and scum) samples of about 0.1 g were mixed with a 10 ml mixture (chloroform/methanol at 2:1) and shaken for 16 h in a shaker at 180 rpm. The extraction mixture then had 2.5 ml water added to it, was vortexed for 1 min, and finally centrifuged for 7 min at 5000 xg. Finally, the organic layer was collected and the lipid was harvested from the mixture by filtrating the organic layer through a 0.45 μm filter as filtrate. The filtrate was subjected to a solvent that was evaporated in the oven, and the remaining lipid was weighed.

Results

COD removal

Figure 2 shows the time course profile of tCOD, sCOD, and pCOD, and Table 1 shows the average of influent and effluent properties in the control and experiment tanks. The addition of FW did not cause any obvious acidification effects because the pH in the control tank effluent was 7.96, whereas the value in the experiment tank effluent was 7.85. The influent tCOD, sCOD, and pCOD concentrations were averaged at 599, 118, and 481 mg/L for the control tank, and 876, 194, and 682 mg/L for the experiment tank, respectively (Table 1). Comparatively, tCOD, sCOD, and pCOD concentrations of the effluent were averaged at 130, 68, and 63 mg/L for the control tank, and at 172, 79, and 93 mg/L for the experiment tank, respectively (Table 1). The addition of FW induced increases in the influent tCOD, sCOD, and pCOD concentrations at 46.3%, 64.2%, and 42.0%, respectively. The addition also induced COD concentration increases in the effluent, whereas the respective percentage increases in the effluent were 31.9%, 15.6%, and 48.3%. Therefore, tCOD and sCOD originating from FW were removed at higher degrees than COD originating from sewage. Further analysis indicates that the enhanced removal in the FW was mainly a result of the better removal efficiency of the sCOD portion (86% in FW as compared with 42% in sewage, see Table 1). This confirmed the finding from the 1-L bench-scale experiment (Lin et al., 2017), the results of which indicated that FW was considerably more biodegradable than anaerobic sludge in the simulated septic sludge degradation process, and that a larger portion of FW can be degraded, solubilized, and emitted as methane and carbon dioxide rather than being accumulated in septic tanks. Some studies observed synergistic effects in terms of solid degradation and methane yield by combining FW and sewage sludge/human waste solids in anaerobic digestion (Kim et al., 2017; Yun et al., 2015; Xie et al., 2017), and a similar synergistic effect may also exist in the septic sludge degradation process. The increased COD levels in effluent may increase the biofilm growth on soil particles (biomat), which can be either good or bad. A potentially relevant observation was that the discharge tubing of the experiment tank had more biomass formation inside of it, which may

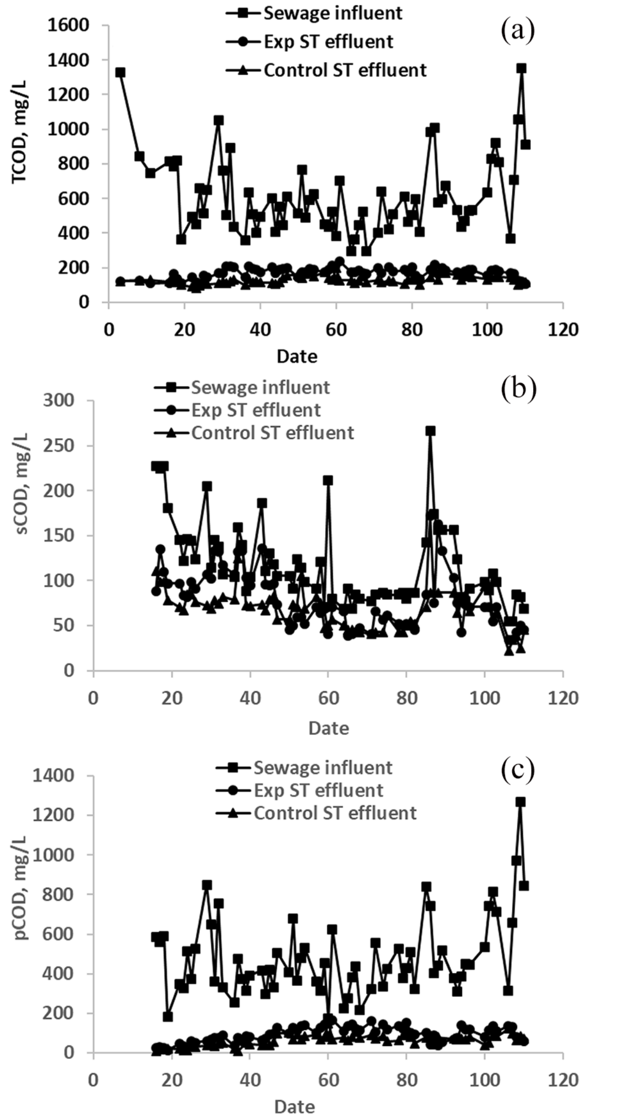

Figure 2. Time course profile of total COD (TCOD) (a), soluble COD (sCOD) (b), and particulate COD (pCOD) (c). The control tank was fed with sewage only, whereas the experiment tank was fed with sewage dosed with food waste. Operated at 15°C with a hydraulic retention time of 6.86 days.

be a result of increased organic content that encourages the biofilm formation.

TSS removal

The average influent and effluent TSS concentrations in the control tank were 378 mg/L and 69 mg/L, respectively, whereas the experiment tank had average influent and effluent TSS concentrations of 498 mg/L and 77 mg/L, respectively. The respective removal efficiencies for the two tanks were 82% and 83% (Figure 3a). The addition of FW to the experiment tank increased its effluent TSS concentration by 12.3%, whereas it increased the influent TSS concentration by 31.6%, compared with the value of the control tank. Further calculation shows that the removal efficiencies of the TSS originating from sewage and FW were 82% and 93%, respectively (Table 1).

Table 1. Influent and effluent properties in the control and experiment septic tanks. The control tank was fed with sewage only, whereas the experiment tank was fed with sewage dosed with food waste. Operated at 15°C with a hydraulic retention time of 6.86 days.

Parameters	Influent				Effluent				Increase in influent	Increase in effluent	% remaining in control tank	% remaining in experiment tank	Calculated remaining % of FW
	Control		Experiment		Control		Experiment						
	Mean	SE	Mean	SE	Mean	SE	Mean	SE					
pH	7.55	0.04	NA	NA	7.96	0.05	7.85	0.04	NA	NA	NA	NA	NA
tCOD, mg/L	599	26	876	56	130	3	172	4	277	42	78	80	85
sCOD, mg/L	118	6	194	10	68	3	79	4	76	11	42	59	86
pCOD, mg/L	481	25	682	60	63	4	93	5	202	30	87	86	85
TSS, mg/L	382	29	501	44	70	5	77	6	119	7	82	85	94
TN, mg/L	54.3	1.1	58.4	1.6	41.1	0.7	42.2	0.7	4.1	1.0	24	28	74
TP, mg/L	7.62	0.23	8.09	0.25	5.16	0.06	5.49	0.07	0.47	0.32	32	32	31

FW: food waste; tCOD: total chemical oxygen demand; sCOD: soluble COD; pCOD: particulate COD; TSS: total suspended solids; TN: total nitrogen; TP: total phosphorus.

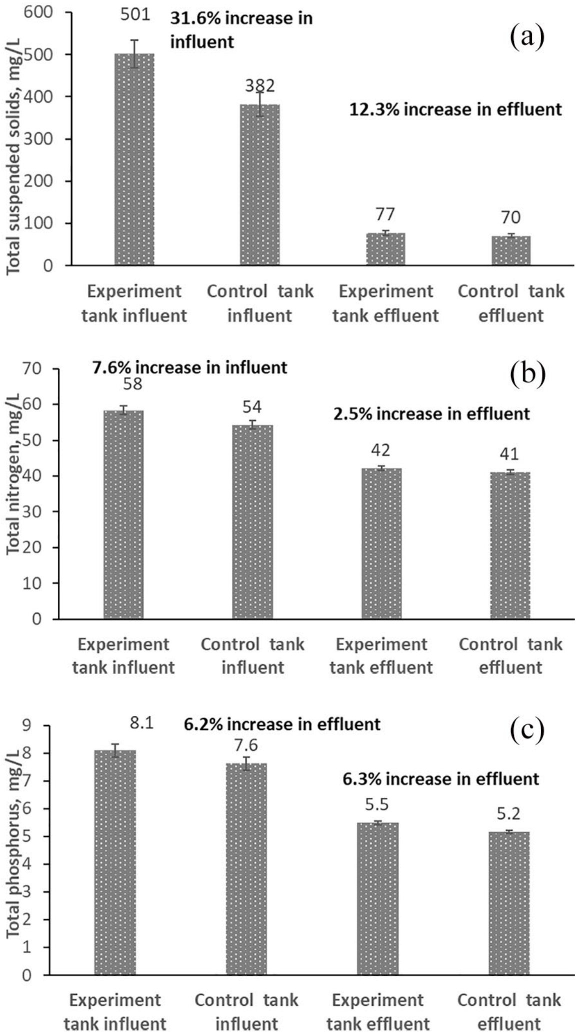

Figure 3. Average concentrations of total suspended solids (a), total nitrogen (b), and total phosphorus (c). The control tank was fed with sewage only, whereas the experiment tank was fed with sewage dosed with food waste. Operated at 15°C with a hydraulic retention time of 6.86 days.

Therefore, the TSS from FW was subjected to a greater level of treatment than the TSS from sewage, either by better degradation or better settling. Because TSS is a critical parameter that dictates the performance of a septic tank, and poor TSS removal can induce hydraulic failure, this result indicates that the addition of FW to the septic tank does not cause significantly poorer tank performance.

TN and TP removal

TN was analyzed from Day 47. The addition of FW induced a 7.6% increase in TN concentration in the influent, and a 2.5% increase in the effluent, comparing the experiment and control tanks (Figure 3b). The increase was minimal, and may not result in any substantial effects on water quality or additional problems with treatment as far as a septic system is concerned. The removal efficiency of TN in both the control and experiment tanks was within the literature data range for conventional septic tanks (Lin et al., 2017; Lowe et al., 2009) and UASB septic tanks (Bogte et al., 1993; Luostarinen and Rintala, 2007) except some UASB septic tanks showing outstanding performance that were fed with black water (Luostarinen and Rintala, 2005). The FW increased TP concentration from 7.6 mg/L to 8.1 mg/L in the influent, and from 5.2 mg/L to 5.5 mg/L in the effluent between the control tank and the experiment tank (Figure 3c). Similar to the case of TN, the increase in effluent TP concentration by 0.3 mg/L between the control tank and the experiment tank was minimal, and may not create additional problems with treatment as far as a septic system is concerned, given the excellent TP removal efficiency of a septic system. After the preliminary treatment in the septic tank, the tank effluent will be further treated in a leach field of a septic system for COD, N, and P before the system effluent enters surface or ground water bodies (Wilhelm et al., 1994). Different from N removal, P removal was more a result of mineralogical processes such as absorption and crystallization (mineral precipitation). The increased P loading in the septic system may require a better P removal capacity. Nevertheless, a field study that evaluated a 20-year-old septic system filter bed showed that P was mainly immobilized in the filter bed within 1 m of tile lines so that no substantial contamination downstream occurred (Robertson, 2012). The examination of sand surfaces revealed that iron and aluminum were abundant and that the P content of sand grains was increasing over time. In the same study, it was found that groundwater P concentration had not increased over six years of monitoring data. Given a filter bed material that mineralizes P, the 0.3 mg/L of TP increase due to the use of a FWD can be well within the treatment capacity for a prolonged period of time.

These results, together with COD and TSS degradation, are significant in determining the use of a FWD in septic systems. Currently, the use of a FWD in a property that has a septic system is sometimes discouraged by various rules and regulations. Septic systems have long been troubled with limited efficiencies with regard to handling domestic wastewater, especially in relation to nutrients and pharmaceuticals (Arrubla et al., 2016; Shahraki et al., 2018). Often, the concern with regard to the use of a FWD is that food waste does not break down in septic tanks and overloads the system. Some exceptions are allowed, but in some cases, a 50% size increase in the septic tanks is required if a FWD is going to be installed. The research results showed that FW TSS was much better degraded than sewage TSS, and additional FW input due to the installation of a FWD actually had very little impact on the overall septic effluent water parameters.

Sludge accumulation and solids balances

Visual inspection, both the top view and side view of the two tanks, suggested that the addition of FW substantially increased scum layer formation. More gas bubbles were present on top of the scum layer of the experiment tank. The depth of the scum layers was 1.91 cm and 0.97 cm for the experiment and control tank, respectively (Figure 4). When collected, the volume of the

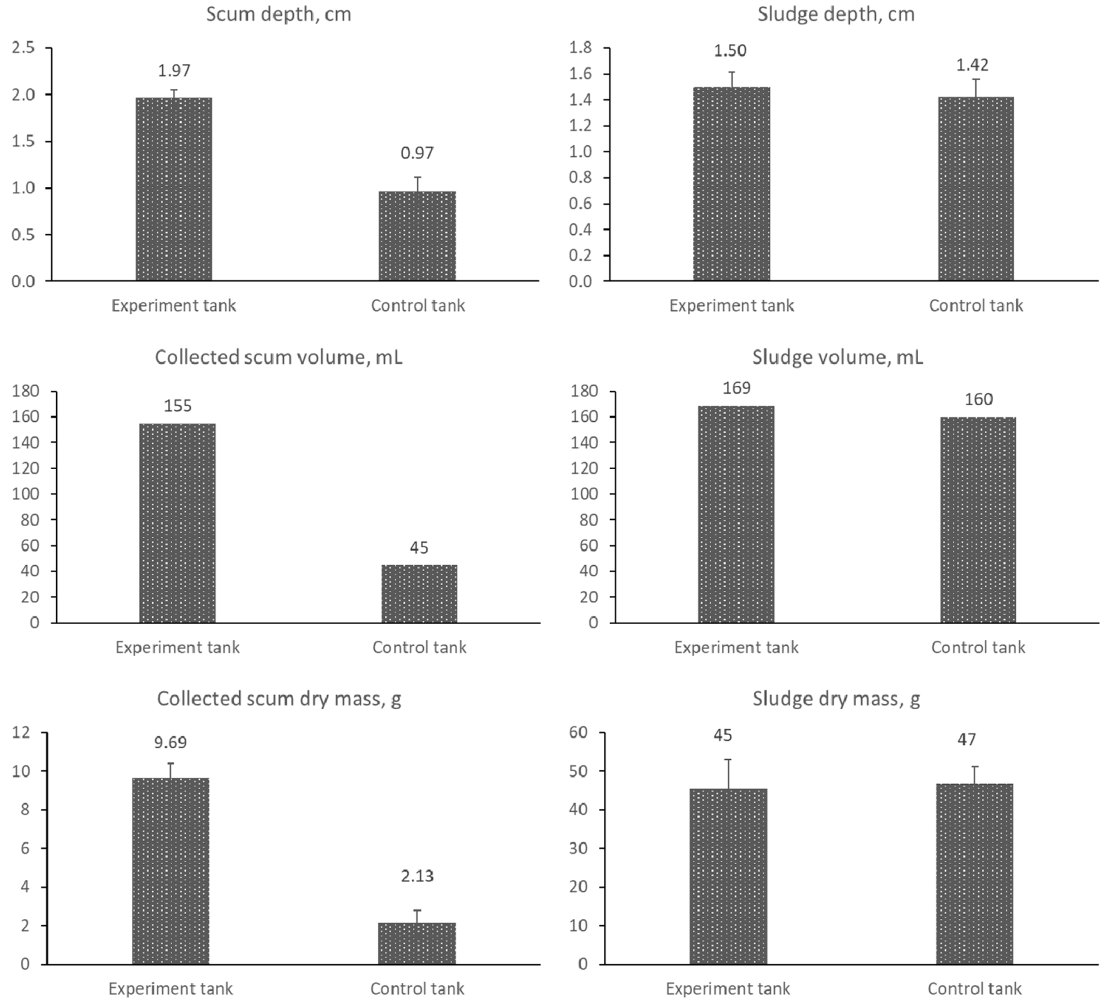

Figure 4. Sludge and scum formation in the experiment and control tanks. The control tank was fed with sewage only, whereas the experiment tank was fed with sewage dosed with food waste. Operated at 15°C with a hydraulic retention time of 6.86 days.

scum was 155 mL and 45 mL, respectively. After being dried, the mass of the scum was 9.69 g and 2.13 g, respectively. In the 1-L bench-scale test, the scum layer was also more obvious in the experiment tank; however, the amount of scum was so small it was not practically recoverable and, therefore, a conclusion had not been reached (Lin et al., 2017). The operating modes of the bench-scale (1-L) study and pilot-scale (20–L) study were different, and the 20-L experiment was more closely aligned to the parameters of a typical septic tank that had a relatively constant liquid level without substantial disruption at the liquid surface.

The two tanks had almost the same sludge depth, volume, and amount (Figure 4). It seems that the effect of additional solids from the FW did not extend beyond the scum layer. Although the mass of the scum layer was small compared with the mass of the sludge, the depth and the volume of the scum and sludge layers were similar because of the loose matrices of the scum layer. The increased scum generated by FW might occupy more storage space in the septic tank, although the layer may just float on the liquid surface. Assuming the input of TSS from sewage and FW was 100%, respectively, the majority of the suspended solids from the sewage accumulated in the septic tank (44.9%), whereas the majority of the suspended solids from FW were solubilized or degraded (75.8%) (Figure 5a). The proportion of FW solids discharged to effluent or accumulated in the tank was similar to the 1-L bench-scale study (Lin et al., 2017). The 1-L bench-scale study did not reveal any solids accumulation in the scum layer because the reactor was too small to form the scum layer; all solids were categorized into the sludge layer. However, this 20-L large-scale study showed significant scum accumulation, especially in the experiment tank, due to the addition of FW.

Apparently, the FW substantially induced the formation of the scum layer, and the composition of the sludge and scum was analyzed to explore the potential reason. The protein content of the scum in the experiment tank was substantially higher than that of the control tank (Figure 5b), indicating that proteins had floated to the surface. However, scum in the control and experiment tanks had a similar lipid content of 12.6% and 14.3%, respectively. Interestingly, the sludge in the experiment tank had a much higher crude lipid content than that of the control tank, 18.0% and 5.3%, respectively (Figure 5c). This contradicted with the conventional theory that lipids float to the scum layer. A possible reason could be that the enhanced microbial activity in the experiment tank

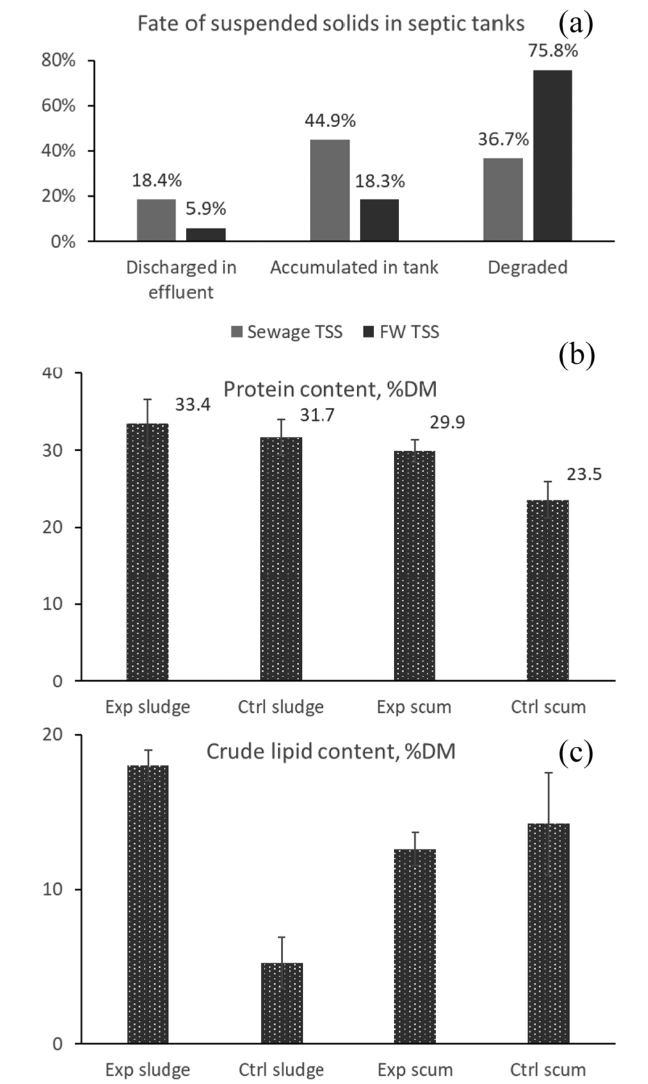

Figure 5. Composition of suspended solids (a), amount of protein (b), and crude lipid (c) in sludge and scum The control tank was fed with sewage only, whereas the experiment tank was fed with sewage dosed with food waste. Operated at 15°C with a hydraulic retention time of 6.86 days.
Exp.: Experiment tank; Ctrl: control tank; TSS: total suspended solids; %DM: weight percentage on a dry matter basis

sludge encourages the degradation of carbohydrates, and the gas ebullition brings a filamentous biomass to the scum layer, therefore creating a scum with a higher protein content and leaving a sludge layer with a higher lipid content.

Conclusions

This study evaluated the effect of FW on septic effluent water quality and solids accumulation in 20-L simulated tanks for a period of 110 days. The treatment performance and solids accumulation were compared between the control tank without FW and the experiment tank with the addition of FW. No acidification was observed at the TSS increase of 31.3% and the tCOD increase of 46.3% when FW was added. Although there was a dramatic increase in influent water quality between the experiment and the control tanks due to the addition of FW, the effluent water quality with regard to TSS, tCOD, sCOD, TN, and TP increased only very slightly in the experiment tank compared with the control. The tCOD increase in the effluent may have an impact on the performance of the septic system, whereas the increases in TN and TP were minimal. It was found that in the experiment tank, 75.8% of FW TSS was degraded, whereas 36.7% of sewage TSS was degraded. As far as the accumulation of TSS was concerned, 18.8% of FW TSS and 44.9% of sewage TSS accumulated in the experiment tank. The FW substantially increased the depth and volume of the scum layer in the experiment tank, although the dry mass of the scum layer was small compared with the sludge layer. The addition of the FW increased the lipid content in the sludge rather than in the scum. The increased amount of scum layer (9.69 g vs. 2.13 g of scum layer in the experiment and control tanks, respectively) is due to the increase in protein from the addition of FW. Overall, compared with sewage TSS, FW TSS tends to be more biodegradable and accumulate more in the scum layer. This better degradation of FW TSS indicates that the impact of the addition of FW on septic performance and then on pumping frequency will be insignificant or negligible.

Declaration of conflicting interests

The authors declared no potential conflicts of interest with respect to the research, authorship, and/or publication of this article.

Funding

The authors disclosed receipt of the following financial support for the research, authorship, and/or publication of this article: This study was funded by Emerson Commercial & Residential Solutions.

ORCID iDs

Hongjian Lin https://orcid.org/0000-0002-1368-5120
Bo Hu https://orcid.org/0000-0001-9786-3336

References

Abu-Orf MG, Tchobanoglous HD, Stensel R, et al. (2014) *Wastewater Engineering: Treatment and Resource Recovery*. Boston, MA: McGraw Hill Education.

American Society of Sanitary Engineering (1986) *Performance Requirements for Household Food Waste Disposer Units*. ASSE Standard 1008. Bay Village, OH: ASSE.

Arrubla JP, Cubillos JA, Ramirez CA, et al. (2016) Pharmaceutical and personal care products in domestic wastewater and their removal in anaerobic treatment systems: Septic tank – up flow anaerobic filter. *Ingenieria E Investigacion* 36: 70–78.

Battistoni P, Fatone F, Passacantando D, et al. (2007) Application of food waste disposers and alternate cycles process in small-decentralized towns: A case study. *Water Research* 41: 893–903.

Bogte J, Breure A, Van Andel J, et al. (1993) Anaerobic treatment of domestic wastewater in small scale UASB reactors. *Water Science and Technology* 27: 75–82.

Crites R and Technobanoglous G (1998) *Small and Decentralized Wastewater Management Systems*. Boston, MA: McGraw-Hill.

Davidsson Å, Saraiva AB, Magnusson N, et al. (2017) Technical evaluation of a tank-connected food waste disposer system for biogas production and nutrient recovery. *Waste Management* 65: 153–158.

EPA (2017) Decentralized systems. Available at: www.epa.gov/small-and-rural-wastewater-systems/learn-about-small-wastewater-systems (accessed 2 September 2019).

Guven H, Eriksson O, Wang Z, et al. (2018) Life cycle assessment of upgrading options of a preliminary wastewater treatment plant including food waste addition. *Water Research* 145: 518–530.

Iacovidou E, Ohandja D-G, Gronow J, et al. (2012) The household use of food waste disposal units as a waste management option: A review. *Critical Reviews in Environmental Science and Technology* 42: 1485–1508.

Kim M, Chowdhury M, Nakhla G, et al. (2015) Characterization of typical household food wastes from disposers: Fractionation of constituents and implications for resource recovery at wastewater treatment. *Bioresource Technology* 183: 61–69.

Kim M, Chowdhury M, Nakhla G, et al. (2017) Synergism of co-digestion of food wastes with municipal wastewater treatment biosolids. *Waste Management* 61: 473–483.

Kujawa-Roeleveld K, Fernandes T, Wiryawan Y, et al. (2005) Performance of UASB septic tank for treatment of concentrated black water within DESAR concept. *Water Science and Technology* 52: 307–313.

Lin H, Liu W, Zhang X, et al. (2017) Microbial electrochemical septic tanks (MESTs): An alternative configuration with improved performance and minimal modifications on conventional septic systems. *Biochemical Engineering Journal* 120: 146–156.

Lin H, Sinchai M, Zamalloa C, et al. (2017) Food waste vs. sewage degradation in septic tanks: Better biodegradability and less sludge accumulation. Paper presented at the NOWRA 2017 Onsite Wastewater Mega-Conference, Dover, DE, USA, 22–25 October 2017.

Lowe KS, Tucholke MB, Tomaras JM, et al. (2009) *Influent Constituent Characteristics of the Modern Waste Stream from Single Sources: Decentralized Systems. Final Report*. Alexandria, VA: Water Environment Research Foundation.

Luostarinen S and Rintala J (2005) Anaerobic on-site treatment of black water and dairy parlour wastewater in UASB-septic tanks at low temperatures. *Water Research* 39: 436–448.

Luostarinen S and Rintala J (2007) Anaerobic on-site treatment of kitchen waste in combination with black water in UASB-septic tanks at low temperatures. *Bioresource Technology* 98: 1734–1740.

Maalouf A and El-Fadel M (2017) Effect of a food waste disposer policy on solid waste and wastewater management with economic implications of environmental externalities. *Waste Management* 69: 455–462.

Marashlian N and El-Fadel M (2005) The effect of food waste disposers on municipal waste and wastewater management. *Waste Management & Research* 23: 20–31.

Nasr FA and Mikhaeil B (2013) Treatment of domestic wastewater using conventional and baffled septic tanks. *Environ Technology* 34: 2337–2343.

NYC Department of Environmental Protection (1997) The impact of food waste disposers in combined sewer areas of New York City. New York: NYC Department of Environmental.

Robertson W (2012) Phosphorus retention in a 20-year-old septic system filter bed. *Journal of Environmental Quality* 41: 1437–1444.

Shahraki ZM, Mao XW, Waugh S, et al. (2018) Impact of legacy nitrogen in conventional septic system on nitrogen removal for onsite wastewater treatment. Poster presented at 255th National Meeting and Exposition of the American Chemical Society - Nexus of Food, Energy, and Water, New Orleans, LA.

Thomas P (2011) The effects of food waste disposers on the wastewater system: a practical study. *Water and Environment Journal* 25: 250–256.

Walia B and Sanders S (2019) Curbing food waste: A review of recent policy and action in the USA. *Renewable Agriculture and Food Systems* 34: 169–177.

Wilhelm SR, Schiff SL and Cherry JA (1994) Biogeochemical evolution of domestic waste water in septic systems: 1. Conceptual model. *Groundwater* 32: 905–916.

Xie S, Wickham R and Long DN (2017) Synergistic effect from anaerobic co-digestion of sewage sludge and organic wastes. *International Biodeterioration & Biodegradation* 116: 191–197.

Yang X, Okashiro T, Kuniyasu K, et al. (2010) Impact of food waste disposers on the generation rate and characteristics of municipal solid waste. *Journal of Material Cycles and Waste Management* 12: 17–24.

Yun YM, Cho SK, Kim HW, et al. (2015) Elucidating a synergistic effect of food waste addition on the enhanced anaerobic digestion of waste activated sludge. *Korean Journal of Chemical Engineering* 32: 1542–1546.