#25 Spring Newsletter Topic

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The characteristics and in-sewer transport potential of solids derived from domestic food waste disposers

Abigail Legge, Andy Nichols, Henriette Jensen, Simon Tait and Richard Ashley

ABSTRACT

This study aims to assess the transportability of food waste disposer particles within a sewer system. A series of laboratory studies has examined the physical characteristics of solid particles derived from domestic food waste disposers. Particle size distributions and maximum settling velocity characteristics were measured for 18 common food types, and stored in a publicly accessible database. Particle size distributions are shown to fit well with a 2-parameter Gamma distribution. Settling velocity is generally higher for larger particles, except when particle density and sphericity change. For most food types, particle specific gravity was close to unity. Egg shell particles had a significantly higher specific gravity. This information, combined with the particle size data have been used to show that there is a very low likelihood of food waste particle deposition in sewers during normal operational flows, other than temporary transient deposits of egg shell particles.

Key words | domestic food waste disposers, food waste, in-sewer deposition, particle fall velocity, sewer solid entrainment thresholds

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HIGHLIGHTS

- Particles characterised for 18 common food types.
- Sizes fit a Gamma distribution, mode of 0.59 to 4.76 mm.
- Most particles had apparent densities close to water.
- Most particles entrained at low boundary shear stress, unlikely to form deposits in sewer pipes.
- Egg shell showed higher entrainment threshold, but still expected to transport during dry weather flows.

INTRODUCTION

There is considerable debate on the best way to manage the disposal of unavoidable domestic food waste, and there is no clear consensus on the optimum approach (e.g. Schanes *et al.* 2018; Slorach *et al.* 2020). In England, the food waste of more than half of households (54%) is still collected with other solid waste by centralised municipal collection and disposal (WRAP 2015). In Europe, Member States are required to encourage householders to separate out their average 173 kg food waste per person per year (Schinkel 2019) for home composting or kerbside collection (EU Amending Waste Framework Directive 2018, Article 22). However, the effectiveness of this approach has been found to be limited to less than 50% of separable food waste (e.g. STOWA 2015). There are also concerns regarding the overall carbon emissions from kerbside collection. In England kerbside collection is seen as the recommended way forward for all domestic food waste by 2023 (Defra Environment Bill 2020), with resource recovery achieved

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primarily via municipal authority street collection trucked to dedicated anaerobic digestion (AD) plants. It is recognised that this approach will require considerable investment in vehicles and digestion plants and may also not provide the minimised carbon emissions compared with other waste management options (e.g. Jenkinson 2020).

In a number of regions around the world, food waste is disposed of by discharging it into the wastewater collection system after processing using domestic food waste disposers (mechanical grinders) that break down food into small particles, e.g. in Surahammar, Sweden (Evans *et al.* 2010). More than 50% of households in the USA have food waste disposers (American Housing Survey 2013), in excess of 34% of households in New Zealand, and 10% in Canada. In the EU, fewer food waste disposers (FWD) are generally in use, with only 5% of households evidenced in the UK (Iacovidou *et al.* 2012). However, there are various initiatives investigating how FWDs can be used to enable householders to separate their food waste at source to enable resource recovery (e.g. Bisschops *et al.* 2019; Run4Life 2020). This shift in domestic food 'waste' as part of wastewater inputs to wastewater treatment plants (WWTPs) becoming seen as a potential resource, has come about due to recent concepts such as the circular economy and the need to better manage carbon (Skambraks *et al.* 2017; van Leeuwen *et al.* 2018; Velenturf *et al.* 2018; Sancho *et al.* 2019).

Although FWDs have been used in domestic kitchens since the 1920s (Atwater 1947), their effectiveness at grinding domestic food wastes into particles that can be reliably conveyed in sewers has been studied rarely. Although some earlier studies were concerned with the implications for solids conveyance and transformation (e.g. Jones 1990), few have considered the physical characteristics and transport mechanics of FWD particles in sewer networks. Many objections to FWD use are based on anecdotal observations rather than objective, testable data, for example, recent studies such as Thomsen *et al.* (2018), the EU DECISIVE project, asserted that ground food introduced to sewers leads to unspecified 'damage' and 'risk' but without providing supporting evidence.

There is only very limited information about how FWD solids move in sewer networks, their deposition likelihood, and their re-entrainment potential. This study aims to assess the transportability of food waste disposer particles within a sewer system.

Sewer solids transport and FWD particles

The variety and range of solids entering, depositing and moving in sewers are broad (Ashley *et al.* 2004). Where there are sanitary sewers separate from stormwater collection systems, the solids are comprised of domestic, commercial and industrial inputs (e.g. Alda-Vidal *et al.* 2020). Increased used of FWDs could result in food waste comprising a significant organic load input to sanitary sewers.

Settling and transport of solids by turbulent flows are dependent primarily on particle and flow characteristics. The particle characteristics include particle diameter (d), density (ρ_s) and shape. Equation (1) reflects the balance between flow and particle characteristics. In this, w is the particle settling velocity and u^* is the boundary shear velocity, given by Equation (2), for which τ is the boundary shear stress and ρ is the fluid density. This reflects the ability of turbulent flows to transport solids, while the particle settling velocity reflects the ability of the solid particle to settle, incorporating particle size, density and shape in a single parameter (Breusers & Raudkivi 1991):

$$\xi = w/u^* \quad (1)$$

$$u^* = \sqrt{\frac{\tau}{\rho}} \quad (2)$$

Equation (1) presents the sedimentation parameter, ξ, which reflects the balance between fluid mobilising forces and the inertia of solid particles. According to Breusers & Raudkivi (1991), particles in a turbulent flowing fluid would be expected to settle onto the bed when $\xi > 6$ or to move along the bed as bedload when $6 > \xi > 2$. Below 2, particles will move either in suspension or by intermittent contact with the bed. However, the ranges of this non-dimensional ratio have been determined from observations of granular particles with high sphericity, and while indicative of the potential movement of organic particles of low density, investigation is required to confirm these thresholds for low density, irregularly shaped food particles.

Numerous studies have determined that the particle size of wastewater derived organic solids conveyed in sewers are <0.1 mm (e.g. Levine *et al.* 1985; Ashley *et al.* 2004) and that the settling velocities of these particles vary widely. For example, Pisano (1996) gives a range from 0.001 to 1 cm/s for all particles conveyed in dry weather flow from samples in the USA and Canada. Michelbach & Whorle (1992) determined settling velocities for particles in dry weather flows for 55 sites in Germany as ranging from 0.01 cm/s to 8.7 cm/s. Given the wide range of organic solids already present, FWD inputs may not substantially change the composition, but the relative impact of FWD inputs have not generally been considered, thus a robust investigation is needed to characterise the properties of FWD derived solids specifically.

The American Society of Sanitary Engineering (ASSE 2019) provides performance requirements for food waste disposers, primarily that particles no greater than 12.7 mm should discharge from the device, and particles greater than 6.4 mm should comprise less than 6.25% of the input load. This was specifically for a 454 g food mix comprised of steer ribs, carrots, celery and lettuce in equal proportions. The Association of Home Appliance Manufacturers (AHAM 2009) provided a more detailed protocol for testing, using the same food mix. They suggest using a sieve stack to characterise the spread of particle sizes between 0.425 mm, 2.360 mm, 6.350 mm and 12.700 mm sieves, based on the standard phi scale.

Previous studies on FWD-derived solids have used sieve testing to determine particle size, but without a consistent sieve stack, consistent procedure, or consistent food mix. Therefore comparisons of results for the characteristics of FWD derived solids is problematic. Kegebein *et al.* (2001) used six sieve sizes and considered 16 foods (some mixed), and also the settling behaviour of food mixes. Most particles were smaller than 2 mm and the settling velocity was up to around 0.06 m/s. Galil & Shpiner (2001) used five sieve sizes to examine unspecified food mixes from FWDs with different grind speeds to determine that most particles were <2.9 mm in size and that 'scouring' velocities were from 0.5 m/s for the lightest particles up to 0.84 m/s for some particles of egg shell and bone (although it was noted that this high scour velocity could correspond to only a 'very small part of the ground material'). These results were based on an adjusted Camp's formula (Equation (3)) using particle relative densities (by comparing with sucrose solutions of known density), of 1.0 to 1.1 for 'ordinary basket' particles (no egg shell or bone) and up to 2.3 for bones and egg shell:

$$Vc = \frac{1.486}{n} R^{\frac{1}{6}} \sqrt{B(Sp - 1)d} \tag{3}$$

In Equation (3), Vc is the scouring velocity in m/s; Sp relative particle density; d particle diameter; n Manning's roughness coefficient; R hydraulic radius; B is a non-dimensional coefficient related to particle type (0.04 for initiation of movement of granular particles; 0.06 for 'sticky' particles; 0.8 for fine cleaning of sewer). $B = 0.06$ was used in the calculations for the 'ordinary basket' particles and even with a range of sewer sizes (up to 800 mm) and relative flow depths (from 0.25 to 0.75), the particles were found to be conveyed at velocities as low as 0.5 m/s. These findings indicated that FWD solids will mainly be transported without deposition in the sewers considered in the study. However, the denser particles, including ground egg shells (2,241 kg/m^3) were found likely to deposit temporarily during low flow periods, as found by Mattsson *et al.* (2014).

Channon *et al.* (2013) used food mixes and five types of FWD, with only two sieve sizes, to show that most emitted particles were <4 mm in size, although there were variations in the results depending upon the type of FWD used. Drinkwater *et al.* (2015) used only three sieve sizes to determine that most FWD particles were <5.6 mm in size. In these studies, it was claimed that FWD solids could lead to blockage problems if input to sewer networks, while the other studies mentioned above suggested the heaviest FWD particles would only temporarily deposit before being scoured during the peaks of dry weather flows.

Critically, the literature described above provides little means to predict deposition risk of FWD derived particles in sewer systems. This paper reports on work designed to determine the physical nature of FWD derived solids, using a repeatable and rigorous set of tests, to address the question as to when, where and how FWD derived solids can be conveyed or deposited in sewer networks.

Objectives

A key knowledge gap in the assessment of the use of FWD is the risk associated with using conventional wastewater collection systems (sewers) as the transport conduit for the ground solids. There have been a number of individual observations in the field that FWD particles can deposit in sewer systems and possibly create problems as outlined above (e.g. Mattsson *et al.* 2014; Drinkwater *et al.* 2015). In the study reported here the intention was to establish an experimental protocol to collect high quality (repeatable) particle characterisation data in order to determine when there may be an in-sewer deposition risk from FWD particles.

Laboratory measurements are described which aimed to determine the physical size and fall velocity distributions of FWD derived particles for a wide range of food types. The food types selected were the more common components of food mixes currently found in the UK and USA, so that the impact of individual food type characteristics on representative particle mixtures may be examined.

MATERIALS AND METHODS

Food types and food mixes

This study considers the term 'food type' to represent an individual food (e.g. carrot) while 'food group' refers to the broader category (e.g. vegetables). The study aimed to characterise a range of common food types (e.g. potato, onion and carrot) spanning several food groups, e.g. vegetables and fruit. FWD particles from these different food types were expected to exhibit a variable range of physical characteristics.

The study has used published data for food waste generation in UK households (WRAP 2009), and US households (Kim *et al.* 2015) to select a range of foods for study. Table 1 shows the typical overall composition of food waste (henceforth referred to as a 'food waste mix') in the UK (WRAP 2009) and in the USA (Kim *et al.* 2015). This shows that: (i) many of the same food types appear in both mixes; (ii) there are substantial differences in the proportions of individual foods; (iii) different food groupings are used in the UK and USA.

The present study characterised 18 different solid food types shown to be significant in UK and US diets in Table 1 (indicated by *). These food types span all major food groups. The food types examined in this study are shown in Table 2, and were selected to provide a range of common foods found in both UK and US food mixes and that were expected to demonstrate a range of different properties when processed by FWD. Foods were raw unless otherwise stated in Table 2. Beef and chicken were purchased in cooked form, while pasta and rice were cooked according to manufacturer instructions.

Experimental overview

The experimental work was undertaken in several stages – (i) initial food processing; (ii) particle size characterisation; (iii) measurement of particle settling velocity; (iv) examination of re-entrainment of particles most likely to settle.

The primary equipment is shown in Figure 1, comprised of a FWD linked to a sealed unit to collect all the food particles, a water supply, a set of calibrated, graduated sieves, and a 290 mm diameter 1,293 mm length settling column.

All aspects of the particle measurement and characterisation took place on the same working day for each sample of ground food waste to ensure that the particles did not degrade between the different measurements. A detailed measurement protocol was followed according to the laboratory procedure described in detail by Nichols *et al.* (2020), and is summarised here. The entire process (from initial food processing to particle size and fall velocity measurement) was repeated three times for each food type to quantify experimental variability and the data were then averaged.

Initial food processing

Food samples were obtained from a standard commercial source (Table 2) and stored according to the supplier

Table 1 | UK and US food waste mixes, groups and types (WRAP 2009; Kim *et al.* 2015)

UK food waste mix (WRAP 2009)

Vegetables (38%)		**Bakery (16%)**	
Potato*	40.1%	Bread*	82.5%
Mixed	13.0%	Speciality	10.1%
Onion	6.8%	Morning bread	1.9%
Carrot*	6.2%	Other	5.5%
Cabbage*	4.4%	*Characterised	82.5%
Lettuce	3.5%	Scale factor	1.21
Tomato	33%	Meat/Fish (12%)	
Roots	2.5%	Poulty*	48.8%
Cucumber	2.3%	Pork	19.5%
Corn	22%	Fish*	7.0%
Broccoli*	2.1%	Lamb	5.2%
Cauliflower	2.1%	Other*	19.5%
Salad	1.9%	*Characterised	75.3%
Bean	1.5%	Scale factor	1,33
Pepper	1.2%	Processed vegetables (4%)	
Leek	1.0%	Potato*	36.3%
Mushroom	0.8%	Slaw/humus	14.7%
Spring onion	0.4%	Other	49.0%
Other	4.5%	*Characterised	36.3%
*Characterised	52.9%	Scale factor	2.76
Scale factor	1.89	Staples (4%)	
Fruit (22%)		Cereal*	36.8%
Banana	28.5%	Rice*	31.4%
Apple*	23.9%	Pasta*	20.6%
Orange*	12.0%	Flour	0.0%

(*continued*)

Table 1 | continued

UK food waste mix (WRAP 2009)

Melon	9.2%	Other	11.3%
Stone fruit	6.2%	*Characterised	88.7%
Other citrus	4.1%	Scale factor	1.13
Bernes	4.1%	Dairy/Eggs (3%)	
Other	12.0%	Egg shell*	38.6%
*Characterised	35.9%	Cheese*	27.1%
Scale factor	2.78	Egg	17.1%
		Other	17.1%
		*Characterised	65.7%
		Scale factor	1.52

US food waste mix (Kim *et al.* ⊠⊠⊠⊠)

Fruit 37%)		Grains (21%)	
Grapefruit	31.3%	Spaghetti	222%
Banana peel	15.6%	Mac & cheese	16.7%
Watermelon	15.6%	Rice, cooked*	16.7%
Pineapple*	12.5%	Corn flakes*	11.1%
Apple*	9.4%	Cheerios	11.1%
Orange peel*	9.4%	Bread, white*	11.1%
Cantaloupe	6.3%	Sugar	11.1%
*Characterised	31.3%	*Characterised	38.9%
Scale factor	3.20	Scale factor	257
Vegetables (28%)		Meat (9%)	
Cabbage*	24.5%	Beef*	40 0%
Potato*	22.4%	Pork	26.7%
Lettuce	16.3%	Raw chicken skin	20 0%
Broccoli*	12.2%	Hot dog	13.3%
Carrot*	8.2%	*Characterised	40 0%
Celery*	8.2%	Scale factor	2.50
Cucumber	4.1%	Dairy (6%)	
Pepper	4.1%	Cheese*	40.0%
*Characterised	75.5%	Cottage cheese	40 0%
Scale factor	1.32	Butter	20.0%
		*Characterised	40 0%
		Scale factor	2.50

Percentages are of unprocessed (not dried) food waste by mass. Percentages of food groups (e.g. vegetables) indicate the proportion of each food mix (UK or US), while percentages of food items (e.g. potato) indicate their proportion within each food group. The percentage characterised indicates the proportion of each food group characterised in this study, and the scale factor is thus used to scale the results to represent the whole group.

instructions. Food was prepared by cutting into pieces small enough to fit into the FWD unit (3–4 cm approximately in each dimension). Foods were prepared in samples of around 500 g (±5% as per AHAM 2009), with the exact mass of each sample being recorded. Egg shells were mostly in a halved state (not crushed), and were rinsed before being introduced to the FWD.

The FWD used was an Insinkerator Evolution 100-1B (serial number 16093104329). The same FWD unit was used for all food types. The water supply to the FWD was turned on and supplied a constant flow of 0.17 l/s. Water was always below 27 °C (AHAM 2009). The entire 500 g (±5%) food sample was added into the FWD. The water supply was maintained until no visible particles could be seen exiting the disposer. This period lasted around 50–60 seconds in all tests for this constant flow rate. Any variation in water used between tests did not appear to link to food type.

The mixture of water and food particles exiting the disposer was collected in a clean and dry laboratory container.

Measurement of particle size distribution

The purpose of this measurement was to determine the mass proportion of the original food sample ground into certain sieve size fractions. A stack of sieves was used according to BS ISO 3310-1:2016 and BS ISO 3310-2:2013 to characterise the particle size distribution. The sieve sizes used ranged from -3ϕ to $+4\phi$ and were arranged in 0.5ϕ increments (where sieve size in mm is given by $2^{-\phi}$, and thus ranged from 0.06 mm to 8.00 mm). This provides a broader range and higher resolution than that suggested in the AHAM (2009) protocol. The water and particle mixture collected from the FWD was stirred to fully suspend the particles and tipped smoothly into the top of the sieve stack, ensuring all of the particles were emptied from the container, undamaged by rinsing.

Beginning with the top sieve, a small water flow was used to gently wash particles through into the next sieve if they were smaller than the sieve size, without visibly damaging the particles. This was repeated sieve by sieve, down the stack, spending at least 5 minutes on each sieve to ensure all particles smaller than the sieve size were carefully washed through. Once the particles had been separated on the sieves, the sieves had the excess water removed by firmly tapping them one-by-one repeatedly above a sink until no more excess water was being released. Each sieve (including the particles) was then weighed using a calibrated electronic balance, with a resolution of 0.1 g.

Table 2 | Food types used in this study

Food type	UK food group	US food group	Details	Brand
Apple	Fruit	Fruit	Pink Lady	Tesco
Beef	Meat/fish	Meat	Cooked slices	Tesco Finest
Broccoli stem	Vegetables	Vegetables	Pre-packed	Tesco
Cabbage	Vegetables	Vegetables	Sweetheart	Tesco
Carrot	Vegetables	Vegetables	Batons	Tesco
Celery	Vegetables	Vegetables	–	Tesco
Cheese	Dairy/eggs	Dairy	Mature Cheddar	Cathedral City
Chicken carcass	Meat/fish	–	Pre-cooked, meat removed	Tesco
Cornflakes	Staples	Grains	–	Kellogg's
Egg shell	Dairy/eggs	–	Chicken eggs	Various
Orange peel	Fruit	Fruit	Cambria Naval	Tesco
Pasta	Staples	–	Fresh penne (cooked)	Tesco
Pineapple	Fruit	Fruit	Costa-Rica	Co-op
Potato	Vegetables	Vegetables	Maris Piper	Tesco
Rice	Staples	Grains	Basmati pouch (cooked)	Tilda
Sunflower seeds	–	–	–	Tesco
White bread	Bakery	Grains	Toastie	Warburton's
Whole mackerel	Meat/fish	–	Gutted	Independent fishmonger

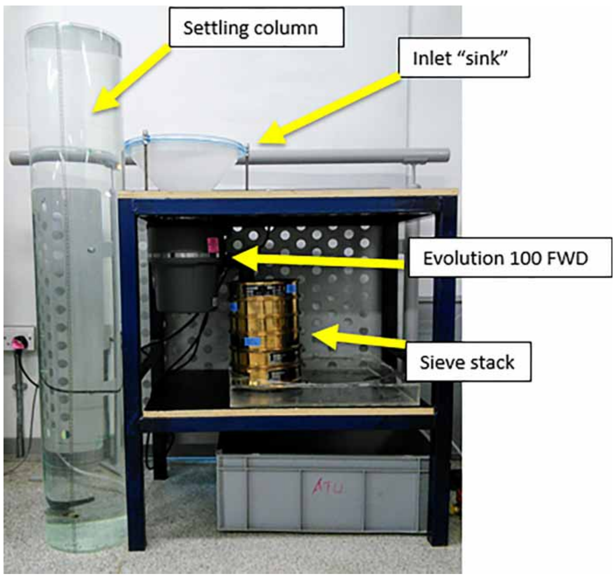

Figure 1 | Laboratory equipment – for the testing of an Evolution 100 Food Waste Disposer.

Particles were collected from the sieves to be used in the particle settling velocity measurement, and the sieves were thoroughly washed. The wet sieves were then tapped again to remove excess water, and were weighed (without particles). The wet sieve mass was subtracted from the wet sieve mass with particles to give the mass of wet particles collected in each sieve. The proportion in each sieve was calculated as the ratio of the wet food mass in each sieve to the total wet food mass across all sieves multiplied by 100%, following the AHAM (2009) protocol. This process was repeated three times for each sample for each food type and averaged, again according to AHAM (2009).

Settling velocity

The maximum settling velocity of the food particles within each sieve size fraction for each food type was measured. This provided the information needed to determine the likelihood of those particles settling within a sewer flow (Equation (1)). Here, 2 g samples of food particles were taken from each sieve, mixed carefully to ensure uniformity. The 2 g sample was mixed with 15 ml of water to form a suspension before being carefully tipped into the centre of the 295 mm diameter settling column's water surface, without giving the food particles any initial vertical velocity. Settling time was recorded at regular intervals throughout the 1,293 m long column to determine the point at which a stable terminal velocity was reached. For all foods, terminal velocity occurred by 385 mm below the water level. The time taken for the fastest falling particle to travel a distance of 710 mm below this height was recorded. The fastest falling particle within each size fraction was tracked as it represented the greatest settling velocity. The settling velocity of each size fraction for each of the food types was measured three times to assess variability and then averaged. The maximum settling velocity reported was therefore an average of three separate measurements.

Particle entrainment

As both the particle size distribution (psd) and the fall velocity distribution by mass fraction had been obtained from all the food groups it was possible to estimate the solid density of particles of a particular size fraction. This was done to estimate the boundary shear stress at the threshold of motion. The size fraction through which 95% of the mass is finer (d_{95}) was selected as the practical maximum particle size for the ground food waste of each food group. Once this was calculated by interpolation of the psd data, the fall velocity for that particle size (V_{95}) was also estimated by interpolation of the fall velocity data. The Reynolds Number ($Re = \rho V_{95} d_{95}/\mu$, where μ is dynamic viscosity of the fluid) associated with the size faction d_{95} was calculated and this was used to estimate the drag coefficient C_D using Equation (4) (Barati et al. 2014). Once this had been obtained then the solid density of the ground food waste (ρ_s) for the d_{95} size fraction could be obtained using Equation (5) (force balance equation for a sphere falling in a fluid at terminal velocity). Both Equations (4) and (5) assume that the particles are spherical in shape:

$$C_D = \frac{24}{Re} + \frac{3}{\sqrt{Re}} + 0.34 \tag{4}$$

$$\rho_s = \left[\frac{3V_{95}^2 C_D \rho}{4gd_{95}}\right] + \rho \tag{5}$$

Egg shell was identified by previous field studies (Mattsson et al. 2014) as a food type more likely to settle within sewers. Given the higher particle density and the irregular shape of egg shell particles, additional experiments were carried out to better ascertain the shear stress required to mobilise deposited egg shell particles as a function of their size, density and the ambient flow conditions. The results were then used to determine the equivalent spherical particles with similar behaviour, as used in conventional threshold of particle motion relationships.

An erosion meter based on the design of Liem et al. (1997) was used. First, a shear stress calibration was performed using sands of different sieve sizes, and the frequency of rotation at the threshold of motion for each size was determined, so that a bed shear could be estimated with a fixed value of Shields' number, as given in Equation (6):

$$\Theta = \frac{\tau_c}{(\rho_s - \rho)gd} \tag{6}$$

where Θ is the Shields' number, τ_c the critical shear stress, ρ_s is the particle density, ρ is the fluid density, g is the acceleration due to gravity, and d is the particle diameter. This follows the methodology described in Seco et al. (2014). This procedure enabled a linear fit to characterise the relationship between the angular velocity of the propeller and bed shear stress:

$$\tau = 0.075\omega - 1.055 \tag{7}$$

where ω is the angular velocity of the propeller in revolutions per minute. This expression fitted the data with a coefficient of determination of 0.995. The expression was used to determine the applied bed shear stress at the threshold of motion for egg shell particles based on the measured angular propeller velocity.

Egg shells were processed using the FWD according to the method described in the 'Initial food processing' section. The shell particles were sieved into nine size fractions ranging from 0.16 mm to 4.5 mm. For each size fraction, a sample was collected and placed in the base of the erosion meter such that an even bed was formed with the surface of the egg shell deposit 30 mm below the propeller. The angular velocity of the propeller was increased from zero in increments of one revolution per minute (RPM) until sustained motion of particles was observed (taken as several particles in motion at all times). Equation (7) was used to convert this angular velocity into a shear stress for the egg shell particles at the threshold of motion. Measurements using the egg shells were repeated twice for each size fraction to quantify a representative average and assess experimental variability.

RESULTS

Particle size distribution

Figure 2 shows the particle size distributions by mass on a phi scale for all 18 food types tested. Figure 3 presents the cumulative mass distribution. In both figures error bars represent standard deviation observed for the repeated

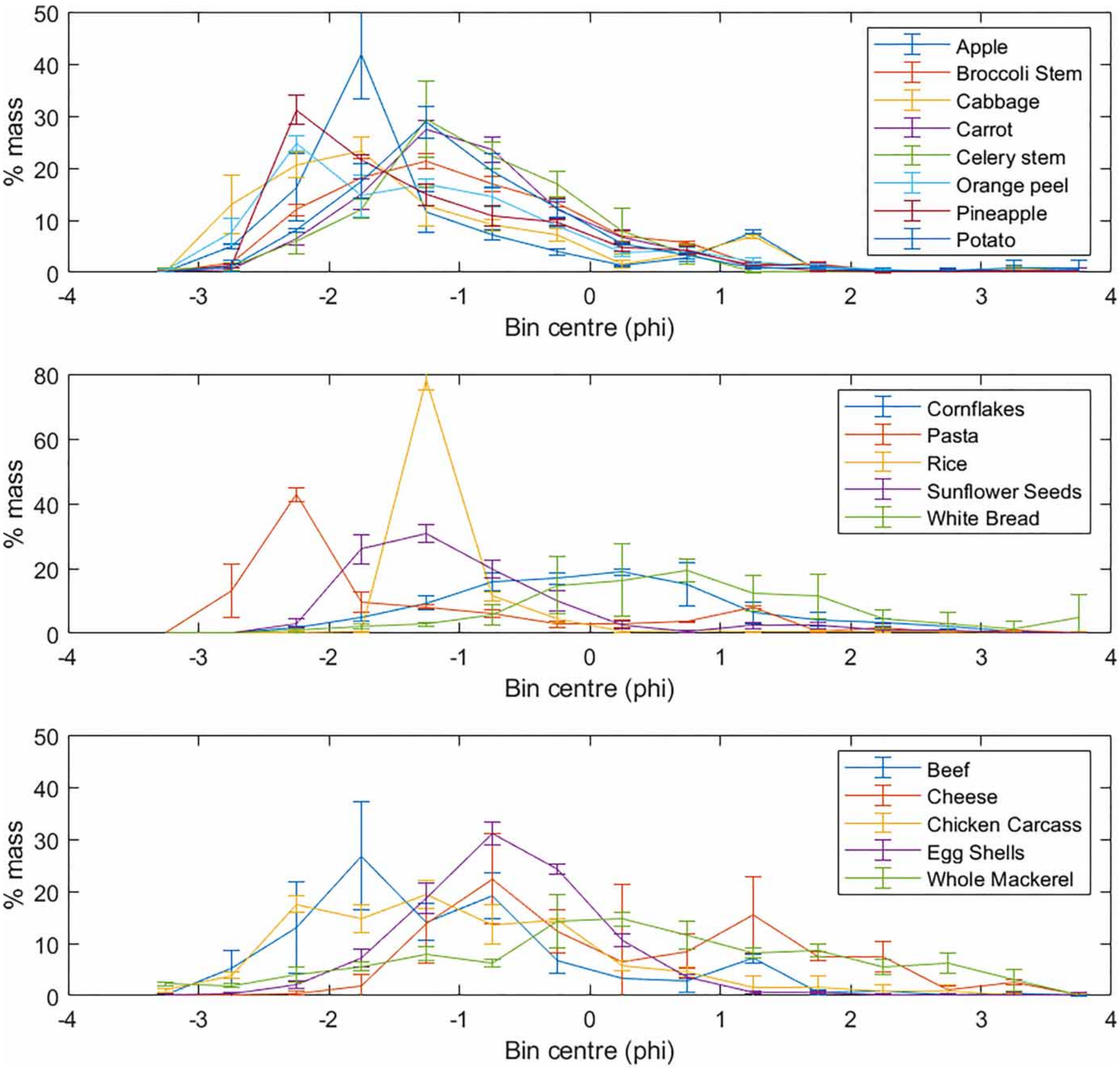

Figure 2 | Particle size distribution for 18 food types: (top) vegetables & fruit; (middle) bakery & staples; (3) meat/fish & dairy/eggs. Error bars represent standard deviation from repeated measurements.

measurements. The phi scale is a logarithmic scale that enables more nuanced inspection of trends for the finer particles. It is a standard scale for measurement and interpretation of sewer solids. The phi unit is calculated from the sieve opening size in mm in Equation (8):

$$phi = log_2(d) \tag{8}$$

So, a small phi value indicates a large sieve size (e.g. $-3\text{phi} = 8$ mm) and a large phi value indicates a small sieve size (e.g. $4\text{phi} = 0.0625$ mm). The bin centre on the horizontal axis is the centre of the size range captured by each sieve, in units of phi. Figure 2 shows that the particle size distributions were generally unimodal and demonstrated a wide range of sizes. For the 18 food types measured, the modal particle size occurred in the range of 0.59 mm to 4.76 mm. The mean particle size of each distribution ranged from 0.58 mm to 2.70 mm. The narrowest size distribution was for rice, which showed a much more prominent mode (most common size fraction), as the rice particles were already close to this modal size when entering the FWD. The width of each distribution is quantified via the standard deviation, as shown in Table 3.

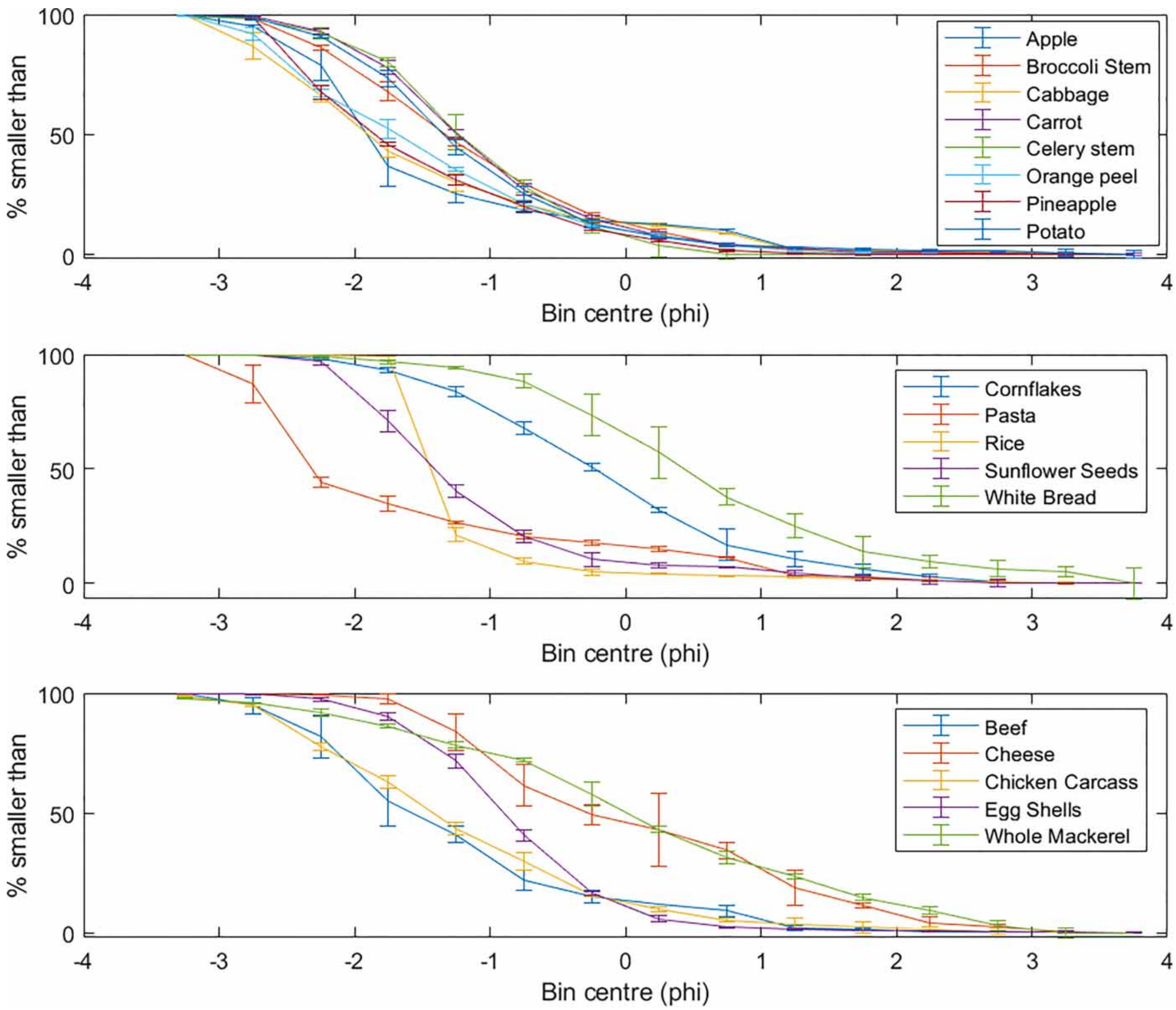

Figure 3 | Cumulative size distribution for 18 food types: (top) vegetables & fruit; (middle) bakery & staples; (3) meat/fish & dairy/eggs. Error bars represent standard deviation of repeated measurements.

Some analytical distribution types are known to be used to characterise particle size distributions in soils and other granular materials. This enables empirically derived distributions to be approximated by a simple analytical expression with a small number of parameters. A common distribution function for particle size distributions is the Gamma distribution (Equation (9)):

$$f(x) = \frac{(x/b)^{a-1}\exp(-x/b)}{b\Gamma(a)} \tag{9}$$

where x is the positive particle size, a is the shape parameter (producing a unimodal skewed distribution for $a > 1$, with less skew as a increases), b is the scale factor (which has the effect of stretching or compressing the range of the distribution), and Γ is the Gamma function.

For each food type, a Gamma distribution was fitted to the particle size distribution data using a least-mean-squares optimisation method. The optimised values of a and b are presented in Table 4, along with the root-mean-square error in units of percentage points. The data are presented in order of fit quality (best to worst).

The a parameter is always above 1, meaning a 'humped' distribution shape, and varying generally between 2 and 13 as distributions are more or less skewed. Rice is a clear outlier with $a = 39.21$ as the distribution is a very clear and symmetrical peak (see Figure 2). The b parameter generally varies between 0.3 and 1.5 as the distributions are broader or narrower, again with rice as an outlier at $b = 0.06$ as

Table 3 | Mean particle size and standard deviation for the 18 characterised food types, ordered by mean particle size

Food type	Mean particle size (phi)	Standard deviation (phi)	Mode (phi)	Mean particle size (mm)	Standard deviation (mm)	Mode (mm)
Pasta	−1.43	1.38	−2.25	2.70	0.39	4.76
Pineapple	−1.34	0.95	−2.25	2.53	0.52	4.76
Cabbage	−1.31	1.25	−1.75	2.48	0.42	3.36
Orange peel	−1.28	1.09	−2.25	2.42	0.47	4.76
Apple	−1.26	1.22	−1.75	2.39	0.43	3.36
Beef	−1.08	1.13	−1.75	2.11	0.46	3.36
Chicken carcass	−1.02	1.16	−1.25	2.03	0.45	2.38
Rice	−1.01	0.70	−1.25	2.01	0.62	2.38
Broccoli stem	−0.94	1.01	−1.25	1.92	0.50	2.38
Sunflower seeds	−0.94	0.96	−1.25	1.92	0.51	2.38
Cheese	−0.93	0.76	−1.25	1.90	0.59	2.38
Potato	−0.92	1.02	−1.25	1.89	0.49	2.38
Carrot	−0.85	0.94	−1.25	1.80	0.52	2.38
Egg shell	−0.61	0.77	−0.75	1.53	0.59	1.68
Cornflakes	0.06	1.11	0.25	0.96	0.46	0.84
Whole mackerel	0.27	1.54	0.25	0.83	0.34	0.84
Celery	0.28	1.27	−0.75	0.82	0.41	1.68
White bread	0.78	1.25	0.75	0.58	0.42	0.59

Table 4 | Gamma distribution parameters and root-mean-square error of optimised Gamma distributions, ordered by best fit

Food type	a	b	RMS error (% points)
Rice	39.21	0.06	1.93
Egg shell	6.13	0.33	0.93
Apple	10.14	0.39	4.45
White bread	2.92	0.39	1.64
Sunflower seeds	6.77	0.44	1.90
Pasta	12.20	0.44	5.47
Celery	5.07	0.52	1.51
Carrot	4.90	0.57	1.11
Cornflakes	2.99	0.60	1.21
Potato	4.86	0.62	1.16
Pineapple	4.69	0.98	4.11
Broccoli stem	3.46	1.01	0.83
Cheese	2.28	1.01	3.94
Beef	3.76	1.08	3.10
Whole mackerel	1.87	1.21	1.96
Cabbage	3.87	1.33	2.61
Chicken carcass	3.00	1.39	1.94
Orange peel	3.27	1.49	2.76

the distribution is very narrow. There appears to be no clear pattern of certain food groups exhibiting certain distribution parameters.

It can be seen that the majority of foods have a root-mean-square error below three percentage points, indicating that the Gamma distribution fits very well. The worst fits were obtained for pasta, apple and pineapple. This is likely due to the partially irregular and/or bimodal nature of their size distributions (see Figure 2). The psd and fitted Gamma curves for these three foods are shown in Figure 4 along with the best case fit (broccoli stem) for reference. The highest error of 5.47 percentage points for pasta is still a reasonably good fit and characterises the general shape of the distribution as shown in the figure.

Maximum settling velocity

Figure 5 shows the settling velocity of each food type as a function of each particle size fraction, while Figure 6 shows the cumulative mass percentage by maximum settling velocity. The maximum settling velocities for all food types, except egg shell, were below 0.1 m/s. Fruits, vegetables, meat/fish, pasta, and cheese were all well below 0.1 m/s, with grains such as rice and pasta showing slightly higher maximum

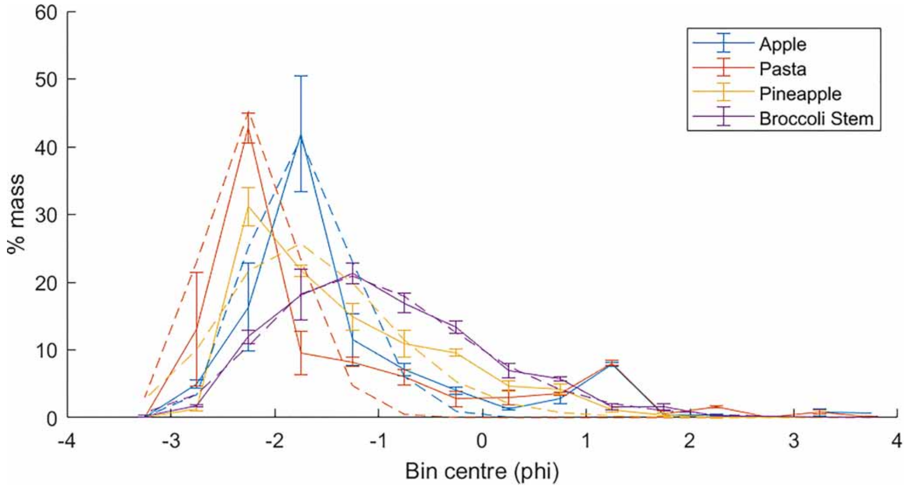

Figure 4 | Gamma distribution fitted to four food types showing the best (broccoli stem) and worst (pasta, pineapple, apple) fitting cases. Solid lines are measured data. Error bars are standard deviation of repeated measurements. Dashed lines are fitted Gamma distributions.

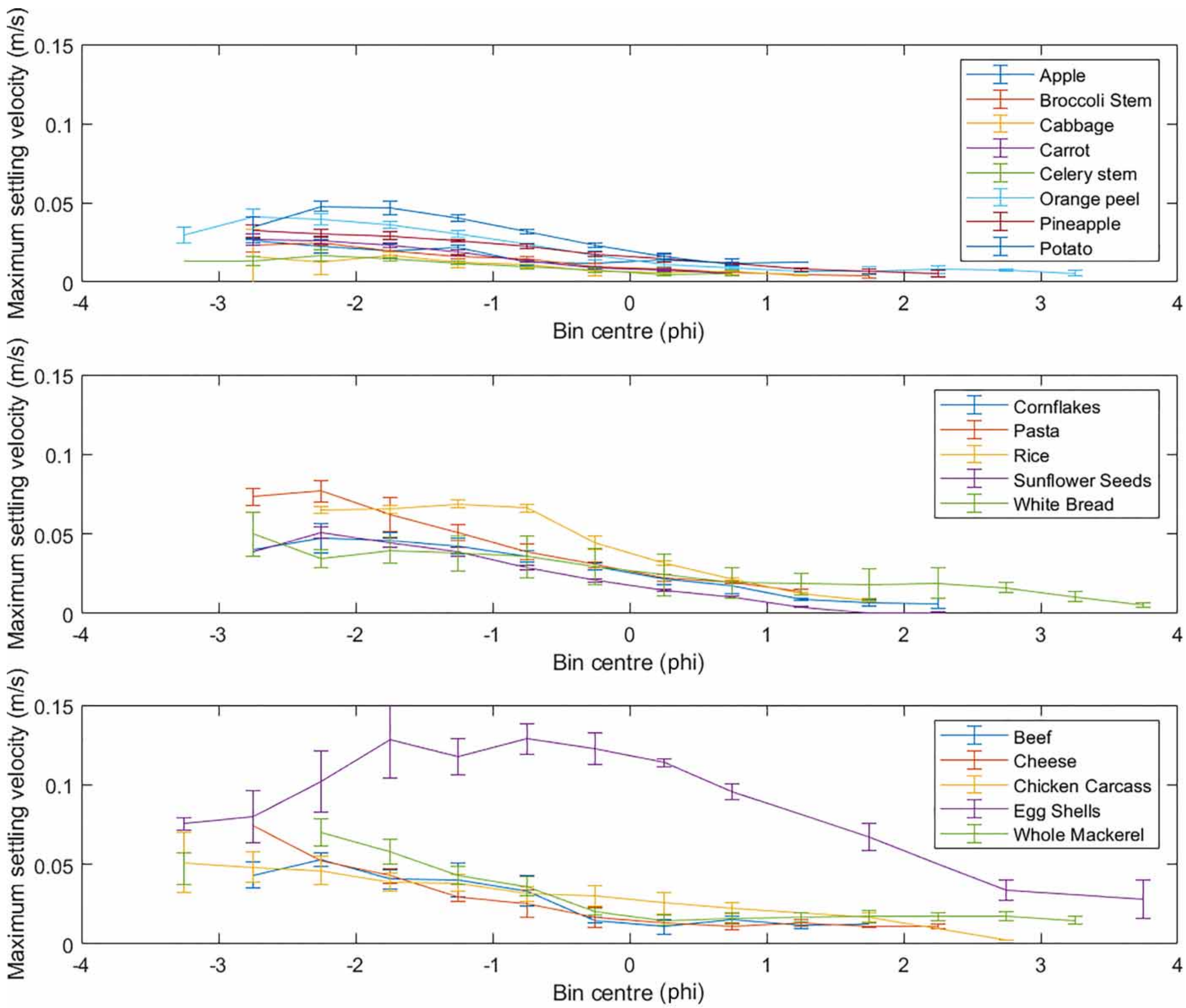

Figure 5 | Maximum settling velocity by particle size for all food types, (a) fruits and vegetables, (b) staples and grains, (c) meats, fish and dairy.

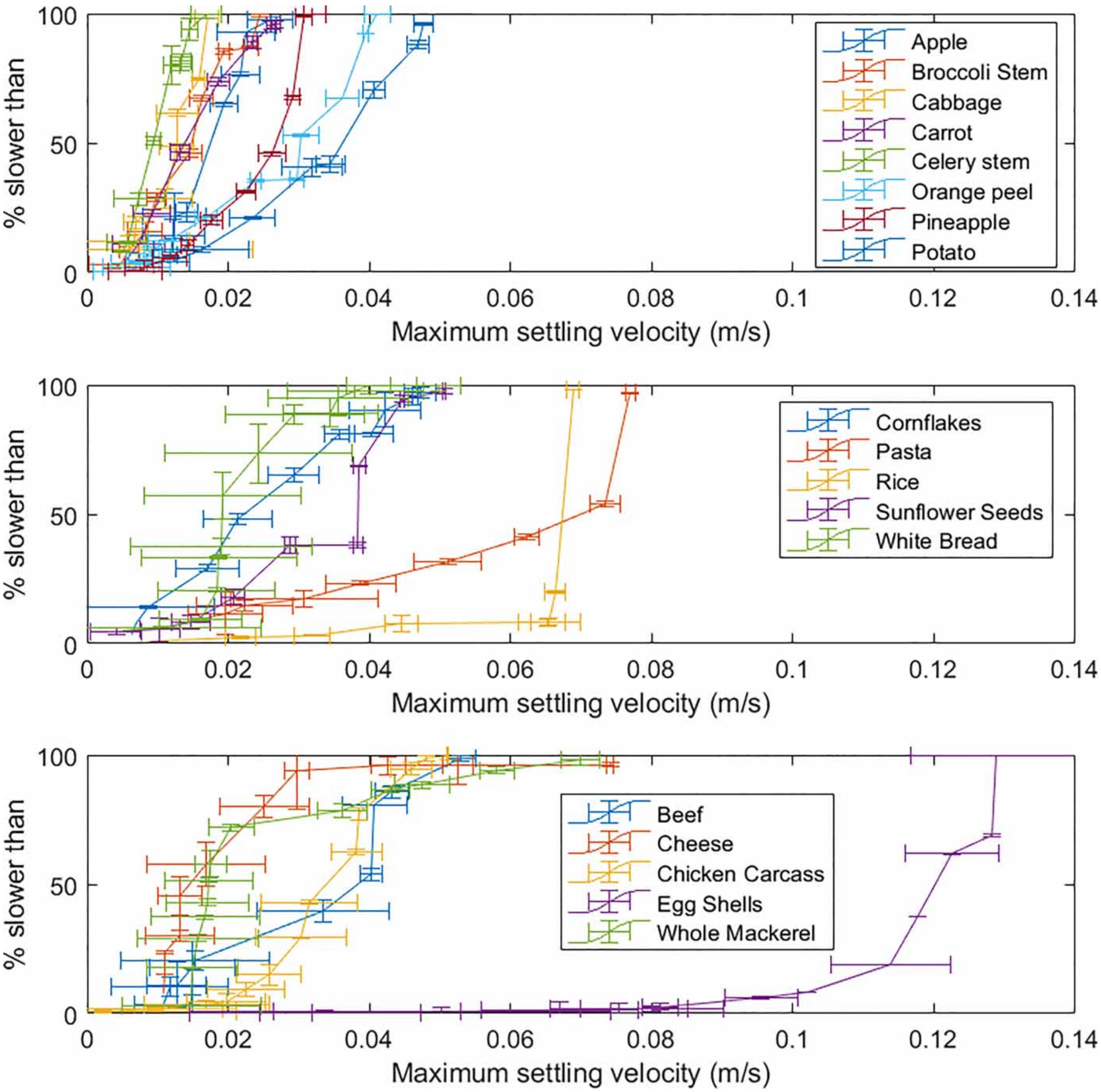

Figure 6 | Cumulative mass percentage by maximum settling velocity.

particle settling velocities. The clear outlier was egg shell which showed maximum settling velocities over 0.1 m/s for many particle sizes and for the largest particle sizes up to almost 0.13 m/s. For some foods the maximum settling velocity of particles within some sieve sizes could not be measured as the number of particles collected from this fraction was too low to enable measurement. The standard deviation between repeated measurements of particle fall velocity was calculated for size fractions of each food type. Averaged across all sizes and foods, the standard deviation of the particle fall velocity was around 4 mm/s within a size fraction. Generally the standard deviation of the maximum particle fall velocity within a size fraction averaged across each food type was below 5 mm/s, except for chicken carcass (7 mm/s), white bread (9 mm/s) and egg shell (11 mm/s). This is likely due to the complex nature of chicken carcass (mixture of bone, sinew, flesh etc.), variability of white bread size fractions (see Figure 3) and the larger measurement uncertainty for egg shells, possibly due to the particle shape and also as the fall velocity was much higher than for other foods.

Figure 6 illustrates that egg shells, ground pasta and rice are likely to provide the food particles with the highest likelihood of deposition. It can be seen for all three food types that the majority of the ground food has high maximum settling velocities. This indicates that rice, pasta and especially egg shells, are the food types that need to be examined for the risk of deposition in downstream sewers.

Particle transport potential

The data of particle density, calculated according to the 'Particle entrainment' section, indicated that for all the studied food types except egg shells, the particle densities ranged from 1,006 kg/m^3 to 1,059 kg/m^3 (see Table 5). Only the egg shells indicated a higher density of around 1,165 kg/m^3. Using the d_{95} values and the particle density values it was possible to estimate a boundary shear stress (τ_{crit}) that would entrain the maximum particle sizes for each food group using the widely used Shields criterion – Equation (6). As can be seen in Table 6 these boundary shear stresses (estimated using a conservative value of Shields Number of 0.065) ranged from 0.01 to 0.15 N/m^2, values that would be commonly encountered in many foul and combined sewers during dry weather flow. Only the egg shells with an apparent particle solid density of 1,165 kg/m^3 required a boundary shear stress of 0.38 N/m^2, a significantly higher value. It was decided to examine the entrainment behaviour of egg shells in more detail for two reasons: (i) it is the food group that has a significantly higher shear stress threshold than all the other food groups; (ii) visual inspection indicated that the egg shell particles were not spherical in shape and so weaken the assumptions used in Equations (4) and (5).

Erosion meter tests were conducted for egg shell particles as described in the 'Particle entrainment' section. The shear stress observed to entrain deposited egg shell particles is shown in Figure 7 and is higher than estimated and reported in Table 6. Error bars on the data indicate the maximum and minimum shear stress measured for repeated tests. While the apparent density of egg shell based in its settling velocity was 1,165 kg/m, direct measurements of egg shell density by Carter (1968) indicate that the density of egg shell is 2,241 kg/m^3 ± 4 kg/m^3. If this value is used with the estimated shear stress from the erosion meter tests, it can be seen that the Shields number (Equation (6)) is close to 0.065 on average (threshold for sustained particle movement), varying non-linearly from 0.036 to 0.078 depending on particle size, and suggesting that the shape of the egg shell particles at the different size fractions may also have an effect on their entrainment. Larger egg shell particles are observed to have a plate-like shape with lower sphericity. This leads to a larger deviation from

Table 5 | Apparent particle density for the largest practical particle sizes of FWD-derived particles for 18 food groups

	Apple	Beef	Broccoli stem	Cabbage	Carrot	Cheese	Celery stem	Chicken carcass	Cornflakes	Egg shells	Orange peel	Pasta	Pineapple	Potato	Rice	Sunflower seeds	White bread	Whole mackerel
d_{95} (mm)	5.69	5.79	5.17	7.11	4.68	2.64	4.74	5.65	3.26	3.59	6.59	7.10	4.48	4.86	2.79	3.91	2.28	7.88
V_{95} (m/s)	0.026	0.043	0.024	0.017	0.026	0.040	0.016	0.048	0.047	0.111	0.037	0.073	0.031	0.041	0.043	0.050	0.039	0.050
ρ_s (kg/m^3)	1,006	1,015	1,006	1,002	1,009	1,049	1,005	1,022	1,046	1,165	1,011	1,033	1,015	1,021	1,051	1,040	1,059	1,015

Table 6 | Estimated entrainment threshold shear stress value for 18 common food groups

	Apple	Beef	Broccoli stem	Cabbage	Carrot	Cheese	Celery stem	Chicken carcass	Cornflakes	Egg shells	Orange peel	Pasta	Pineapple	Potato	Rice	Sunflower seeds	White bread	Whole mackerel
τ_{crit} (N/m^2)	0.02	0.06	0.02	0.01	0.03	0.08	0.02	0.08	0.10	0.38	0.05	0.15	0.05	0.07	0.09	0.10	0.09	0.08

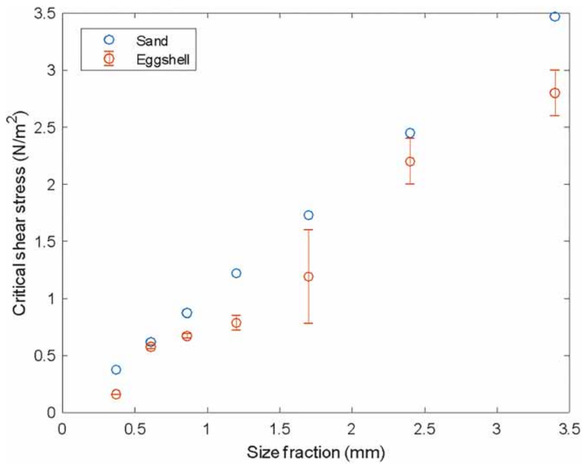

Figure 7 | Egg shell mobility.

spherical behaviour for the larger particles. Error bars are also larger for larger particle sizes due to the plate-like behaviour and the larger size intervals. It should also be noted that at all size fractions the shear stress required to mobilise egg shell particles was lower than the shear stress required to move equivalent sized sand particles.

DISCUSSION

The tests reported here are intended to contribute to the better understanding of the nature and potential behaviour of FWD derived particles and the implications of their input into sewer systems. The careful testing and clearly defined and followed protocols for examining individual food types provide scientific robustness and confidence that the results are both repeatable and realistic.

Careful laboratory testing has provided detailed descriptions of particle size distributions (psd) at ½ phi intervals for ground food waste from a single FWD model for 18 food types that are commonly found in the UK. These psd descriptions have a single mode, with a range of modal sizes and widths of the distributions. The shape of the particle size distributions is repeatable for particular food types but there are no clear similarities among food types within a given food group. The distributions were described well by Gamma distributions, which agrees with other studies of granular and ground materials.

Samples from the individual size fractions were collected and the maximum fall velocity was determined for each particle size fraction. This work has demonstrated that the highest fall velocities were found for pasta, rice and egg shell. The shape of the mass distributions for these food types showed that significant amounts of each had fall velocities above 0.06 m/s.

The values of maximum fall velocity did not link directly with particle size for different food types, indicating a variation in particle density. Taking the maximum practical size fraction (d_{95}), its fall velocity and assuming the particles were spherical, it was seen that there was a variation in particle density, and that for 17 out of the 18 food types these values were close to the density of water. One food type, egg shells, indicated a higher density and this food type was subjected to further investigation.

The detailed particle size distributions measured in this study correspond with the limited particle data obtained in earlier studies (Galil & Shpiner 2001; Kegebein *et al.* 2001; Channon *et al.* 2013; Drinkwater *et al.* 2015), although the data from these studies were generally of very low resolution so an objective comparison is difficult. The study by Drinkwater and colleagues using cooked food appears to be an outlier with this and other studies with regard to the particle size distribution of ground food waste, generally showing larger particle sizes.

Analysis using the maximum practical size fraction (d_{95}) for all the food groups indicated that the boundary shear stress needed to entrain FWD particles was low in comparison to boundary shear stresses found in most foul and combined sewer pipes. For egg shells further tests indicated that the boundary shear stress required to entrain these particles is considerably higher than for FWD-derived particles of other food types, most likely due to the higher density and is likely to be also affected by lower particle sphericity. It is clear that particle density is the most important particle parameter in determining the entrainment threshold for FWD particles. While the likelihood of egg shell settling is higher than other food types, egg shell deposits can be assumed to be moved by normal peak dry weather flows, and nonetheless egg shells only comprise around 1% of the overall mass of food waste so the likelihood of creating significant in-sewer deposition in sewer networks is very low.

CONCLUSIONS

It has been shown that for 18 common food types the modal particle size varied between 0.59 mm and 4.76 mm and the standard deviation varied between 0.34 mm to 0.62 mm. Particle size distributions are shown to conform well to Gamma distributions, meaning they can be characterised by just two parameters.

Particle densities were estimated using particle size and fall velocity data. This analysis demonstrated that most FWD particles had particle densities close to that of water. This results in these particles being entrained into motion at low values of boundary shear stress. The ease of entrainment means that the vast majority of food types is highly unlikely to form persistent deposits in sewer pipes.

Egg shell particles showed a submerged density estimate considerably higher than the other food types, and thus the entrainment threshold was considerably higher than for the other food types. The deposition risk of egg shells is thus higher than for other food types, however its overall prevalence in waste food is very low (around 1%) so it is unlikely to cause significant practical deposition issues.

These studies have shown that, by employing the robust experimental method described, the deposition risk of FWD derived particles can be assessed. Further work should expand the range of food types, and explore the implications when applied to flows in a range of sewer systems.

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DATA AVAILABILITY STATEMENT

All relevant data are available from an online repository or repositories. Available online: https://doi.org/10.5281/zenodo.3697302.

REFERENCES

AHAM, Association of Home Appliance Manufacturers 2009 *Food Waste Disposers FWD-1-2009*.

Alda-Vidal, C., Browne, A. & Hoolohan, C. 2020 'Unflushables': establishing a global agenda for action on everyday practices associated with sewer blockages, water quality and plastic pollution. *WIREs Water* **7**, e1452.

American Housing Survey 2013 U.S. Census Bureau, Current Housing Reports, Series H150/11, American Housing Survey for the United States: 2011, U.S. Government Printing Office, Washington, USA. Available online: https://www.census.gov/content/dam/Census/library/publications/2013/demo/h150-11.pdf.

Ashley, R. M., Bertrand-Krajewski, J.-L., Hvitved-Jacobsen, T. & Verbanck, M. 2004 *Sewer Solids – State of the Art.*

International Water Association Scientific and Technical Report No. 14. IWA Publishing. ISBN 1900222914.

ASSE, The American Society of Sanitary Engineering 2019 Standard #1008-2019, Plumbing Aspects of Residential Food Waste Disposer Units.

Atwater, R. M. 1947 The kitchen garbage grinder. *American Journal of Public Health and the Nations Health* **37**, 573–574.

Barati, R., Seyed, A. & Goodarz, A. 2014 Development of empirical models with high accuracy for estimation of drag coefficient of flow around a smooth sphere: an evolutionary approach. *Powder Technology* **257**, 11–19. doi:10.1016/j.powtec.2014.02.045.

Bisschops, J., Kjerstadius, H., Meulman, B. & van Eekert, M. 2019 Integrated nutrient recovery from source-separated domestic wastewaters for application as fertilisers. *Current Opinion in Environmental Sustainability* **40**, 7–13.

Breusers, H. & Raudkivi, A. 1991 *Scouring: Hydraulic Structures Design Manual*. A.A. Balkema, Rotterdam.

Carter, T. C. 1968 The hen's egg: density of egg shell and egg contents. *Journal of British Poultry Science* **9** (3), 265–271. doi:10.1080/00071666808415718.

Channon, D., Calderon, S., Groves, L. & Torrance, F. 2013 Food Waste Disposal Units and their possible impact on sewer rat populations in the United Kingdom. *Water Practice & Technology* **8** (2), 202–214. doi:10.2166/wpt.2013.022.

Defra Environment Bill 2020 UK Parliament, Parliamentary Bills. Available online: https://bills.parliament.uk/bills/2593.

Drinkwater, A., Homewood, S., Moy, F., Palfrey, R., Sivil, D. & Spain, K. 2015 *Food Waste Disposers – Consequences for the Water Industry of Widespread Market Penetration and Consideration of Rodent Issues*. UKWIR Report Ref. No. 15/SW/01/10, ISBN 184057 761 4. 3 – Laboratory and Rig Testing Results and consideration of rodent issues.

EU Amending Waste Framework Directive 2018 Directive (EU) 2018/851 of the European Parliament and of the Council of 30 May 2018 amending Directive 2008/98/EC on waste. Available online: https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32018L0851&rid=5.

Evans, T., Andersson, P., Wievegg, Å. & Carlsson, I. 2010 Surahammar: a case study of the impacts of installing food waste disposers in 50% of households. *Water and Environment Journal* **24**, 309–319.

Galil, N. & Shpiner, R. 2001 Additional pollutants and deposition potential from garbage disposers. *Water and Environment Journal* **15**, 34–39.

Iacovidou, E., Ohandja, D., Gronow, J. & Voulvoulis, N. 2012 The household use of food waste disposal units as a waste management option: a review. *Critical Reviews in Environmental Science and Technology* **42** (14), 1485–1508. doi:10.1080/10643389.2011.556897.

Jenkinson, S. 2020 *Reviewing Biowaste Treatment In The UK*. Available from: https://resource.co/article/reviewing-biowaste-treatment-uk.

Jones, P. H. 1990 *Kitchen Garbage Grinders (KGGs/Food Waste Disposers) the Effect on Sewerage Systems and Refuse Handling*. Institute for Environmental Studies, University of Toronto, Toronto. Available from: https://images.insinkerator.com/enviro/Jones_U-Toronto_Canada_(1990).pdf.

Kegebein, J., Hoffmann, E. & Hahn, H. 2001 Co-transport and re-use. An alternative to separate bio-waste collection? *Wasser-Abwasser GWF* **142** (6), 429–434.

Kim, M., Chowdhury, M., Nakhla, G. & Keleman, M. 2015 Characterisation of typical household food wastes from disposers: fractionation of constituents and implications for resource recovery at wastewater treatment. *Bioresource Technology.* **183**, 61–69.

Levine, A., Tchobanoglous, G. & Asano, T. 1985 Characterisation of the size distribution of contaminants in wastewater: treatment and reuse implications. *Journal (Water Pollution Control Federation)* **57** (7), 805–816.

Liem, R., Spork, V. & Koengeter, J. 1997 Investigation on erosional processes of cohesive sediment using an in-situ measuring device. *International Journal of Sediment Research* **12** (3), 139–147.

Mattsson, J., Hedstrom, A. & Viklander, M. 2014 Long-term impacts on sewers following food waste disposer installation in housing areas. *Environmental Technology* **35** (21), 2643–2651.

Michelbach, S. & Whorle, C. 1992 Settleable solids in a combined sewer system – measurement, quantity, characteristics. *Water Science & Technology* **25** (8), 181–188.

Nichols, A., Jensen, H., Tait, S. & Legge, A. 2020 Food waste disposer particle characterisation [Data set]. *Zenodo*. http://doi.org/10.5281/zenodo.3697303.

Pisano, W. 1996 Summary: United States sewer solids settling characterisation and methods: results, uses and perspectives. *Water Science & Technology* **33** (9), 109–116.

Run4Life 2020 *Recovery and Utilisation of Nutrients for low Impact Fertiliser*. Demonstration site fact sheet – Helsingborg. Available from: https://run4life-project.eu (Accessed date: 2 May 2020).

Sancho, I., Lopez-Palau, S., Arespacochaga, N. & Cortina, J. 2019 New concepts on carbon redirection in wastewater treatment plants: a review. *Science of the Total Environment* **647**, 1373–1384.

Schanes, K., Dobernig, K. & Gözet, B. 2018 Food waste matters – A systematic review of household food waste practices and their policy implications. *Journal of Cleaner Production* **182**, 978–991.

Schinkel, J. 2019 Review of policy instruments and recommendations for effective food waste prevention. *Proceedings of the Institution of Civil Engineers – Waste and Resource Management* **172** (3), 92–101. https://doi.org/10.1680/jwarm.18.00022.

Seco, I., Gómez Valentín, M., Schellart, A. & Tait, S. 2014 Erosion resistance and behaviour of highly organic in-sewer sediment. *Water Science and Technology* **69** (3), 672–679.

Skambraks, A.-K., Kjerstadius, H., Meier, M., Davidsson, A., Wuttke, M. & Giese, T. 2017 Source separation sewage systems as a trend in urban wastewater management: drivers for the implementation of pilot areas in Northern Europe. *Sustainable Cities and Society* **28**, 287–296.

Slorach, P. C., Jeswani, H. K., Cuéllar-Franca, R. & Azapagic, A. 2020 Assessing the economic and environmental sustainability of household food waste management in the UK: current situation and future scenarios. *Science of the Total Environment* **710**, 135580.

STOWA 2015 *Principles for Implementing LCA: Food Waste in the Water Chain*. Stichting RIONED/STOWA 2015-W-02.

Thomsen, M., Romero, D., Caro, D., Seghetta, M. & Cong, R.-G. 2018 Environmental-economic analysis of integrated organic waste and wastewater management systems: a case study from Aarhus City (Denmark). *Sustainability* **10**, 3742. doi:10.3390/su10103742.

van Leeuwen, K., De Vries, E., Koop, S. & Roest, K. 2018 The energy and raw materials factory: role and potential contribution to the circular economy of the Netherlands. *Environmental Management*. https://doi.org/10.1007/s00267-018-0995-8.

Velenturf, A. P. M., Purnell, P., Tregent, M., Ferguson, J. & Holmes, A. 2018 Co-producing a vision and approach for the transition towards as circular economy: perspectives from government partners. *Sustainability* **10**, 1401. doi:10.3390/su10051401.

WRAP 2009 Household Food and Drink Waste in the UK. ISBN 1-84405-430-6. Available online: https://wrap.org.uk/sites/default/files/2020-12/Household-Food-and-Drink-Waste-in-the-UK-2009.pdf.

WRAP 2015 Household food waste in the UK. Available online: https://wrap.org.uk/resources/report/household-food-waste-uk-2015#:~:text=HHFW%20in%20the%20UK%20was,in%202015%20compared%20to%202007.

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LITERATURE OVERVIEW

Impacts of Disposers and Food Waste Management

Summary prepared by:
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Revised:
December, 2020

Table of Contents

List of Acronyms

AD	Anaerobic Digestion
BNR	Biological Nutrient Removal
BOD	Biological Oxygen Demand
CO_2e	Carbon Dioxide Equivalent
COD	Chemical Oxygen Demand
CFWC	Curbside Food Waste Collection
FFA	Free Fatty Acid
FOG	Fats, Oils, and Greases
FW	Food Waste
FWD	Food Waste Disposer
GWP	Global Warming Potential
LCA	Life Cycle Analysis
MLE	Modified Ludzack-Ettinger
MRF	Material Recovery Facility
MSW	Municipal Solid Waste
rbCOD	Readily Biodegradable Chemical Oxygen Demand
SP	Smog Potential
TKN	Total Kjeldahl Nitrogen
TN	Total Nitrogen
TSS	Total Suspended Solids
VS	Volatile Solids
VSS	Volatile Suspended Solids
WTE	Waste-to-Energy
WW	Wastewater
WWTP	Wastewater Treatment Plant

1.0 Introduction

Food waste disposers were invented in the 1940's, initially as a convenience for residential kitchens and cooks. As interest developed in the post-WWII era's housing boom, disposers were thoroughly evaluated by municipalities to assess their efficacy with respect to local solid waste and wastewater collection and treatment systems.

By the end of the 20th century, disposers had become a standard appliance, installed in the majority of U.S. homes and nearly ubiquitous in new residential construction. The market for commercial food waste disposers – in a variety of food-serving establishments, such as restaurants, cafeterias, and markets – also has grown. International acceptance of food waste disposers also is growing, in response to significant concerns about diverting organic food waste from landfills and increasing the beneficial use of food waste for land application. Everything municipalities normally do with food waste is environmentally noxious: stored inside buildings (even refrigerated); piled in bags on sidewalks; collected in trucks; and shipped to distant landfills, where it generates leachate and greenhouse gases. This process is not cheap, hygienic, environmentally friendly, nor sustainable.

In sum, food waste disposers form an impressive part of an integrated modern waste management system in many parts of the world.

This document reviews fifty (50) of the most recent studies and reports, three (3) executive summaries, two (2) literature reviews, one (1) textbook, two (2) specifications and requirements, and one (1) internal calculation, for a total of fifty-nine (59) research references. All information in this document was conducted by universities, research institutions, and government agencies across the United States and in many countries that examine the efficacy of food waste disposers. It compiles the findings regarding all facets of the sewage collection, treatment, and disposal process and organizes the information according to major concerns and assumptions regarding garbage disposers. In sum, these studies have largely determined that the impacts of disposers are manageable, and that disposers provide a significant set of environmental benefits that merits their acceptance and use in conjunction with (rather than in competition to) other alternatives to divert organic waste from landfills.

2.0 Food Waste Disposer Background

2.1 Advantages

- Removing kitchen waste from compost produces cleaner and better compost [de Koning, 2004].
- Reduced transportation noise [de Koning, 2004].
- Reduced space concerns for food waste storage [de Koning, 2004].
- Renewable energy value of Wastewater Treatment Plant (WWTP) anaerobic digestion biogas [de Koning, 2004][Hernandez, 2002, "Los Angeles Digesters"][Karlberg, 1999][Karrman, 2001][Kegebein, 2001][Rosenwinkel, 2001][Shpiner, 1997].
- Reduced incidence of disease-causing vector attraction in comparison to food waste storage/collection [de Koning, 2004][Diggelman, 1998][Shpiner, 1997][Terpstra, 1995].
- Reduced truck collection, which blocks narrow streets [de Koning, 2004][Kegebein, 2001].
- Natural selector of organic wastes, whereas, composting relies on the education and goodwill of the participants [CECED, 2003].
- Reduces the potential of uncontrolled biochemical processes in landfills (i.e., leachate treatment) [Rosenwinkel, 2001].
- Reduced transportation emissions and costs [Karlberg, 1999][Karrman, 2001][Kegebein, 2001].
- High carbon content of food waste improves the overall WWTP nitrogen and phosphorus nutrient removal process [Diggelman, 1998][Kegebein, 2001][Rosenwinkel, 2001][Kim, 2019].
- Improved hygienic environment in comparison to food waste storage/collection [Kegebein, 2001][Rosenwinkel, 2001][Shpiner, 1997]Terpstra, 1995].
- Healthier Municipal Solid Waste (MSW) working environment [Karlberg, 1999].
- Less expensive and complicated than source-sorting food wastes [Karlberg, 1999].
- Reduced MSW garbage collection amount and frequency [Diggelman, 1998][New York City Department of Environmental Protection, *Executive Summary*, 1997][New York City Department of Environmental Protection, 1997][Shpiner, 1997].
- Promotes nutrient recycling from organic wastes when WWTP biosolids are land-applied [Diggelman, 1998].
- Environmentally friendly and sustainable food waste disposal option [Diggelman, 1998].
- As food waste is 70% water, the WWTP is a more natural system of waste processing than hauling the waste to a "solid waste" facility [Diggelman, 1998].
- As food waste is 70% water, the WWTP system reduces leachate diverted from landfill and compost systems, which reduces potential contamination to groundwater [Diggelman, 1998].
- Most convenient and likely-used source selector of organic kitchen wastes [Diggelman, 1998].
- As food waste is 70% water, the WWTP system anaerobic digestion process will produce a viable energy source, whereas, incineration offers a very small net energy gain that also produces contaminated emissions requiring additional treatment [Diggelman, 1998].

- As food waste is 70% water, the WWTP system is a more natural method of waste processing than composting, which, although enhanced by the additional moisture, does require stricter operational control to avoid anaerobic conditions, and results in the loss of most nutrients to the extent that the final product is of low value [Diggelman, 1998].
- Ease of use [Shpiner, 1997].
- WWTPs are equipped to treat food waste due to high water and organic content [Shpiner, 1997].

2.2 Disadvantages

- Increased potential loadings impact on combined sewer overflows [Rosenwinkel, 2001].
- Increased water consumption [Rosenwinkel, 2001].
- Increased energy consumption for both disposer use and WWTP aeration [Rosenwinkel, 2001].
- High initial costs for the user (not the municipality) [CRC, 2000][Diggelman, 1998][Karrman, 2001][Rosenwinkel, 2001].
- Potential grease/solids build-up in the sewer collection system, which increases maintenance costs [Kegebein, 2001][New York City Department of Environmental Protection, *Executive Summary*, 1997][New York City Department of Environmental Protection, 1997].
- Increased WWTP biosolids generation and disposal costs [de Koning, 1996][Karrman, 2001][Rosenwinkel, 2001][Shpiner, 1997][Terpstra, 1995].
- Increased loadings of BOD and TSS to the WWTP [Shpiner, 1997].

2.3 Disposer Specifications

- Association of Home Appliance Manufacturers (AHAM) provided testing specifications to test for the criteria in the ASSE Standard #1008 in a systematic way [AHAM, 2009].
- Sink mounted food waste disposer units should be designed to fit a sink with a 3.5 inch (89 mm) nominal drain opening (this is the normal drain opening size to which sinks are designed) [AHAM, 2009, p. 2].
- Residential Disposer Specifications as set by ASSE International [ASSE, 2006]
    - Discharge not less than 6.0 GPM (0.36 L/s) at a head of 10.0 inches (254.0 mm) [ASSE, 2006, p. 3].
    - Terminal outlet shall be 1.5 inches (40 mm) nominal tube size [ASSE, 2006, p. 1].
    - Ground product retained on the sieve should not weigh more than 1.0 ounces (28 grams) [ASSE, 2006, p. 4].
    - Particles on the inside of the FWD shall not exceed 0.25 inches (6.7 mm) [ASSE, 2006, p. 5].
    - For FWDs with a dishwasher connection, the water level shall not rise more than 1.0 inch (25.4 mm) above the water level in the sink [ASSE, 2006, p. 5].
    - For FWDs without a dishwasher connection, the water level shall not rise above the sink mounting flange to any degree [ASSE, 2006, p. 5].
    - There shall be no evidence of leakage during or after the cycle of running the FWD [ASSE, 2006, p. 6].

- Disposers have a 600W electric motor, used on average 2.4 times/day and 30 seconds each time [Karrman, 2001].
- Approximately 98% of all particles pass through a 2 mm sieve [Kegebein, 2001].
- The food waste disposer can be described as a mill rather than a cutter. It works with a rotary disk in which two hammer-cheeks mobile in horizontal direction are fastened. The disk is provided with 5 mm holes. In opposition to frequently heard prejudices, a disposer does not contain rotating knives [Rosenwinkel, 2001].
- Non-food wastes cannot be ground since the attempt will cause a resistance, which if it becomes excessive, will cause the resistor to switch off [Rosenwinkel, 2001].
- A 1400 rpm rotating disk with a number of 3-4 mm holes [Karlberg, 1999].
- The energy requirement for use is 3-4 kW-h//household/yr [Karlberg, 1999].
- A Japanese study found food waste particle dispersion between 2-5 mm [Karlberg, 1999].
- A grinding distribution of heaviest components show 62% of particles are <1.7 mm, 86% are <2.83 mm, and 94% are <3.36 mm [Shpiner, 1997].

2.4 Food Waste Composition

- Food Waste Composition varies depending on the culture and diets of the local community. Therefore, it is difficult to define a uniform composition of food waste. In some studies a "standard diet" is created in order to study local food waste compositions [Kim, 2015, p. 62].
- Since the COD/N and BOD/N ratios (63 and 27, respectively) were higher than the particulate ratios (42 and 22, respectively), this suggests that the non-settleable fraction (aqueous phase) can enhance the denitrification process and impact secondary aeration [Kim, 2015, p. 69].
- Considering 50 grinded food waste samples, the relative mass ratios of COD: BOD5: TSS: TN: TP: dry food waste was 1.21: 0.58: 0.36: 0.025: 0.013: 1 [Nakhla, 2014, p. 11].
- Assuming wet food waste consists of 30% dry waste and 70% water, dry food waste is defined by the following chemical formula: $C_{21.53}H_{34.21}O_{12.66}N_{1.00}S_{0.07}$ [PE Americas, 2011, p. 19].
- Assuming wet food waste is 30% dry waste and 70% water and 95% of the dry food waste is VS with the remaining 5% being inert, then 100kg of wet food waste would equate to a dry food waste of 44 kg COD with a 1.54 kg COD/kg dry food waste ratio applied [PE Americas, 2011, p. 19].
- For 30 kg of dry food waste, 17 kg is TSS while the remaining 13 kg is soluble and is removed during biological treatment [PE Americas, 2011, p. 20].
- The impact of FWDs depends on the food waste composition, which depends on the type of food waste [Thomas, 2010, p. 6].
- Daily person equivalent contributions due to organic food waste through disposers is 75 g/person/day for Chemical Oxygen Demand (COD), 50 g/person/day for Total Suspended Solids (TSS), 2.5 g/person/day for Total Kjeldahl Nitrogen (TKN), and 0.25 g/person/day for Total Phosphorus (P). This equates to a COD/TKN ratio of 30 [Bolzonella, 2003].
- Typical organic waste composition is 25.6% TS (74.4% water), 96.5% VS, 3.2% TKN, 0.2% P, and 1,200 mg/L COD [Bolzonella, 2003].

- Grindable food waste is about 35% of total household waste, which equates to 235 g/person/day (85 kg/person/yr.) [CECED, 2003].
- Airport food waste sample analysis results were moisture 72.9%, Total Solids (TS) 27.1%, and Volatile Solids (VS) 94.9% [Hernandez, 2002, "Los Angeles Digesters"].
- Generated food waste is 76 kg/person/yr., with 67% able to be ground through a disposer (i.e., 50.9 kg/person/yr.) [Karrman, 2001].
- Food waste generation is about 40-60 kg wet/person/yr. [Kegebein, 2001].
- Average food waste generation is 182 kg/household/yr. or 0.24 kg/person/day [CRC, 2000].
- Lagerkvist & Karlson, 1983 and Nilsson et al, 1990 both indicate that about 20% of food waste suitable for composting is not suitable for disposer grinding [Karlberg, 1999].
- Olsson & Retzner, 1998 indicates that 75 kg/person/yr. of food waste are generated [Karlberg, 1999].
- De Koning & Van der Graaf, 1996 assume that the total amount of food waste that can be ground through a disposer is 44 kg/person/yr. [Karlberg, 1999].
- Nilsson et al, 1990 state that about 75% of food waste Biochemical Oxygen Demand (BOD) is in particle form and 25% in dissolved form [Karlberg, 1999].
- The average person generates 0.29 lb./day of food waste with 0.21 lb./day (75%) able to be processed through a disposer [Diggelman, 1998].
- Food waste is 70% water and 30% solids [Diggelman, 1998].

Table 1 – Waste Compositions. Typical Food and Human Waste Compositions [Diggelman, 1998].

Waste Compositions					
Type	**C**	**H**	**O**	**N**	**S**
Food Waste	50.5%	6.72%	39.6%	2.74%	0.44%
Human Waste	59.7%	9.5%	23.8%	7.0%	0%

- Food waste is 64.3% water (35.7% solids) with 75.5 g/person/day generated through a disposer [Shpiner, 1997].
- Food waste moisture content is 60% with a production of 0.08 wet kg/person/day (0.048 dry kg/person/day) [de Koning, 1996].
- Average household food waste disposal is 260 g/person/day [Terpstra, 1995].
- Food waste is 30% dry solids (70% water) [Terpstra, 1995].

3.0 Food Waste Disposer Common Concerns

3.1 Water Use

- Disposers account for only about 1% of a household's daily use of water [Nakhla, 2014, P. 6].
- Estimate 1 gal/capita/day with disposer use [Nakhla, 2014, p. 6] [New York City Department of Environmental Protection, *Executive Summary*, 1997].
- Disposer water usage is 3-6 L/household/day [Karlberg, 2012].
- After installation of FWDs, the extra water consumption was marginal (less than 2% increase in water use) [Clauson-Kaas, 2011, p. 6].

- DeOreo et al, 2011 found that residential disposers save 13 gallons of water/household/day [DeOreo, 2011, p. 205].
- Based on grinding food waste in the laboratory, the water usage per capita added after the use of FWD increases only 4.45%, and the utility fee is only 0.02 Chinese Yuan (CNY) (per capita per day) [Tongji University, 2010, p. 59].
- The change in water use from FWDs is trivial [Evans, 2007, p. 23].
- After the introduction of FWDs, no water consumption changes were noticed [Imanishi, 2005, p. 14]. [Yoshida, “Impacts of Food Waste Disposers”].
- The use of disposers does not result in a noticeable increase in the volume of wastewater [de Koning, 1996].
- Nilsson et al, 1990 estimate that water consumption does not change because of disposer use [Karlberg, 1999].
- Increased water demand from disposers is 0.02% at 3% market penetration, and 0.24% at 38% market penetration (assuming a 1% disposer market growth per year). Therefore, no significant impacts on the city water supply from disposers are expected [New York City Department of Environmental Protection, *Executive Summary*, 1997].
- There is no statistical evidence that city water consumption has changed since the installation of disposers [New York City Department of Environmental Protection, 1997].

Table 2 – Water Consumption Rates. FWD Water Consumption Rates (L/person/day).

Source	FWD Water Consumption Rate (L/person/day)
Clauson-Kaas, 2011, p. 6	3-6
CECED, 2003	3-4.5
Kegebein, 2001	3-4.5
Cooperative Research Centre, 2000	2.95
Shpiner, 1997	1.01
de Koning, 1996	4.5
Terpstra, 1995	4.48
Waste Management Research Unit, 1994	4
Average	**3.99 (1.05 gallons/person/day)**

3.2 Electricity

- Assuming a FWD is used for 30 seconds per person daily with a power draw of 1000 W, the estimated power consumption for FWDs is 0.008 kWh/capita/day [Leverenz, 2013, p. 11].
- The power consumption for FWD is 0.119 kWh/capita/day, which equals a utility fee of 0.073 CNY/capita/day considering 0.617 CNY/kWh in Shanghai [Tongji University, 2010, p. 60].
- With electricity being roughly $0.10 per kilowatt-hour and a disposer using 2.3 uses per day with each use running for 30 seconds while the average disposer uses 500 watts while in use, the average cost is about $0.35 per year [Strutz, 2005].
- The Plumbing Foundation City of New York, 2001 indicates that using the upper time limit for disposer usage of 2 min/day and the most common 0.5 hp motor, the disposer consumes less than a 75W light bulb uses in 10 minutes [Marashlian, 2004].

Table 3 – FWD Energy Consumption Estimates and Price Estimates. Prices were calculated based on the average cost of electricity estimated at $0.10 per KWh. The energy consumption estimates range from less than 3 to 6 KWh/home/year, which is small in comparison to the average household energy consumption.

Source	FWD Energy Consumption Estimate (KWh/home/year)	Price (US$ per home per year)
Leverenz, 2013	4	$0.40
PE Americas, 2011	4	$0.40
Tendaj, 2008, p. 11	5-6	$0.60
Evans, 2007	2-3	$0.30
Balzonella, 2003	4.3	$0.43
Waste Management Research Unit, 1994	< 3	$0.30
Average	**4.1**	**$0.41**

3.3 Plumbing and Sewers

- Most food particles discharged from disposers are between 2mm and 4mm with a unimodal range of distribution [Nichols, 2019, p. 3].
- Settling velocity is always less than 0.1 m/sec, except for eggshells, which had maximum fall velocities of 0.13 m/sec [Nichols, 2019, p. 3].
- A study of 181 concrete pipes serving single family households comparing FWD usage with sewers revealed that FWDs have an impact on the use of sewers, but the majority of deposits were small, indicating that the impact of FWDs on sewer performance is minor [Mattsson, 2014, p. 1].
- The long-term impacts of FWDs on small diameter sewer systems of residential areas were shown to be minor [Mattsson, 2014, p. 1].
- More troubles aroused in sewers when households used food waste that was not compatible with FWDs, such as eggshells, which suggests the importance of proper education and use of FWDs [Mattson, 2014, p. 9].
- Many of the problems observed with the use of FWDs and sewers/plumbing could be avoided by having pipes with a steep inclination [Mattsson, 2014, p. 8].
- Deposits in pipes with large inclinations could be caused by sags in the pipes [Mattsson, 2014, p. 8].
- In a nine month study in PuDong, the use of FWDs did not result in sewer blockages or sedimentation [Tongji University, 2013].
- Long term impacts on sewer degradation is unknown. The Sustainable Food Waste Evaluation assumes a 5% aerobic and 10% anaerobic degradation [WERF, 2012, p. A31].
- Processing food waste will not increase sedimentation and blockages since the density of ground food waste usually has a lower specific density than waste water [Clauson-Kaas, 2011, p. 57].
- The PE Americas LCA assumes a negligible (0%) degradation [PE Americas, 2011, p. 113].

- The FWDs effect on the sewer system will be small [Tendaj, 2008, p. 40].
- Nilsson et al, 1990 showed that a stimulated optimal usage of disposers for 15 years did not exhibit operational problems within the plumbing system. Regular inspection and videotaping of the piping system found a buildup of sewage was reported at water level with a width of 2-3 cm along the envelope surface at a thickness of 0.5-1.5 cm [Marashlian, 2004].
- Some trouble could arise from increased O&G discharge in sewers. However, studies have shown that no problems were caused [Bolzonella, 2003].
- Sewage velocity is sufficient enough to maintain sewers clean. Generally, self-cleansing velocity is in the range of 0.5-1.6 m/s for sewers with a diameter of 200-2000 mm [Bolzonella, 2003].
- Study results revealed that only 16.8% of TS (from ground organic wastes) settled in sewers, whereas, the residual 83.2% reached the WWTP. Therefore, sewers should be considered a feasible way to transport food waste [Bolzonella, 2003].
- Another aspect to consider is to avoid disposer installations in areas where blockages or hydrogen sulfide formation already are problems in the sewage system [Karrman, 2001].
- A daily minimum flow velocity of 0.5 m/s is seen as sufficient for food waste transport free of sedimentation. The density and settling velocities of food waste particles is very much less in comparison to mineral particles [Kegebein, 2001].
- Increased costs in sewer maintenance (from disposers) cannot be ruled out. At 100% market penetration, a 20% increase could result [Kegebein, 2001].
- At a 50% market share, disposers contribute <0.1% flow to instantaneous maximum flow in sewer systems [CRC, 2000].
- At a 50% market share, disposers increase hydrogen sulfide generation in the sewerage system by 30% [CRC, 2000].
- Up to a 15% market penetration, the use of disposers in multi-unit dwellings would have a small impact on sewage collection systems [CRC, 2000].
- About 91% of solids in disposer effluent are <1 mm (0.25 in) in size, therefore, this small size would be unlikely to clog or become deposited in sewers or plumbing pipes [CRC, 2000].
- De Koning and Van der Graaf, 1996 state that the concern over grease and fats (from disposers) clogging sewers is invalid because the use of cold water causes grease and fat to congeal and attach to other food waste solids [CRC, 2000].
- There does not appear to be any sound evidence in literature to suggest that disposers cause clogging or deposits of solids in pipes [CRC, 2000].
- In a 1993 apartment disposer use study, sewer pipes were flushed and videotaped with no differences observed (i.e., no additional particle, sludge, or grease accumulation) after both 1 and 3 years following installation [Karlberg, 1999].
- Disposers may cause increases in TSS and Oil & Grease (O&G) in the sewer system. There may be an increase in sewer maintenance costs estimated at 0.61% at a 3% market penetration and 7.6% at a 38% market penetration (assume 1% market penetration per year) [New York City Department of Environmental Protection, 1997].
- In combined sewer systems built with an adequate self-cleaning velocity (ex., sanitary sewers 2.0-2.5 ft./sec or about 0.61-0.76 m/s and storm sewers 2.5-3.0 ft./sec), no additional deposits are expected due to ground food waste since its specific gravity of 1.01 is less than that of sewage (1.05), and much less than the suspended solids carried by

storm runoff (specific gravity 2.65) [New York City Department of Environmental Protection, 1997].

- In combined sewer systems, the introduction of disposers will cause increases in suspended solids of about 20% on a per capita basis, and expected to increase O&G discharges. As a result, combined sewer systems with insufficient self-cleaning velocities will require routine cleaning, which will increase maintenance costs [New York City Department of Environmental Protection, 1997].
- Videotaping done before and after the study detected no noticeable deposits of solids build-up. Therefore, no potential significant adverse impacts on the sewer system are expected from disposer use [New York City Department of Environmental Protection, *Executive Summary*, 1997].
- Wicke, 1987 states that a concentration of less than 1% solids (10,000 mg/L) will not cause an increase in solid sedimentation, or for every 12 gal of water (45 L) there should be no more than 1 lb. (454 g) of ground garbage [Shpiner, 1997].
- There is no literature example to prove that the use of disposers causes clogging or deposits in sewers. Most food solids have a density about equal to water and are easily suspended in water. Thus, it is unlikely that ground food waste contributes to sewer clogging [de Koning, 1996].
- Discharged with cold water, any grease or fat found in food waste will congeal and attach itself to the other ground waste particles. Running cold water will prevent coating of the sewer with grease [de Koning, 1996].
- Disposers pose negligible impacts on water and wastewater infrastructure [Jones, 1990, p. 14].

Table 4 – Cleaning Velocity. Cleaning Velocities/Minimum Flow Velocities

Source	**Cleaning Velocity/Minimum Flow Velocity (m/s)**
Evans, 2007	0.48-0.9
Bolzonella, 2003	0.5-1.6
Kegebein, 2001	0.5
New York City Department of Environmental Protection, 1997	0.61-0.76
Average	**0.94**

3.4 Wastewater Treatment Plant Impacts

3.4.1 Pollutant Loading

- Figures 1-3 show estimated loading increases in TSS, COD, BOD, oil and grease, potassium, total P, inorganic P, organic P, TKN, organic N, and NH_3.

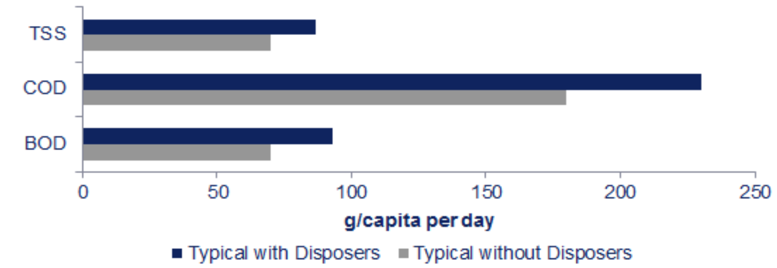

Figure 1 – Estimated Loading Increase from Disposers (TSS, COD, and BOD) [Metcalf, 2014]

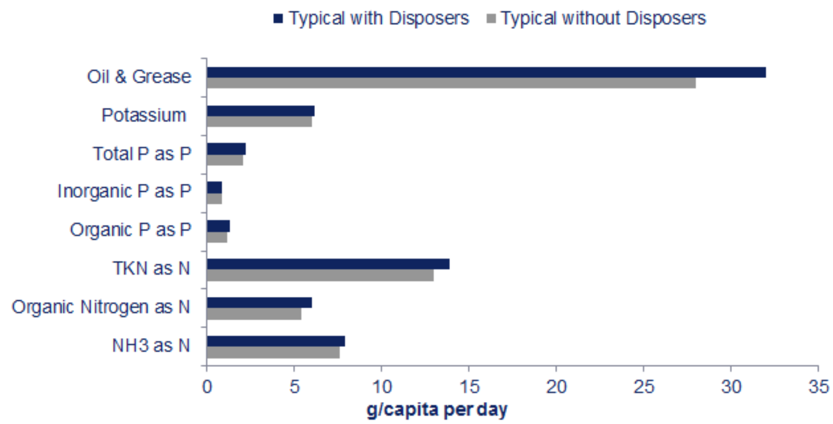

Figure 2 – Estimated Loading Increase from Disposers [Metcalf, 2014]

	%Increase
TSS	33
COD	28
BOD	24
Oil & Grease	4
K	11
Total P	7
Inorganic P	8
Organic P	0
TKN	5
Organic N	3
NH_3	14

Figure 3 -- Estimated Percent Increase in Loading from Disposers [Metcalf, 2014]

- The experimentally determined COD and BOD of food wastes were 22% lower than the theoretical values, suggesting that the impact of food wastes on the WWTP is lower than originally supposed [Nakhla, 2014, p. 11].
- Installing FWDs will not affect hydraulic load of the WWTP [Clauson-Kaas, 2011, p. 58].
- WWTPs are designed to treat biodegradable material suspended in water, i.e. similar to the output of FWD [Evans, 2007, p. 4].
- Additional pollutant loading due to disposer use is 66 g/person/day BOD, 60 g/person/day TSS, 2.1 g/person/day TKN, 0.3 g/person/day P, and 2.5-5% biosolids [de Koning, 2004].
- The effect of disposers on WWTP processes is very limited [de Koning, 2004].
- At a 15-20% disposer market share, loadings do not result in significant variations in the characteristics of sewage. At a 20-35% disposer market share, an increased WWTP system energy consumption is observed due to greater respiration of the active biomass and a larger production of excess biosolids. Beyond a 35-40% disposer market share, additional works must be done at the WWTP. European Union (EU) market levels will not exceed 15% in 25-30 years, thus, normal WWTP upgrades will allow for an accommodation of increased disposer loading [CECED, 2003].
- Disposer discharge to a WWTP equates to 73 g/person/day dry matter, 25 g/person/day BOD, 0.25 g/person/day phosphorus, and 1.3 g/person/day nitrogen [Karrman, 2001].
- At 100% disposer market share, additional loadings from disposers are 3-5% for flow, 5-10% for screenings, 5% for grit, 10-25% for BOD, 40-60% for TSS, 5-10% for TKN, 7-14% for P, 50-70% for primary sludge, 10-40% for waste activated sludge, 30-50% for digested sludge, and 90-100% for biogas [Rosenwinkel, 2001].
- The additional loads for wastewater treatment and sludge digestion can be estimated very well and, due to slow market penetration, will not lead to uncontrolled overloading to the WWTP “overnight” [Rosenwinkel, 2001].
- At a 50% disposer market share, increases in sewage flows are very small (additional 0.5% to the mean average daily flow) [CRC, 2000].
- At a 15% disposer market share, no operational problems should be caused in terms of BOD, TSS, or O&G loadings [CRC, 2000].
- Up to a 15% disposer market share, the use of disposers in multi-unit dwellings would have a small impact on sewage treatment systems. Beyond this figure are increasing impacts, with potentially significant impacts at a 50% market share. However, this level of market share is unlikely in the near future [CRC, 2000].
- No operational problems are expected for market levels up to 15% in regard to BOD and O&G loadings, or up to 20% market for additional TSS loadings [CRC, 2000].
- Up to a 50% disposer market share, the transport and treatment of disposer effluent would require an additional 0.5% energy, and total WWTP costs would increase 0.5% [CRC, 2000].
- Up to a 50% disposer market share, additional loadings from disposers are <1% for TSS and nutrients, and <2% for BOD [CRC, 2000].
- At a 100% disposer market share, flows would increase 0.4%, biosolids production would increase 18.1%, BOD would increase 16.5%, and nutrients would increase 3.0% for TKN and 4.6% for P [Waste Management Research Unit, 1994].

3.4.2 Preliminary Treatment

- Using FWD to divert food waste results in a waste stream that is fairly free from contaminants and debris, so it is not subject to additional processing, cleaning, and preliminary treatment at the WWTP [Leverenz, 2013, p. 10].
- It is expected that food waste will contain no grit [Hernandez, 2002, "Hyperion Digestion Pilot Program"].
- With disposer usage, WWTP screens and grit chambers will only be affected to a small extent [Rosenwinkel, 2001].
- Screenings are not expected to be added by food waste disposers [New York City Department of Environmental Protection, 1997].
- Grit was assumed to be 5% of TSS. A method to evaluate scum or grit production impact could not be determined [New York City Department of Environmental Protection, 1997].

3.4.3 Primary Treatment

- Fractionation of food waste was 40% soluble, 0% colloidal, 60% particulate of the total COD; N and P were predominantly in the particulate form; settling over 3 hours removes 59-62% of TSS, 46-53% of BOD_5, and 49-56% of COD [Chowdhury, 2016, p. 664].
- The large particulate fraction of FW tends to be removed in primary sedimentation while the soluble fraction of FW in primary effluent can be utilized for nutrient removal [Kim, 2015, p. 68].

Table 5 – Particulate Fractions. Particulate Fractions in 50 grinded food waste samples [Kim, 2015, p. 69]

Parameter	Particulate Fraction
COD	58%
BOD_5	67%
TN	74%
TP	100%

Table 6 – Percent Removal. Removal percentages of TSS, BOD_5, and COD after a 3 hour (180 minute) time period [Nakhla, 2014, p. 11]

Parameter	Percent Removal after 3 hours
TSS	59-62%
BOD_5	46-53%
COD	49-56%

- During primary sedimentation, 80% of the solids and 90% of ground food waste are removed [PE Americas, 2011, p. 26].
- FWD use increases COD and TSS by 12% and 24% respectively, still in the allowable range for municipal sewers [Tongji University, 2013].

- The use of FWDs increases the COD in the sewage, as well as the C/N and C/P ratios, therefore sewage treatment will benefit with improved biological nitrogen and phosphorus removal [Tongji University, 2013].
- Thomas's study in 2010 noted that when the food waste settled in buckets (in order to simulate primary clarification), the results indicated roughly 62% TP and 90% ammonia were in the supernatant while 77% and 90% of the TSS and RSS were in the sediment fractions [Thomas, 2010, p. 7].
- Battistoni, 2007 did not find any solid sedimentation [Battistoni, 2007, p. 896].
- Primary settling food waste removal is 20% BOD, 90% TSS, 5% TKN, and 10% P [de Koning, 2004].
- The average settling velocity of food waste is 13.2 m/hr. (43.3 ft./hr.) [Bolzonella, 2003].
- According to lab experiments, 75% (of disposer food waste) is assumed to be settled in the pre-sedimentation step [Karrman, 2001].
- With disposer usage, most of the particulate food waste fraction will settle in the WWTP primary clarifier [Rosenwinkel, 2001].
- Disposer solids settle readily under gravity. Sinclair Knight, 1990 state that the addition of disposer solids enhances the settling characteristics of sewage [CRC, 2000].
- The portion of BOD from disposer use that does not settle in primary treatment was determined using filtrate BOD. The portion of BOD from food waste that settled was 68.7% [New York City Department of Environmental Protection, 1997].
- According to literature, over 90% of food waste is removed in primary sedimentation [Shpiner, 1997].
- Brillet et al., 1986 reported that sedimentation removed 80% BOD and 90% TSS from disposer waste [Shpiner, 1997].
- Nilsson et al, 1990 reported that 75% of TS in wastewater and 90% of solids from disposer grinding are removed in primary sedimentation, thus, overall removal is 80% [Shpiner, 1997].
- Normal wastewater TSS removal is 58-64% and the food waste mixture TSS removal is 78-86% [Shpiner, 1997].
- The majority of additional BOD/COD and nutrient from disposer loading is concentrated in settled primary sludge [de Koning, 1996].

3.4.4 Secondary Treatment (Biological Treatment)

- The additional soluble food waste fraction will lead to higher BOD/COD loading within the biological treatment steps, which on one hand will cause a higher oxygen demand, but on the other can serve as a cheap and continuously available carbon source (for nutrient reduction). A basic condition for the appropriate biological nitrogen and phosphorus removal is a sufficient supply of easily degradable substrate (i.e., carbon) [Rosenwinkel, 2001].
- At a 25% disposer market share, influent BOD would increase 12%, TKN and P would increase 2% [Karlberg, 1999].
- After a decade of city-wide disposer distribution, costs would increase $4.1M for the most expensive N-control measure (a 0.27% increase). This represents a *de minimis*

impact [New York City Department of Environmental Protection, *Executive Summary*, 1997].

- Brillet et al., 1986 reported that at a 100% disposer market share, biological treatment loading increased 9.5-16% BOD and 7.5-10% TSS [Shpiner, 1997].
- Increased loading to the biological processes from disposer usage is negligible (at 10% market share) [de Koning, 1996].

3.4.5 Anaerobic Digestion and Food Waste Energy Recovery

- Food waste sent to plants with AD and biological nutrient removal results in a net energy gain, lower nitrogen and phosphorus in treated effluent, and lower overall costs for treatment [Kim, 2019, p. 358].
- The efficiency of converting the potential chemical energy contained in food waste to electrical energy is estimated to be about 20% [Leverenz, 2013, p. 17].
- Typical observed values for biogas yield from various food waste digestion studies are 157 and 600 m^3/MT for a wet and dry basis, respectively [Leverenz, 2013, p. 18].
- The power generated from biogas is derived to be approximately 80 kWhe/d [Leverenz, 2013, p. 37].

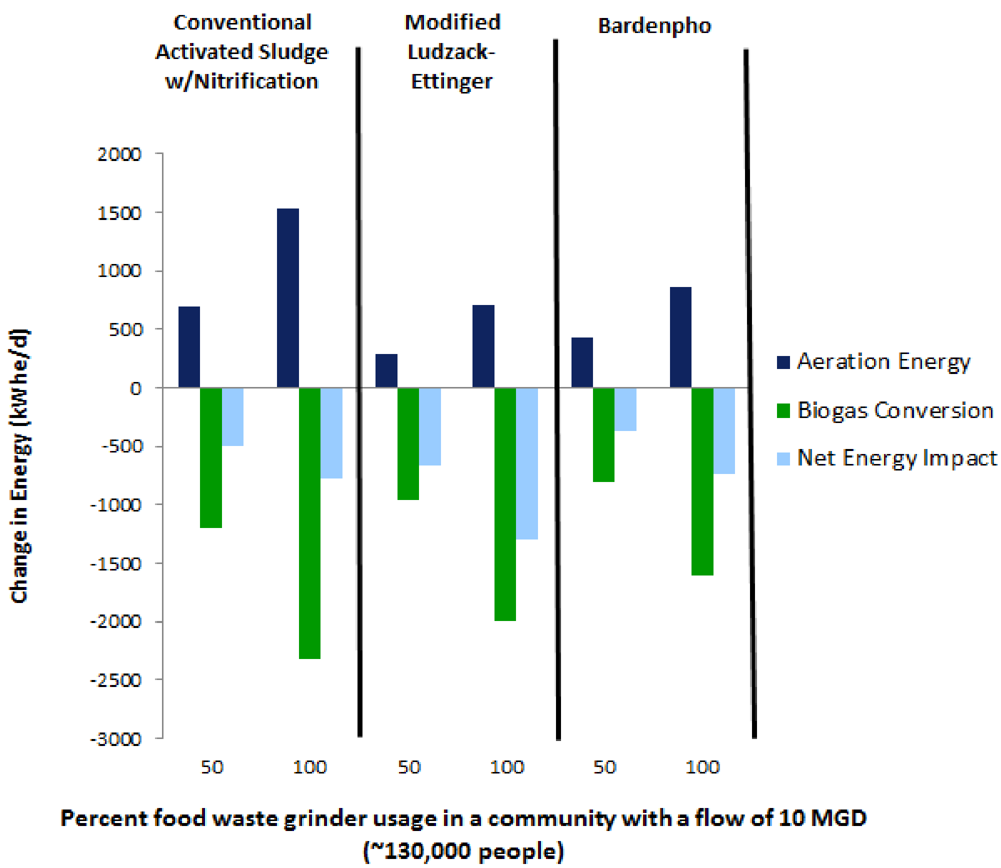

Figure 4 –Change in Energy. Change in Energy in three different wastewater treatment processes at 50 and 100% disposer usage [Leverenz, 2013].

- In a comparison of five studies regarding anaerobic digestion, four found an increase in production of biogas and one study found that BNR was enhanced as the carbon to nutrients ratio increased after FWD were introduced [LGA, 2012, p. 23].
- Upon reviewing 82 studies regarding end of life management methods for source separated organics, it was found that anaerobic digestion and aerobic composting have a lower climate impact than waste-to-energy and landfilling [Morris, 2012, p. 6].
- The town of Surahammar, Sweden had a 0-50% increase in market penetration rate in a ten year period. Digesters produced 46% more biogas after FWDs were installed [Evans, 2010, p. 1].

Table 7 – Added Sludge, Biogas, and Electricity. Various penetration rates' effects on sludge volume, biogas volume, and electricity gain [Tongji University, 2010].

Penetration Rate	Added Sludge Volume (t/d, 80% water content)	Added Biogas Volume (m^3/d)	Electricity Gain (kWh/day)
1%	18	493	68
5%	91	2465	4437
10%	182	4930	8875
100%	1818	49304	88748

- At a disposer market share of 10%, biogas production increased about 3% [Tendaj, 2008, p. 40].
- Biogas production from food waste is 1.15 m^3/day of digested organics with a content of 22,000 kJ/m^3 of biogas [de Koning, 2004].
- The use of disposers will increase electric self-supply from 72% (at 0% disposer market share) to 82% (at 10% market share). Profits gained in electrical supply will cancel out additional biosolids treatment costs [de Koning, 2004].
- Food waste (with 90% settling in primary treatment sludge) contains a high percentage of easily digestible organics (i.e., 80% VS) [de Koning, 2004].
- The potential energy value from food waste by anaerobic digestion was assumed insignificant [Marashlian, 2004].
- At a 60% disposer market share, an increase of additional energy potential due to anaerobic digestion in the range of 54-73% was observed [Bolzonella, 2003].
- Food waste Volatile Solids Destruction (VSD) is 83.7% (for thermophilic digestion at 55°C and food waste fruit and vegetables ground to a slurry) [Hernandez, 2002, "Hyperion Advanced Digestion Pilot Program"]
- The optimum digester operating temperature was found to be 55°C and 57°C (thermophilic digestion). As the temperature increased from that point, VSD and gas production decreased and volatile acids increased [Hernandez, 2002, "Los Angeles Digesters"].
- The value of the biogas produced from food waste anaerobic digestion appears to exceed the cost of processing the food waste and disposing of the residual biosolids (based on a LAX Airport proposal to divert 8,000 tons/year of bulk food waste) [Hernandez, 2002, "Los Angeles Digesters"].
- Methane gas generated in the anaerobic digesters is transported to a city-owned power generation steam plant, which is used as a supplemental fuel and burned in the production

of steam and electrical energy (15 scf of digester gas produced per lb. of VS destroyed). In digesting fruits and vegetables only, the value of the biogas appears to exceed the cost of processing the food waste and disposing of the biosolids [Hernandez, 2002, "Los Angeles Digesters"].

- The food waste disposer system generates more energy than consumed through the digestion (biogas) [Karrman, 2001].
- Food waste fermentation (anaerobic digestion) has an energy potential of 300 MJ/person/yr., which contributes about 25 kW-h/person/yr. to electric supply (about the electrical usage of 1 WWTP) [Kegebein, 2001].
- As most of the food waste from disposers settles in the WWTP primary clarifier, the majority will reach the anaerobic digester and cause an increase in biogas production and a regenerative energy source [Rosenwinkel, 2001].
- Diverting food waste through a disposer to a WWTP should be encouraged when solids handling systems are adequate, methane is combusted (through anaerobic digestion) to produce energy, and effluent and/or sludge (biosolids) are returned to soil. Food waste is effectively being recycled [Diggelman, 1998].
- Additional gas production is generated from the volatile portion of food waste loading (7 ft^3 of gas is produced per lb. of VS that enter the digester) [New York City Department of Environmental Protection, 1997].
- Anaerobic digester gas production averaged 346 m^3/day before disposer usage and 417 m^3/day after disposer usage for an increase of 20.2% (at 65% methane, this equates to 160,000 kW-h/yr.) [de Koning, 1996].

3.4.6 Biosolids Handling and Disposal

- Biosolids have a chemical formula: $C_5H_7NO_2$ with a carbon content of 53.1% and a nitrogen content of 12.4% by mass [PE Americas, 2011, p. 30].
- For every 100 kg wet food waste, there are approximately 7.3 kg biosolids for the conventional treatment with anaerobic digestion[PE Americas, 2011, p. 30].
- Concerns about increased biosolids generation persist, and its potential environmental and economic implications may differ with location [Marashlian, 2004].
- Disposer usage showed minimal to no impact on the WWTPs total biosolids production and handling processes as the high VSD from food waste yielded a minimum amount of solids in the residue [Hernandez, 2002, "Hyperion Advanced Digestion Pilot Program"].
- Bench-scale jar testing showed food waste dewaters easily and used less polymer than primary sludge/thickened waste activated sludge [Hernandez, 2002, "Hyperion Advanced Digestion Pilot Program"].
- Food waste appears to possess a natural settling capability [Hernandez, 2002, "Los Angeles Digesters"].
- Before disposers are installed in large scale a long-term solution for the use of sludge should be agreed, because disposers will increase sludge production [Karrman, 2001].
- It is unlikely that biosolids produced by disposer usage would affect the contaminant level or reuse options of biosolids [CRC, 2000].
- Ground food waste will significantly increase the quantity of biosolids, however, Nilsson et al, 1990 notes that these biosolids will decompose easier than regular wastewater biosolids and more gas can be produced [Shpiner, 1997].

Table 8 – Disposer Market Share Effects. Various sources reporting on effects at certain disposer market share values are presented.

Source	Disposer Market Share	Effect
de Koning, 1996	10%	Solids to thickeners and digesters increase of 5%
Terpstra, 1995	10%	5% more biosolids
CRC, 2000	25%	No adverse effects to solids processing
Karlberg, 1999	25%	Sludge volume increase of 4%
Karrman, 2001	50%	Sludge increase is 7.2%, a 10% increase compared to no FWD use
Terpstra, 1995	100%	50% more biosolids

3.4.7 Effluent Characteristics

- The soluble and colloidal fraction of food waste that passes through primary treatment has a positive impact on the removal of nitrogen and phosphorus from wastewater [Leverenz, 2013, p. 6].
- The effluent total N was determined to range from 8.9 to 13.0 mg/L, decreasing as the percent of FWDs in use increased [Leverenz, 2013, p. 25].
- The effluent total Phosphorus was determined to be around 6.7-7.0 mg/L, decreasing as the amount of FWDs in use reached 100 percent [Leverenz, 2013, p. 25].
- Compared to no FWD usage, FWD usage can increase TN removal by 7 to 12 percent for the biological nutrient removal (BNR) process with 50 and 100 percent FWD usage, respectively, and TP removal could be increased by 52 to 74 percent in the BNR process with 50 to 100 percent FWD usage, respectively [Leverenz, 2013, p. 30].

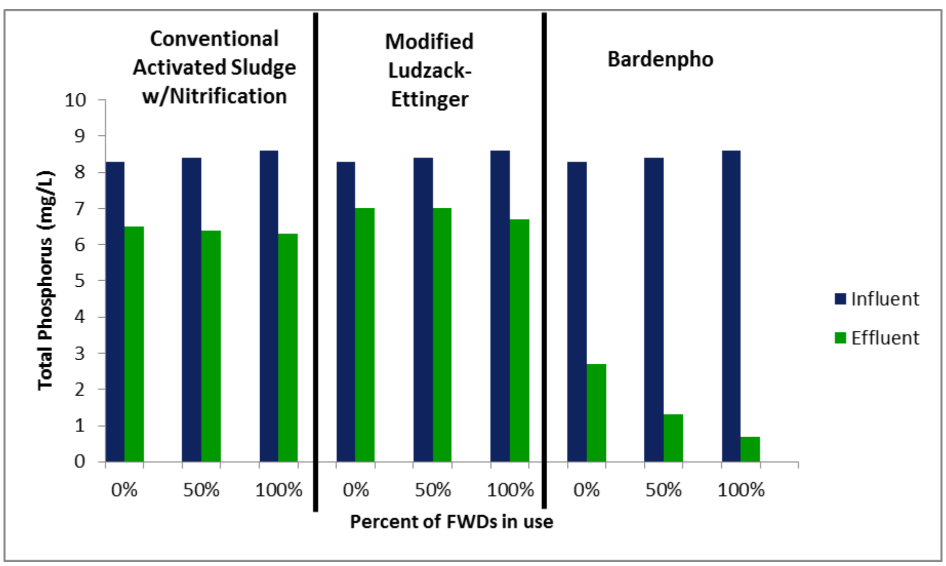

Figure 5 – Change in total Phosphorus. The influent and effluent TP in three wastewater scenarios [Leverenz, 2013].

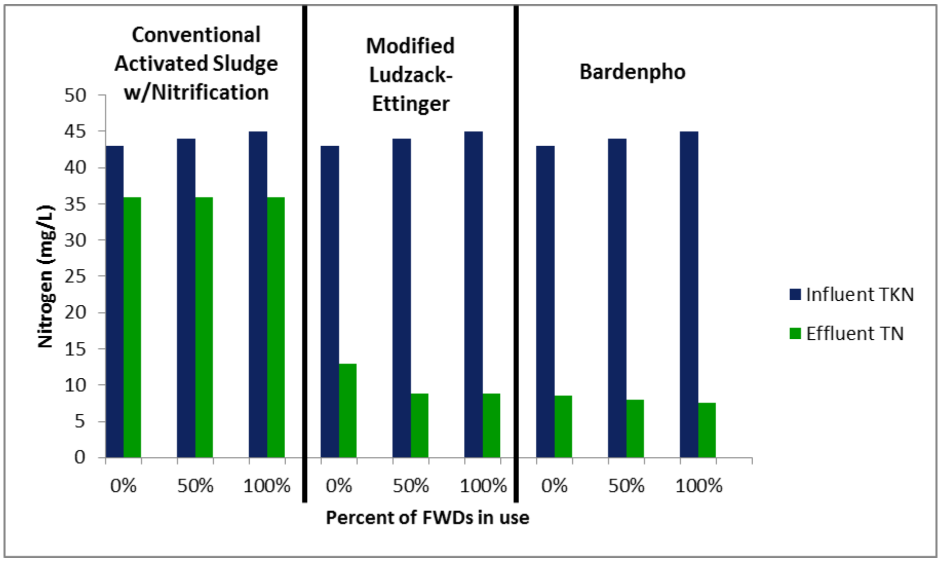

Figure 6 – Change in Nitrogen. The influent (TKN) and effluent (TN) in three wastewater scenarios [Leverenz, 2013].

- Both the COD removal and total nitrogen removal increased. The rbCOD/COD ratio increased from 0.20 to 0.25 and the COD/TN ration increased from 9.9 to 12 with a specific denitrification rate of about 0.06 kg NO_3-N/(kg MLVSS day) [Battistoni, 2007, p. 893].
- The COD and total nitrogen removal increased, creating a denitrification efficiency of 85% and a 39% reduction of energy requirements [Battistoni, 2007, p. 893].
- The composting system does not impact the waterborne wastewater system, while the food waste disposer system is estimated to cause some minor increases in discharges of nutrients and heavy metals to water. All impacts in both systems are rather small [Karrman, 2001].
- At a 50% disposer market share, disposers are unlikely to affect biosolids reuse, the marine environment, or energy consumption [CRC, 2000].
- The BOD increase in the effluent due to disposer usage equates to a 0.01 mg/L dissolved oxygen decrease in New York Harbor in 10 years (*de minimis* impact) [New York City Department of Environmental Protection, *Executive Summary*, 1997].
- Combined sewer overflow total stream BOD concentration increased 5% and TSS 2% over baseline from disposer usage. In the worst case area, the 4 mg/L minimum dissolved oxygen standard was exceeded by 1.5% over the baseline. This increase is considered to be *de minimis* [New York City Department of Environmental Protection, *Executive Summary*, 1997].
- At a 20% disposer market share, effluent quality can be maintained through operative WWTP adjustment. A higher market share will necessitate plant expansion, but will take many years to occur [Shpiner, 1997].

3.5 Fats, Oils, and Greases

- FOG accumulation in sewer lines has become a global challenge for the management and sustainability of sanitary sewer systems [He, 2017, p. 1212].

- FOG deposits are formed through saponification between calcium and FFAs and aggregation of excess calcium or FFAs [He, 2017, p. 1212].
- A small number of FOG deposits were found in pipes connected to upstream FWDs. However, FWDs were not the major contributors to the formation of FOG deposits [Mattsson, 2014, p. 8].
- Williams, 2012 research identified two possible mechanisms that may affect the formation and properties of FOG deposits in sewers [Williams, 2012, p. 6327]
  - Transformation of fatty acids from unsaturated to saturated form [Williams, 2012, p. 6327].
  - Biocalcification where higher levels of water hardness lead to harder deposits with higher melting points [Williams, 2012, p. 6327].
- An introduction of FWDs will probably not lead to a significant increase in fat in the sewer system considerably [Clauson-Kaas, 2011, p. 57].
- FFAs are likely to react with calcium ions by means of van der Waals forces or electrostatic repulsion (DLVO theory) [He, 2011, p. F].
- The primary lipid reacting in the FOG deposits was palmitic acid ($C_{16}H_{32}O_2$). Other lipids commonly found include oleic ($C_{18}H_{34}O_2$) and linoleic acid ($C_{18}H_{32}O_2$) [He, 2011, p. C].
- Deposits are likely formed primarily from free fatty acids (FFAs) reacting with ions such as calcium [He, 2011, p. F][Keener, 2007, p. 2241].
- The saturated fats and calcium levels in the FOG deposits are higher than background levels, suggesting that a chemical process is responsible for deposit formation [Keener, 2007, p. 2246].

3.6 Septic Systems

- People in rural settings should be able to have the same appliances as those living in cities, including garbage disposals, but proper design of the septic system is important [Seabloom, 2004, p. 41].
- The increase of the organic strength of the wastewater from garbage disposals may have an impact on the performance of the septic tank [Seabloom, 2004, p. 41].
- In-sink garbage disposal devices increase scum accumulation by approximately 34 percent but increase sludge only 2 percent [Seabloom, 2004, p. 41].
- It is unclear whether septic tanks must be larger to accommodate garbage disposals [Seabloom, 2004, p. 41].
- COD increase in the effluent may have an impact on the performance of the septic system [Lin, 2019, p. 7].
- Increases in TN and TP were minimal [Lin, 2019, p. 7].
- FW substantially increased the depth and volume of the scum layer in the experiment tank [Lin, 2019, p. 7].
- FW more biodegradable and accumulates more in the scum layer [Lin, 2019, p. 7].
- Addition of FW on septic performance and pumping frequency will be insignificant or negligible [Lin, 2019, p. 7].

4.0 Alternative Management Comparisons

- Integrating disposers in a developing economy with a high fraction of food waste is a viable option to reduce emissions for carbon trading; carbon emissions are reduced by 42%; cost savings of ~28%; economic savings attractive even after adding wastewater and sludge management costs [Maalouf, 2017, p. 461].
- New disposer owners typically process 30% of their food waste with disposers [InSinkErator, 2016, p. 2].
- Using a disposer in combination with advanced wastewater treatment results in the lowest primary energy demand and a lower global warming potential as compared to alternative food waste management methods [InSinkErator - *Executive Summary* “Systems for the Management and Disposal of Food Waste,” 2011, p. 2].

4.1 Landfills

- According to the EPA, after MSW has been recovered by recycling and composting, food waste is the largest component of MSW discarded to the landfill in the US at 21.1% [EPA, 2013, p. 7].
- FWDs would decrease the amount of moisture in the garbage by 10%. This reduction of moisture in the garbage results in a 16% higher calorific value for incineration from 15,345 kJ/kg to 17,783 kJ/kg [Tongji University, 2013].
- In a study in PuDong, the use of FWDs reduced the amount of wet waste in garbage by 10%. Based on current waste generation rates, this could equate to a reduction of over 1000 tons per day in Shanghai [Tongji University, 2013].
- If organic waste is removed from garbage collection, the amount of garbage is reduced by approximately 20-30%, which also results in fewer odor problems and better hygiene for workers who collect the garbage [Clauson-Kaas, 2011, p. 59].
- Food waste is the single largest component of municipal solid waste sent to landfills and many communities worldwide are focusing efforts to divert this organic waste in order to reduce greenhouse gas emissions at landfills [InSinkErator - *Executive Summary* “Systems for the Management and Disposal of Food Waste,” 2011, p. 2].
- Using a disposer in conjunction with any of the eight wastewater treatment systems results in lower global warming potential than alternative landfilling options [InSinkErator - *Executive Summary* “Systems for the Management and Disposal of Food Waste,” 2011, p. 2].
- Using a wastewater treatment route rather than a landfill in an area with 30,000 households would result in a carbon footprint reduction of 1.9 million kg, which is the equivalent of not driving 4.6 million miles [InSinkErator - *Executive Summary* “Systems for the Management and Disposal of Food Waste,” 2011, p. 2].
- A study in Surahammar, Sweden, reported that the waste diverted to landfills decreased from 3600 tons/year in 1996 to 1400 tons/year in 2007, after the installation of FWDs increased from 0-50% [Evans, 2010, p. 1].
- Characteristics of Collected Garbage in landfills as compared to cases with no FWD use [Yang, 2009, p. 17]
  - Dry ratio of food waste declined by more than half [Yang, 2009, p. 17]
  - Moisture content decreased at least by half [Yang, 2009, p. 17]
  - Combustible matter increased about at least 1.7 times [Yang, 2009, p. 17]
  - Lower heating value increased about at least 2.0 times [Yang, 2009, p. 17]

- Lower heating value of flammable garbage collected after FWD installation increased to 12,500 kJ/kg [Yang, 2009, p. 23].
- Raising the lower heating value leads to less fuel needed for the incineration process of waste [Yang, 2009, p. 23].
- Raising the lower heating is more suitable for use as a solid fuel, rather than being solid waste [Yang, 2009, p. 23].
- Flammable garbage with a higher lower heating value (greater than 12,560 kJ/kg) and lower moisture content is best for use as a solid fuel, and the presence of food waste disposers provides these conditions [Yang, 2009, p. 23].
- According to Yang (2007), through field surveys regarding the use of FWDs in a village of 327 people with a 97% disposer penetration, the average reduction rate of garbage being sent to landfills was 31% [Yang, 2009, p. 24].
- Using FWDs causes a decrease in the generation rate of garbage being sent to landfills, so a cost-savings benefit in terms of garbage collection and transportation can be obtained and materials recycling and thermal energy recovery will become easier [Yang, 2009, p. 24].
- In this study, the introduction of food disposers into the waste and wastewater management systems led to net economic benefits that ranged between 7.2% and 44.0% of the current solid waste management cost. Food waste disposers can constitute a viable option (economically and environmentally) that could reduce the load on the solid waste stream and minimize the amount of end waste requiring landfilling [Marashlian, 2004].
- The neuslavage study shows increased upper respiratory infections for garbage collectors than supervisors related to microbiological exposure during work [CRC, 2000].
- The Department of Sanitation recognizes the potential of disposers to make a positive impact in New York City residential waste management. Benefits include reduced odors and pest attraction, and better separation of recyclables [New York City Department of Environmental Protection, 1997].
- At a 38% disposer market share, grinding 50% of the food waste through disposers will save $4 M in solid waste export costs [New York City Department of Environmental Protection, *Executive Summary*, 1997].

4.2 Composting and Curbside Food Waste Collection

- FWD can be used in conjunction with curbside food waste collection (CFWC) to maximize the overall diversion. Therefore, FWD and CFWC are complementary, rather than competing technologies [Leverenz, 2013, p. 4].
- FWD eliminates the need for additional processing at the wastewater treatment plant, unlike CFWC. CFWC requires additional processing besides grinding to eliminate the additional debris and contaminants in the food waste [Leverenz, 2013, p. 10].
- Total cost for composting facility is estimated at $40 per ton [WERF, 2012, p. A23].
- Based on several steps for the compost alternative, such as transportation to compost facility, ventilation, handling, turning, composition, land application, application as fertilizer the total CO_2 emissions for composting is 1050 tons/year [WERF, 2012, p. A25].
- The GWP for composting is -14 $kgCO_2e/tKFW$ whereas the GWP for using a FWD in conjunction with AD is -168 $kgCO_2e/tKFW$ [Evans, 2007, p. 4].

- FWDs are not meant to discourage composting. Rather, FWDs can be seen as a convenient and hygienic method to divert food waste from landfills [Evans, 2007, p. 5].
- The main difference between using a FWD with AD and composting is that using a FWD creates energy (renewable fuel from the CH_4), whereas the composting method consumes energy [Evans, 2007, p. 42].
- Food waste collection followed by anaerobic digestion and biogas utilization in power plants has been judged more positive than separate collection followed by composting [de Koning, 2004].
- The food waste disposer is designed to grind only food waste. Materials other than food waste (ex., bottle caps, textiles, etc.) will lead to device jamming. Thus, the disposer is a natural selector of food waste. In contrast, composting largely depends upon the education and goodwill of participants as to the quality of collection [CECED, 2003].
- The food waste disposer system appears to be slightly less costly than central composting when only the costs for water and refuge handling are considered, and the user pays for the purchase and installation of the disposer themselves [Karrman, 2001].
- The food waste disposer alternative causes 3 times less global warming than the composting alternative, due to the reduction of truck transport [Karrman, 2001].
- Concentrations of bacteria and molds that can interfere with human health and wellbeing are greater when there are organic waste buckets and bins used for composting purposes [CRC, 2000].
- Most people are unwilling to separate food scrap for Department of Sanitation pickup [New York City Department of Environmental Protection, 1997].
- Home composting produces a high strength (BOD) leachate when food waste is present. There is no readily available mechanism to manage this leachate [Waste Management Research Unit, 1994].
- Methane has a much greater greenhouse effect (on the environment) than the equivalent of carbon dioxide. Environmentally, therefore, it is desirable to minimize methane release. There is no readily available mechanism for achieving this with household composting. In contrast, landfills and sewage treatment works can be constructed to maximize methane recovery as a fuel [Waste Management Research Unit, 1994].

5.0 Life Cycle Analyses

5.1 "Sustainable Food Waste Evaluation"

Water Environment Research Foundation (WERF)

The Life Cycle Analysis is a comparison of five systems for the processing wastes based on a representative community of 100,000 people in North America. The five systems are:

1. Mixed Material Recovery Facility (MRF)
2. Landfill
3. Wastewater Treatment Plant (WWTPP)/Hauled
4. Composting
5. Wastewater Treatment Plant (WWTP)/Sewered

The LCA analyzed capital and operating costs, carbon footprint, space footprint, labor demands, diesel fuel demand, electricity demand, and water demand for each of the five systems.

Capital and Operating Costs: The costs of the five systems ranked from highest to lowest cost are: Mixed material recovery facility, landfill, WWTP/hauled, composting, and WWTP/sewered.

Carbon Footprint: The carbon footprint (CO_2e) from the five food waste management options ranked from highest to lowest is: landfill, mixed MRF, WWTP/sewers, compost, WWTP/hauled. The lower carbon footprint in the WWTP/hauled method is probably due to the electricity generation from biogas produced by the digesters. The WWTP/sewered alternative has a relatively high carbon footprint, probably due to the uncertainty of the methane released in the sewers. Additionally, since there is little information known about the anaerobic decomposition that occurs in the sewers, so the CO_2e emissions for the WWTP/sewered alternative provides a large degree of uncertainty.

Space Footprint: The space footprint, or area requirements, of the alternative methods ranked from highest to lowest are: composting, landfill, mixed MRF, WWTP/hauled, WWTP/sewers.

Diesel Fuel Demand: The diesel usage for the five systems ranked from highest usage to lowest is: composting, mixed MRF, landfill, WWTP/hauled, WWTP/sewered.

Water Demand: The water demand for the five systems ranked from highest water demand to lowest water demand (measured in Mgal/year) is: WWTP/sewers, WWTP/hauled, mixed MRF, composting, landfill.

Overall, the use of a FWD has the lowest cost of all other alternatives studied, a small space footprint, and low diesel requirements. However, the use of a FWD does require water. There is a greater electricity use to account for the aeration necessary to process the addition of the food waste, but there is also additional energy production due to anaerobic digestion.

[WERF, 2012]

5.2 "Life Cycle Assessment of Systems for the Management and Disposal of Food Waste"

PE Americas

The Life Cycle Analysis (LCA) is a comparison of a total of twelve end-of-life disposal options, including two landfilling options, eight wastewater treatment options that occur in conjunction with a FWD, one incineration system, and one composting system. The twelve systems are:

1. Landfill with Generation
2. Landfill with Flare
3. Extended Aeration
4. Extended Aeration/Landfill
5. Extended Lime/Slab
6. Conventional Treatment/Incineration
7. Conventional Lime Slab
8. Conventional Treatment/Anaerobic Digestion/Flare
9. Conventional Treatment/Anaerobic Digestion/Boiler
10. Conventional Treatment/Anaerobic Digestion/Cogeneration
11. Incineration
12. Composting

This LCA found that using a disposer in conjunction with any of the eight wastewater treatment systems results in lower global warming potential than both landfilling options. Additionally, using a disposer in combination with advanced wastewater treatment results in the lowest primary energy demand as compared to the landfill systems as well as the waste-to-energy and emissions-controlled composting systems.

[PE Americas, 2011]

5.3 "Life-Cycle Comparison of Five Engineered Systems for Managing Food Waste"

Dr. Carol Diggelman, University of Wisconsin

The Life Cycle Analysis (LCA) is a comparison of five systems for the processing of 100 kg of food waste. The five systems are:

1. Food Waste Disposer/Wastewater Treatment Plant (FWD/WWTP)
2. Municipal Solid Waste Collection/Landfill (MSW/L)
3. Municipal Solid Waste Collection/Compost (MSW/C)
4. Municipal Solid Waste Collection/Incineration (MSW/I)
5. Food Waste Disposer/On-Site Septic System (FWD/OSS)

The LCA analyzed land requirements, total system energy, total system materials, total emission to the environment, and total system cost for each method. The ranking for these areas were:

Total land requirements: FWD/WWTP < MSW/I < MSW/L < MSW/C < FWD/OSS

Total system energy requirements: FWD/WWTP < MSW/L < MSW/C < MSW/I < FWD/OSS
Total system materials: MSW/C < MSW/I < FWD/WWTP < MSW/L < FWD/OSS
Total flows to environment: MSW/C < MSW/L < MSW/L < FWD/WWTP < FWD/OSS
Total system costs: MSW/L < MSW/C < FWD/WWTP < MSW/I < FWD/OSS

Overall, of the five systems, the FWD/WWTP has the lowest municipality cost (system cost due to disposer cost, which is paid by the homeowner); least air emissions; converts food waste to a recycled resource; is the most convenient system of food waste disposal; is the most likely system for organic source separation; and overall is the most environmentally friendly and sustainable option.

[Diggelman, 1998]

5.4 "Assessment of Food Disposal Options in Multi-Unit Dwellings in Sydney"

CRC for Waste Management and Pollution Control Limited

The Life Cycle Analysis (LCA) is a comparison of four systems for the processing of 182 wet kg of food waste. The four systems are:

1. Food Waste Disposer/Wastewater Treatment Plant (FWD/WWTP)
2. Home Composting (HC)
3. Co-Disposal or Municipal Solid Waste Collection/Landfill (MSW/L)
4. Centralized Composting or Municipal Solid Waste Collection/Compost (MSW/C)

The research was undertaken as five separate but interlinked studies examining technical and operational, environmental, economic, social acceptance, and microbial risk impacts. [Note: The beneficial use of by-products (i.e., compost and biosolids) was not part of the study. Also, the amount of recovered energy is uncertain should biogas be used for energy recovery. Electricity generation from biogas can lead to high environmental improvements for the FWD/WWTP and Co-Disposal (MSW/Landfill) systems. However, little biogas was being recovered at the WWTP (Bondi STP) that was used in the study. In addition, the Bondi STP is a "high rate primary" plant, thus, nutrients (nitrogen and phosphorus) are released to treated effluent, which caused a poor eutrophication rating for the FWD/WWTP system.]

Environmental Impacts: Home Composting has the smallest environmental impacts. The FWD/WWTP system ranked second in terms of energy consumption, global warming potential, and eutrophication potential; but fourth in terms of human, aquatic and terrestrial toxicity potential. Co-Disposal ranked second highest in toxicity potential and eutrophication potential; ranked slightly behind FWD/WWTP for energy consumption and acidification; and had the lowest ranking for global warming potential. Centralized Composting has a relatively poor environmental performance due to its energy intense collection activities, ranking fourth for energy and acidification; and third in the remaining categories.

System Cost Comparison: Home Composting is the least expensive option for multi-unit residents; then Centralized Composting; Co-disposal; and FWD/WWTP is the most expensive (due to a high initial unit and installation cost paid by homeowner). From a system point-of-view, the FWD/WWTP system was again the most expensive; Co-Disposal (the current system utilized by Sydney) has landfill space and does not require capital investment; Centralized Composting would necessitate capital investment. The FWD/WWTP system may require capital investment beyond a 25% market share.

Health Risk Comparison: The FWD/WWTP system may only marginally increase the rate of sanitary sewer overflows during periods when the sewer is flowing at 100%. Home Composting without pet fecal waste or meat products addition should result in acceptably low infection rates for all pathogen groups. Centralized Composting (including human fecal waste) should be satisfactory from the point of no significant pathogen risks. Overall vector-based diseases were not considered significantly different due to the operation of food waste disposal units and domestic composting containers.

Social Impact Comparison: Disposal of food with municipal waste was judged as the least satisfactory option (current Sydney system). Home Composting was judged impractical for multi-unit dwellings. FWD/WWTP and food waste collection with Centralized Composting were much more appropriate, provisional on the level of treatment that would enable reuse of the waste residuals (which was not studied).

[CRC for Waste Management and Pollution Control Limited, 2000]

6.0 Research References

ASSE International. June, 2006. “Performance Requirements for Plumbing Aspects of Residential Food Waste Disposer Units – ASSE Standard #1008.”

Association of Home Appliance Manufacturers (AHAM). 2009. “Food Waste Disposers.”

Battistoni, Paolo, Francesco Fatone, Daniele Passacantando, and David Bolzonella. Water Research, 2007. “Application of Food Waste Disposers and Alternate Cycles Process in Small-Decentralized Towns: A Case Study.”

Bolzonella, David, Paolo Pavan, Paolo Battistoni, and Franco Cecchi. Department of Science and Technology. University of Verona. 2003. “The Under Sink Garbage Grinder: A Friendly Technology for the Environment.”

CECED – European Committee of Manufacturers of Domestic Appliances. Spring 2003. “Food Waste Disposers – An Integral Part of the EU’s Future Waste Management Strategy.”

Chowdhury, M.M.I., M. Kim, B.M. Haroun, G. Nakhla, and M. Keleman. 2016. Water Environment Research. “Flocculent Settling of Food Waste.”

Clauson-Kaas, Jes and Janus Kirkeby. DANVA, August, 2011. “Food Waste Disposers: Energy, Environmental and Operational Consequences of Household Residential use.”

CRC for Waste Management and Pollution Control Limited. December, 2000. “Assessment of Food Disposal Options in Multi-Unit Dwellings in Sydney.”

de Koning, Dr.ir. J. Delft University of Technology. July 2004. “Environmental Aspects of Food Waste Disposers.”

de Koning, Dr.ir. J. and Prof.ir. J.H.J.M. van der Graaf. Delft University of Technology. April 1996. “Kitchen Waste Disposer Effects on Sewer System and Wastewater Treatment.”

Diggelman, Dr. Carol and Dr. Robert K. Ham. Department of Civil and Environmental Engineering – University of Wisconsin. January 1998. “Life-Cycle Comparison of Five Engineered Systems for Managing Food Waste.”

DeOreo, William. Aquacraft, Inc. Water Engineering and Management, July 2011. “California Single-Family Water Use Efficiency Study.”

EPA – US Environmental Protection Agency. June, 2013. “Advancing Sustainable Materials Management: 2013 Fact Sheet.

Evans, Tim. June, 2007. “Environmental Impact Study of Food Waste Disposers."

Evans, Tim, Per Andersson, Asa Wievegg, and Inge Carlsson. Water and Environment Journal, 2010. “Surahammar: A Case Study of the Impacts of Installing Food Waste Disposers in 50% of Households.”

He, Xia, et al. Environmental Science and Technology. April, 2011. “Evidence for Fat, Oil, and Grease (FOG) Deposit Formation Mechanisms in Sewer Lines.”

He, Xia, L. Reyes III, J. Ducoste. 2017. Critical Reviews in Environmental Science and Technology. “A critical review of fat, oil, and grease (FOG) in sewer collection systems: Challenges and control.”

Hernandez, Gerald L., Kenneth R. Redd, Wendy A. Wert, An Min Liu, and Tim Haug. BioCycle Magazine. January 2002. “Los Angeles Digesters Produce Energy From Airport Food Residuals.”

Hernanadez, Gerald L., Kenneth R. Redd, Wendy A. Wert, An Min Liu, and Tim Haug. 2002. “Hyperion Advanced Digestion Pilot Program.”

Imanishi, Akio. National Institute for Land and Infrastructure Management, March 2005. “Report on Social Experiment of Garbage Grinder Introduction.”

InSinkErator. July 2011. “*Executive Summary* – Systems for the Management and Disposal of Food Waste.”

InSinkErator. 2016. “The Food Waste Disposer as a Municipal Tool for Waste Diversion.”

Jones, P.H. 1990. “Kitchen Garbage Grinders - The Effect on Sewerage Systems and Refuse Handling.”

Karlberg, Tina and Erik Norin. VA-FORSK REPORT, 1999-9. “Food Waste Disposers – Effects on Wastewater Treatment Plants. A Study from the Town of Surahammar.”

Karrman, Erik, Mattias Olofsson, Bernt Persson, Agneta Sander, and Helena Aberg. Recycling Board of Goteborg, Sweden. 2001. “Food Waste Disposers – A Solution for Sustainable Resource Management? A Pre-Study in Goteborg, Sweden.”

Keener, Kevin M, Joel Ducoste, and Leon M. Holt. December, 2007. “Properties Influencing Fat, Oil, and Grease Deposit Formation.”

Kegebein, Jorg, Erhard Hoffmann, and Prof. Hermann H. Hahn. Institute for Municipal Water Treatment, University of Karlsruhe, 2001. “Co-Transport and Co-Reuse – An Alternative to Separate Bio-Waste Collection?”

Keleman, Michael. 2013. “Let’s Clear the FOG.”

Kim M. G. Nakhla, M. Keleman. Journal of Environmental Management. February 2019. “Modeling the Impacts of Food Wastes on Wastewater Treatment Plants."

Kim, M, M.M.I. Chowdhury, G. Nakhla, and M. Keleman. Bioresource Technology, February, 2015. “Characterization of Typical Household Food Wastes from Disposers: Fractionation of Constituents and Implications for Resource Recovery at Wastewater Treatment.”

Leverenz, Harold and George Tchobanoglous. May 2013. “Energy Balance and Nutrient Removal Impacts of Food Waste Disposers on Wastewater Treatment.”

LGA (Local Government association). October 2012. “The Potential of Food Waste Disposal Units to Reduce Costs.”

Lin, Hongjian, Y. Wang, L. van Lierop, C. Zamalloa, C. Furlong, M. Keleman, and B. Hu. 2019. Waste Management & Research. “Study of Food Waste Degradation in a Simulated Septic Tank.”

Maalouf, Armani, M. El-Fadel. 2017. Waste Management. “Effect of a Food Waste Disposer Policy on Solid Waste and Wastewater Management with Economic Implications of Environmental Externalities.”

Marashlian, Natasha and Mutasem El-Fadel. American University of Beirut, Lebanon. October 2004. “The Effect of Food Waste Disposers on Municipal Waste and Wastewater Management.”

Mattsson, Jonathan, Annelie Hedstrom, and Maria Viklander. May 2014. Environmental Technology. “Long-Term Impacts on Sewers Following Food Waste Disposer Installation in Housing Areas.”

Metcalf and Eddy. 2014. “Wastewater Engineering. Treatment and Resource Recovery.”

Morris, Jeffrey, Scott Matthews, and Clarissa Morawski. Waste Management, 2012. “Review and Meta-Analysis of 82 Studies on End-of-Life Management Methods for Source Separated Organics.”

Nakhla, Dr. George. Department of Chemical and Biochemical Engineering, Western University, London, Ontario, Canada, September 2014. “Settleability and Detailed Organics Characterization of Food Wastes.”

New York City Department of Environmental Protection. 1997. “*Executive Summary* - The Impact of Food Waste Disposers in Combined Sewer Areas of New York City.”

New York City Department of Environmental Protection. June 1997. “The Impact of Food Waste Disposers in Combined Sewer Areas of New York City.”

Nichols, Andy, Simon Tait, and Abbi Legge. December 2019. University of Sheffield. “Food Waste Disposer Particle Characterisation.”

PE Americas. February 2011. “Life Cycle Assessment of Systems for the Management and Disposal of Food Waste.”

Rosenwinkel, K.H. and D. Wendler. Institute for Water Quality and Waste Management, University of Hanover (ISAH), 2001. "Influences of Food Waste Disposers on Sewerage System, Wastewater Treatment and Sludge Digestion."

Seabloom, Robert W., T. Bounds, T. Loudon. 2004. "University Curriculum Development for Decentralized Wastewater Management."

Shpiner, Ram. Submitted to the Senate of the Technion – Israel Institute of Technology. January 1997. "The Effect of Domestic Garbage Grinding on Sewage Systems and Wastewater Treatment Plants."

Strutz, Bill. Internal Information, October, 2005. "Electricity Use of Food Waste Disposers."

Tendaj, M, et al. 2008. "Kitchen Disposal Units (KDU) in Stockholm, Stockholm Water's Pre-Study on the Preconditions, Options, and Consequences of Introducing KDUs in households in Stockholm."

Terpstra, Prof. drs. P.M.J. Agricultural University Wageningen. April 1995. "Kitchen Waste Disposal Treatment: An Evaluation."

Thomas, Philip. Water and Environment Journal, 2010. "The Effects of Food Waste Disposers on the Wastewater System: A Practical Study."

Tongji University. 2010. "Environmental and Economic Cost Benefit Analysis of Food Waste Disposers."

Tongji University. 2013. "Report for Food Waste Disposer Pilot Program in PuDong and Environmental Impact Assessment."

Waste Management Research Unit – Griffith University. August 1994. *Executive Summary*. "Economic and Environmental Impacts of Disposal of Kitchen Organic Wastes Using Traditional Landfill – Food Waste Disposer – Home Composting."

WERF – Water Environment Research Foundation. *Executive Summary* – Cost Affective, Sustainable Alternatives to Landfills for Managing Food Waste.

WERF – Water Environment Research Foundation. 2012. Sustainable Food Waste Evaluation – Final Report.

WERF – Water Environment Research Foundation. 2015. "A Guide to Net-Zero Energy Solutions for Water Resource Recovery Facilities."

Williams, J.B, C. Clarkson, C. Mant, A. Drinkwater, and E. May. Water Research, 2012. "Fat, Oil, and Grease Deposits in Sewers: Characterization of Deposits and Formation Mechanisms."

Yang, Xinmi, Takao Okashiro, Katsuhiko Kuniyasu. Japan Education Center of Environmental Sanitation, August 2009. “Impact of Food Waste Disposers on the Generation Rate and Characteristics of Municipal Solid Waste.”

Yoshida, Ayako, et al. National Institute for Land and Infrastructure Management, Ministry of Land, Infrastructure and Transport, Japan. “Impacts of Food Waste Disposers on Sewage Systems.”

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Study of food waste degradation in a simulated septic tank

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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

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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

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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°C and a hydraulic retention time (HRT) of 6.86 days. This was

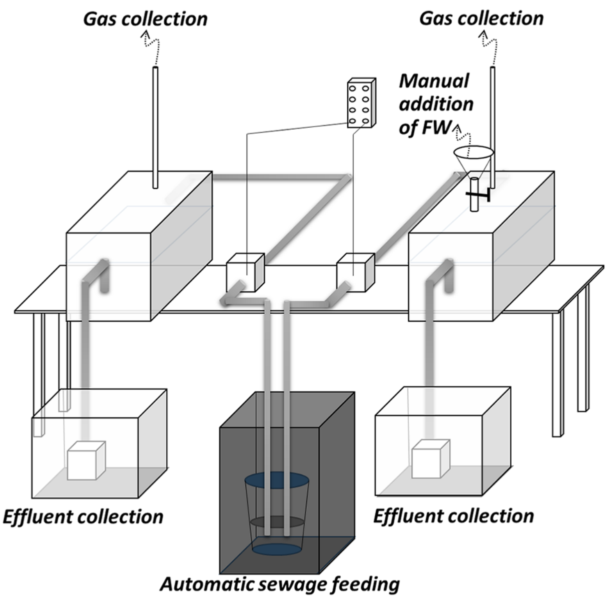

Figure 1. Setup of simulated septic tanks. The control tank was fed with sewage only, whereas the experiment tank was fed with sewage and food waste.
FW: food waste.

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

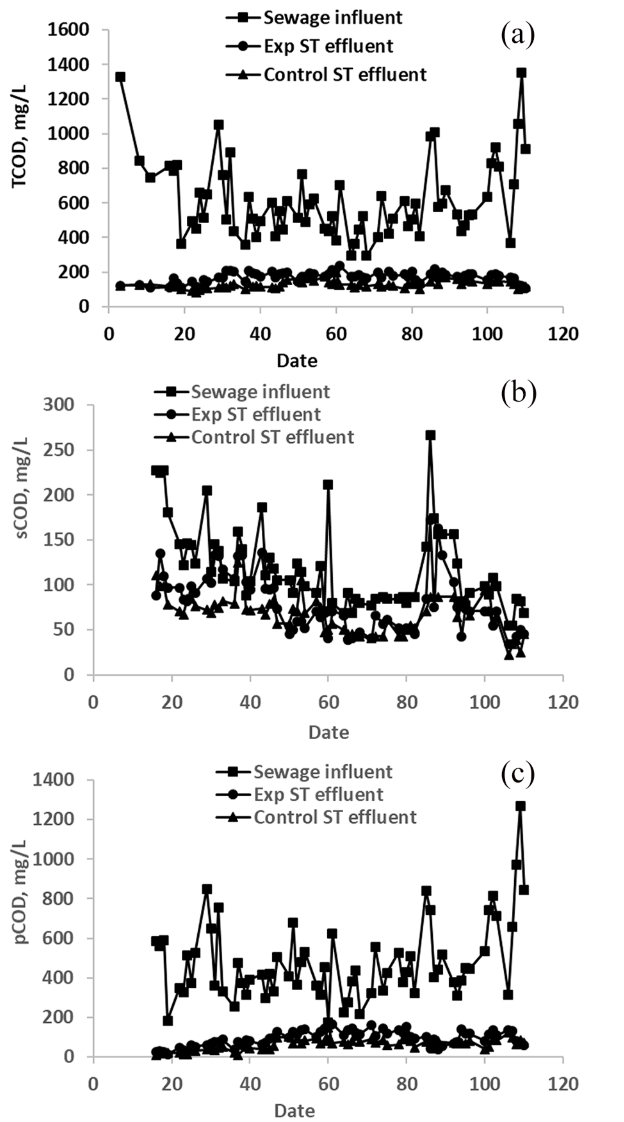

Figure 2. Time course profile of total COD (TCOD) (a), soluble COD (sCOD) (b), and particulate COD (pCOD) (c). The control tank was fed with sewage only, whereas the experiment tank was fed with sewage dosed with food waste. Operated at 15°C with a hydraulic retention time of 6.86 days.

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.

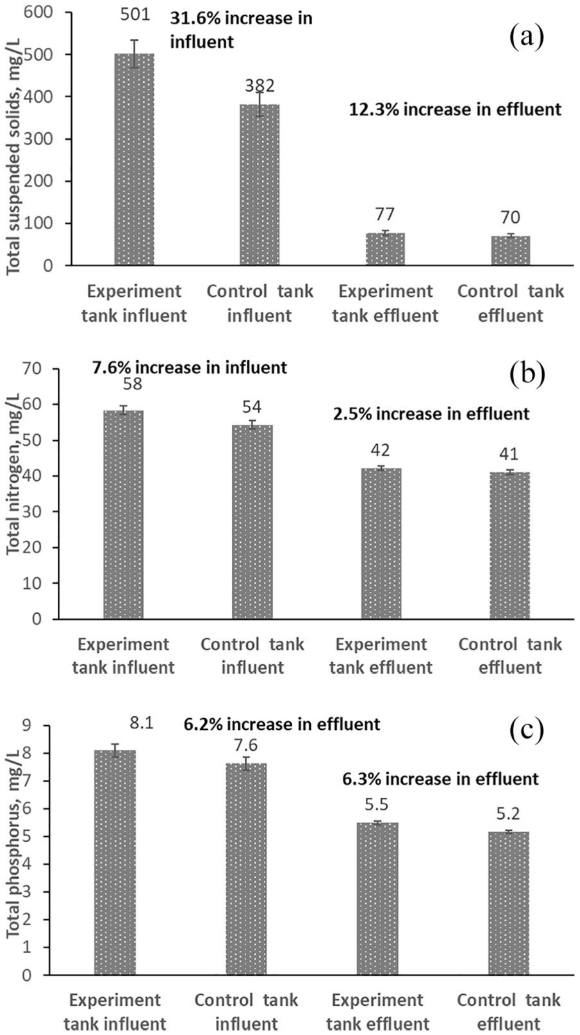

Figure 3. Average concentrations of total suspended solids (a), total nitrogen (b), and total phosphorus (c). The control tank was fed with sewage only, whereas the experiment tank was fed with sewage dosed with food waste. Operated at 15°C with a hydraulic retention time of 6.86 days.

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

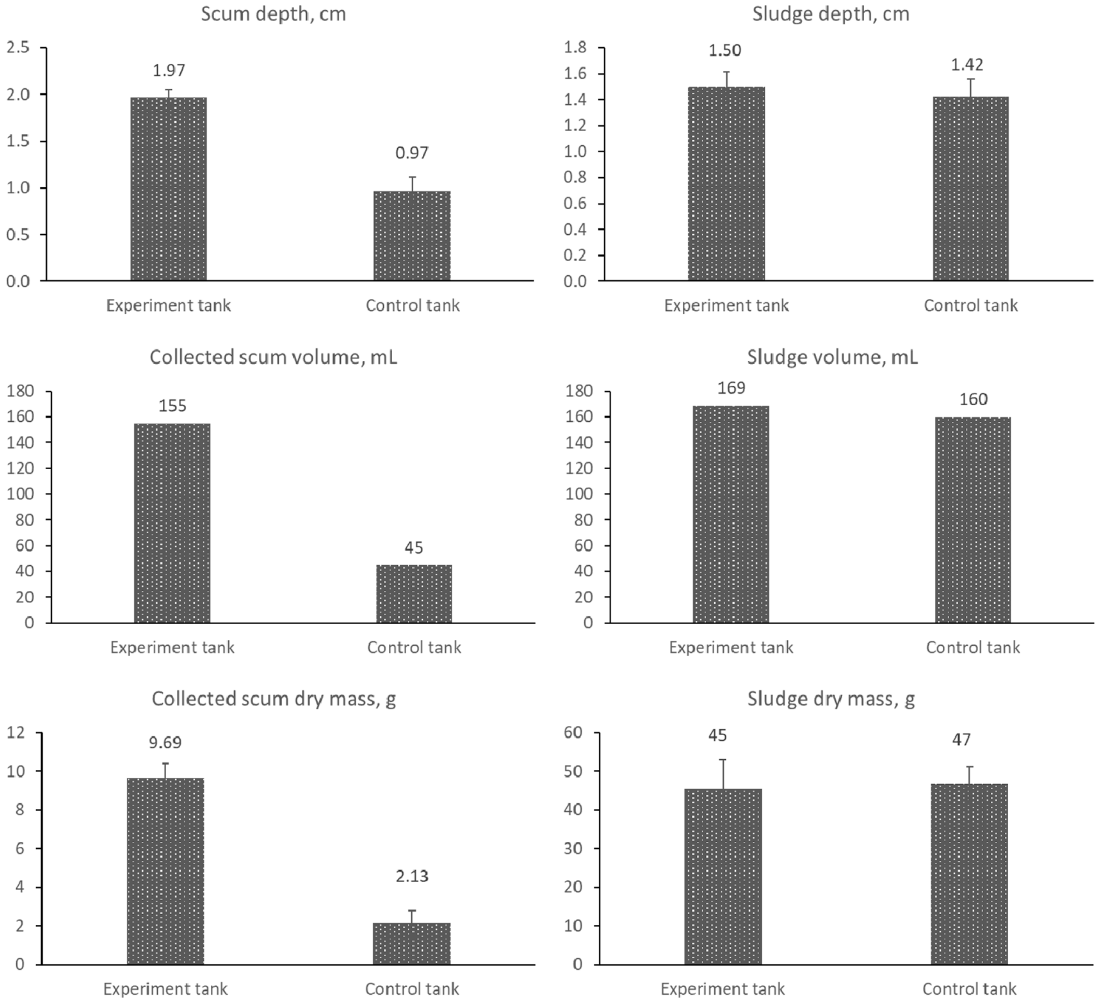

Figure 4. Sludge and scum formation in the experiment and control tanks. The control tank was fed with sewage only, whereas the experiment tank was fed with sewage dosed with food waste. Operated at 15°C with a hydraulic retention time of 6.86 days.

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

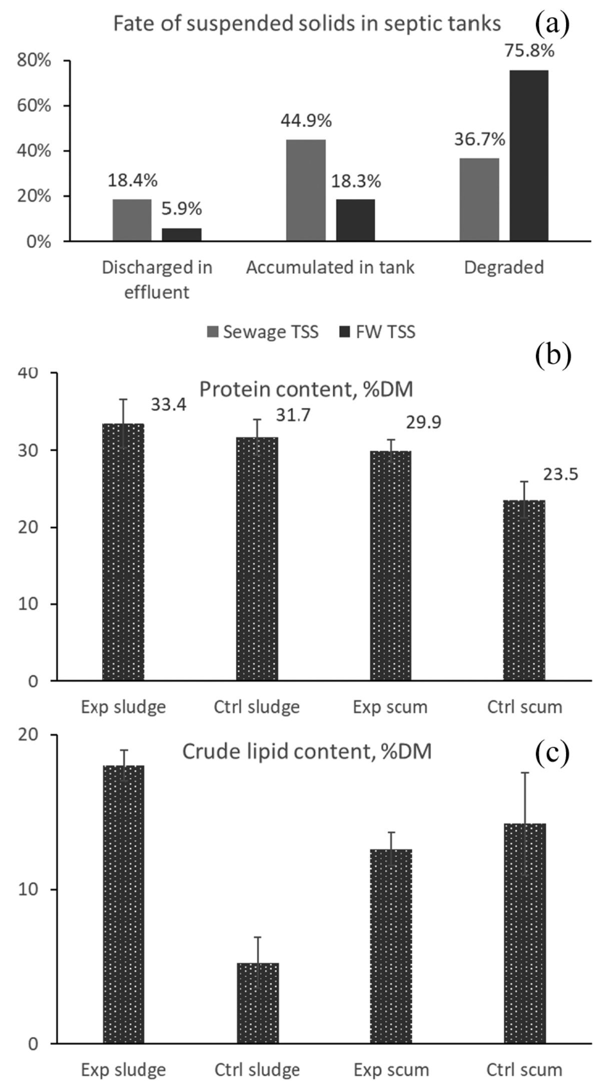

Figure 5. Composition of suspended solids (a), amount of protein (b), and crude lipid (c) in sludge and scum The control tank was fed with sewage only, whereas the experiment tank was fed with sewage dosed with food waste. Operated at 15°C with a hydraulic retention time of 6.86 days.
Exp.: Experiment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

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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.

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Research article

Modeling the impact of food wastes on wastewater treatment plants

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ABSTRACT

Food waste (FW) enriched with readily biodegradable organics can enhance biological nutrient removal (BNR), and biogas production. This study conducted extensive wastewater treatment simulations using BioWin software to predict the impact of food waste on nutrient removal, biogas generation, and energy balance. A total of 114 scenarios were tested to simulate different treatment technologies i.e. conventional activated sludge, Modified Ludzack-Ettinger (MLE), anaerobic-anoxic-aerobic (A2O), Bardenpho, and 2nd generation BNR technologies. The simulations also included sidestream treatment for nitrogen removal, as well as mainstream partial nitrification based on BNR. The results showed that FW addition enhanced nitrogen removal and decreased effluent nitrogen for BNR processes by 3.6–7.9 mg/L for MLE, 0.6–1.3 mg/L for A2O, and 1–2.3 mg/L for Bardenpho. In addition, FW addition decreased net operational cost by 26%–63% for BNR processes operating at mainstream conventional dissolved oxygen (DO) of 2 mg/L, 24%–78% for partial nitrification system, 29%–54% for sidestream, and 23%–76% for sidestream with mainstream partial nitrification process. The total net energy benefit considering both the net change in aeration energy and methane energy for a typical 37,854 m^3/d or 10 MGD plant increased with FW addition by 3300–7900 kWh/d with a variation between BNR types, due to a substantial increase in methane production. Carbon diversion scenarios showed that the higher primary treatment efficiencies decreased the net operational cost and increased net energy gain.

1. Introduction

Food wastes (FW) diversion from landfills to wastewater treatment plants (WWTPs) is a promising strategy to resolve the shorter life-span of landfill sites and mitigate the environmental impacts of landfills as well as utilize the energy of organic-rich food wastes (Iacovidou et al., 2012). According to the food waste hierarchy promoted by the European Union, landfilling is the last option for FW management after the prevention of waste generation and resource & energy recovery, with FW co-digestion being considered one of the best options for recovery of renewable energy (Bolzonella et al., 2019). Previous studies reported the potential use of food waste in wastewater treatment such as source of carbon for nutrient removal and energy resource through biogas production from co-digestion (Kim et al., 2017; Tang et al., 2017; Zheng et al., 2018). Moreover, biofuel generated from FW co-digestion can also be used in the automotive sector (Bolzonella et al., 2019).

In order to understand the fate of FW in WWTP, several studies were conducted on FW characteristics, FW settleability, biological treatment, and anaerobic biodegradability. The previous studies showed that FW characteristics can enhance WWTP performance. According to a study by Kim et al. (2015) who characterized 50 ground food waste samples, the average relative mass ratio of COD:BOD_5:TSS:TN:TP:dry food was 1.21:0.58:0.36:0.025:0.013:1 i.e. the COD:BOD_5:TSS:TN:TP mass ratios for food waste are 93:44.6: 27.7:1.9:1, and the particulate fractions of COD, BOD_5, and N were 58%, 67%, and 74% with SCOD/SN of 63 and PCOD/PN of 42. The 63 was much greater than the optimal COD/N ratio of > 9 for nitrogen removal and COD/P ratio of 26–43 for P removal. As the particulate fraction of FW would go to digestion, the PCOD/PN of 42 or C/N of 15 using carbon to COD ratio of 0.35 is also close to the optimal C/N ratio of 15–30 for nutrient balance requirements for microbes and digestion performance for anaerobic digestion (Zhang et al., 2014). Methane yield was also higher for FW compared with municipal wastewater treatment biosolids. For instance, the aforementioned study by Dai et al. (2013) presented that a mesophilic anaerobic digester fed with 100% FW (SRT 30 days) produced biogas of 0.47 LCH_4/gVS added much higher than a reactor treating 100% municipal sludge (0.24 LCH_4/gVS added).

The previous studies also explored FW settleability to assess the fate of FW in primary clarification (Chowdhury et al., 2016; Thomas, 2011). Thomas (2011) reported that FW TSS were removed at 77% efficiency

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Acronyms

A2O	anaerobic-anoxic-aerobic
BNR	biological nutrient removal
CANON	complete autotrophic nitrogen removal over nitrite
CAS	conventional activated sludge
DO	dissolved oxygen
EBPR	enhanced biological phosphorus removal
FW	food waste
FWD	food waste disposer
HRT	hydraulic retention time
MLE	Modified Ludzack-Ettinger
MLSS	mixed liquor suspended solids
MUCT	Modified University of Cape Town
MWW	municipal wastewater
PTE	primary treatment efficiency
rbCOD	readily biodegradable COD
SON/PON	soluble/particulate organic nitrogen
SRT	solids retention time
TBOD/SBOD/PBOD	total/soluble/particulate biochemical oxygen demand
TCOD/SCOD/PCOD	total/soluble/particulate chemical oxygen demand
TKN/STKN	total/soluble kjeldahl nitrogen
TP/SP	total/soluble phosphorus
TS/VS	total/volatile solids
TSS/VSS	total/volatile suspended solids
VFA	volatile fatty acid
VSR	volatile solids destruction

Wastewater fractionation

C_T	total COD
S_H	–rapidly hydrolysable COD
S_I	soluble inert COD
S_S	readily biodegradable COD
S_T	soluble COD
X_H	active heterotrophic biomass
X_I	particulate inert COD
X_S	slowly biodegradable COD
X_T	total particulate COD

after 2 h of settling at a mixture of FW (dry mass) to water ratio of 1:11.7 in an 8 L bucket. Chowdhury et al. (2016) who conducted a settling test using a 3.2 m column with a diluted FW slurry (a dry FW mass water ratio of 1:132) observed TSS removal of 62% after 3 h of settling, higher than municipal wastewater TSS removal of 50%. These results indicate good settleability of FW in primary clarification.

Similarly, Battistoni et al. (2007) also investigated the impact of FW on an intermittently aerated wastewater treatment plant in a town of 250 inhabitants with a market penetration of food waste disposers of 67%. Battistoni and co-authors observed influent TSS, COD, N increases with the installation of the FWDs of 30%, 44%, 19%, respectively, with the COD/N ratio increasing from 9.9 to 12 and rbCOD/COD increase from 0.2 to 0.24, resulting in 27% enhancement in nitrogen removal efficiency. A recent study by Zheng et al. (2018) who used alkaline fermentation liquid of kitchen wastewater a carbon source showed that nitrogen and phosphorus removal were 7%–8% higher than with acetate. Moreover, Tang et al. (2017) who studied the use of fermentation liquid of food waste (COD/N ratio of 92) for pilot-scale anoxic-oxic bioreactor treating low COD/N ratio (5.5) wastewater reported that nitrogen removal efficiency increased from 20% to 44%–67%.

Studies on anaerobic biodegradability of FW showed that food waste has a high potential for anaerobic degradability with a high VS/TS ratio of > 90% (Xu et al., 2018). Similarly, the food waste co-digestion study by Kim et al. (2017) who operated five semi-continuous anaerobic digesters fed with municipal sludge and food waste with different content of food waste at an SRT of 20 days reported that food waste addition increased biogas production by 18%–20% compared with municipal sludge mono-digestion. Moreover, Dai et al. (2013) reported that FW co-digesters with municipal sludges at the feed mixing ratio of 0%–100% (VS basis) and SRT of 30 days increased VS destruction efficiency from 38% to 86% and enhanced the methane yield from 0.24 to 0.62 LCH_4/gVS_{added} with an increasing content of FW. A recent study by Koch et al. (2016) who monitored a full-scale wastewater treatment plant (WWTP) digester before and after adding FW also reported that methane production enhanced from 0.31 to 0.39 L/kgVS due to the high hydrolysis rate of food wastes and enhancement of acidogenesis and methanogenesis. In addition, Nghiem et al. (2017) who analyzed data collected from FW co-digestion field survey and literature introduced successful cases of achieving energy neutrality in WWTPs through energy recovery from co-digestion although FW inert impurities, regulatory uncertainty, lack of feasible options for biogas utilization, FW collection & handling, biosolids reuse, and lack of design and operational experience are bottlenecks for implementation.

The aforementioned studies demonstrated that food waste can be used as an excellent resource for realizing energy neutrality at WWTP as well as reducing operational costs by saving external carbon addition. In light of the increasing importance of the food-energy-water nexus, understanding of impact of food waste on wastewater treatment plants liquid and solids streams is essential for optimization.

While the abovementioned studies presented the potential advantages of FW treatment in WWTP, little information is still available to understand the impact of FW on WWTP in terms of operational parameters and performance. A study by Bolzonella et al. (2003) evaluated the impact of organic fraction of municipal solid wastes (OFMSW) on different BNR processes i.e. pre-denitrification process and Phoredox using the Activated Sludge Model 2. The authors reported that municipal wastewater + OFMSW which had higher solids, COD, and N concentrations than municipal wastewater only by 280%, 250%, and 40% respectively, increased oxygen consumption by 13%–14%, waste sludge by 59%–86%, and biogas production by 35%–58% for both processes with a higher increase for Phoredox but also decreased iron dosage for chemical P removal decreased by 43% for pre-denitrification and by 25% for Phoredox, indicating the benefits of FW treatment in WWTP operations.

However, no comprehensive studies have been conducted to investigate the diverse factors including energy balance, cost analysis, nutrient removal, and carbon credit related to food waste addition. The overall impact of food waste on WWTP including liquid and solids stream remains unclear. In addition, as next generation BNR technologies have been developed such as short-cut nitrification and sidestream processes, the impact of food waste on the new technologies needs to be explored to achieve energy-neutral or energy-positive plants.

Thus, this study employed extensive modeling using BioWin software to explore food waste impact on various wastewater processes i.e. conventional activated sludge, first generation biological nutrient removal (MLE, A2O, Bardenpho), and emerging technologies (partial nitrification and sidestream processes). To the best of the authors' knowledge, this is the first study that modeled the impact of food waste on both liquid and solid streams of WWTP using emerging BNR processes including sidestream N removal technologies. The objectives of this study included: 1- a preliminary assessment of the impact of FW on effluent quality, aeration energy, biogas energy, digested biosolids, operating MLSS, and secondary clarification solids loading, 2-

quantification of energy and operational cost analysis based on food waste impact on liquid and solids streams, and 3- evaluation of the sensitivity of the FW impact to primary treatment efficiency, and correlating it to carbon diversion.

2. Materials and methods

2.1. System configuration

BioWin 5.3 (EnviroSim Associates Ltd., Hamilton, Canada) was used to simulate a total of 114 wastewater treatment scenarios. The

Table 1
Summary of simulation results for typical condition (Temp 20 °C, DO = 2 mg/L).

Parameters	Unit	CAS					MLE				
		Value			Increment (%)		Value			Increment (%)	
		FW0%	FWd50%	FW100%	FW50%	FW100%	FW0%	FW50%	FW100%	FW50%	FW100%
Effluent quality											
BOD	mg/L	3	4	4	9	18	3	3	3	7	15
Total N	mg/L	36	36	36	1	1	22	18	15	−17	−33
Amm-N	mg/L	0.5	0.4	0.4	−20	−20	0.5	0.5	0.5	0	0
NO3	mg/L	33	33	33	1	1	19	15	12	−20	−39
NO2	mg/L	0.1	0.1	0.1	0	0	0.1	0.1	0.1	0	0
NOx	mg/L	33	33	33	0	1	19	15	12	−19	−39
Aeration and chemical demands											
Air demand	m^3/min	240	266	293	11	22	215	231	248	7	15
Aeration Power	kW	242	269	296	11	22	218	234	250	7	15
Fe dosage	kg/day	702	710	721	1	3	708	716	727	1	3
Biosolids Management											
Primary sludge	kg/day	4867	4867	4867	0	0	4867	4867	4867	0	0
TWAS	kg/day	4371	4700	5040	8	15	4142	4442	4727	7	14
MLSS	mg/L	2272	2438	2609	7	15	2187	2303	2445	5	12
Digested biosolids	kg/day	5890	6224	6559	6	11	5782	6086	6391	5	11
Methane production	m^3/day	2085	2481	2867	19	38	2013	2395	2770	19	38
Glycerol dosage											
Required for full denitrification*	kg/day	NA	NA	NA	NA	NA	4110	3309	2520	NA	NA
Glycerol saved	kg/day	NA	NA	NA	NA	NA	0	800	1589	NA	NA
Operational cost											
Biosolids Disposal	$/day	588	622	655	6	11	577	608	638	5	11
Chemical Cost for dewatering	$/day	104	109	115	6	11	102	107	112	5	11
Chemical Cost for P removal	$/day	773	781	793	1	3	779	788	800	1	3
Chemical for P precipitation in AD	$/day	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Aeration blower cost	$/day	582	645	711	11	22	523	561	601	7	15
Methane energy	$/day	738	879	1016	19	38	713	848	981	19	38
Supplemental carbon to achieve same effluent TN (glycerol)	$/day	NA	NA	NA	NA	NA	0	353	701	NA	NA
Total operational cost	$/day	2046	2158	2275	5	11	1980	2065	2152	4	9
Net operational cost	$/day	1308	1279	1259	−2	−4	1267	863	470	−32	−63
Effluent quality											
BOD	mg/L	3	4	4	11	22	3	3	4	10	20
Total N	mg/L	15	14	14	−5	−8	11	10	9	−11	−21
Amm-N	mg/L	0.9	0.9	0.9	−1	−1	0.5	0.5	0.5	0	0
NO3	mg/L	11	11	10	−7	−13	8	7	6	−15	−29
NO2	mg/L	0.4	0.4	0.4	0	0	0	0	0	0	0
NOx	mg/L	12	11	10	−7	−13	8	7	6	−15	−29
Aeration and chemical demands											
Air demand	m^3/min	205	218	231	6	13	210	222	235	6	12
Aeration Power	kW	207	220	234	6	13	212	225	237	6	12
Fe dosage	kg/day	330	210	140	−36	−58	431	283	156	−34	−64
Biosolids Management											
Primary sludge	kg/day	4508	4508	4508	0	0	4220	4220	4220	0	0
TWAS	kg/day	3755	3913	4145	4	10	3733	3839	3972	3	6
MLSS	mg/L	1986	2069	2190	4	10	1968	2024	2093	3	6
Digested biosolids	kg/day	5746	6039	6343	5	10	5594	5871	6160	5	10
Methane production	m^3/day	2051	2444	2835	19	38	2027	2421	2807	19	38
Glycerol dosage											
Required for full denitrification	kg/day	2526	2354	2204	NA	NA	1816	1539	1285	NA	NA
Glycerol saved	kg/day	0	172	322	NA	NA	0	277	531	NA	NA
Operational cost											
Biosolids Disposal	$/day	574	603	633	5	10	559	586	615	5	10
Chemical Cost for dewatering	$/day	101	106	112	5	10	98	103	108	5	10
Chemical Cost for P removal	$/day	363	231	154	−36	−58	475	311	172	−34	−64
Chemical for P precipitation in AD	$/day	273	273	273	0	0	273	273	273	0	0
Aeration blower cost	$/day	498	529	561	6	13	509	539	569	6	12
Methane energy	$/day	727	866	1004	19	38	718	858	994	19	38
Supplemental carbon to achieve same effluent TN (glycerol)	$/day	0	76	142	NA	NA	0	122	234	NA	NA
Total operational cost	$/day	1808	1741	1733	−4	−4	1913	1812	1737	−5	−9
Net operational cost	$/day	1081	800	587	−26	−46	1195	832	508	−30	−57

*MLE FW50%: (15 $mgNO_3$/L × 7 + 0.1 $mgNO_2$/L × 4)/1000 g/kg × 37,854 m^3/d/1.217 mg COD/mg glycerol = 3309 kg/day.

processes explored in this study were conventional activated sludge (CAS), modified Ludzack-Ettinger (MLE), anaerobic-anoxic-aerobic (A2O), and Bardenpho with/without chemical P removal as summarized in Table A.1 with the detailed process diagrams shown in Fig. A.1. The MLE process consists of anoxic and aerobic tanks for biological nitrogen removal via nitrification aerobically and denitrification anoxically. The A2O system employs enhanced biological phosphorus removal in addition to nitrogen removal by placing anaerobic tanks prior to anoxic tanks for facilitating the activity of phosphate accumulating organisms. A five stage Bardenpho process has a similar configuration to A2O system with respect to the first three reactors with additional anoxic and aerobic tanks in order to enhance nitrogen removal. All hydraulic retention time and recycle ratios are based on the influent wastewater flow of 37,854 m^3/d (10 MGD). As can be seen for EBPR processes like A2O and Bardenpho, fermentation for rbCOD production and chemical P removal of the dewatering centrate were implemented. Selected scenarios involving completely autotrophic nitrogen removal over nitrite (CANON) for sidestream N were evaluated.

In order to account for the high anaerobic biodegradability of FW, FW segregation implementing dedicated primary clarification and anaerobic digestion for FW primary sludges was incorporated. TSS and VSS removal efficiencies for the FW in primary clarification were set at 65% (Chowdhury et al., 2016). The volatile solids removal efficiency for FW digestion was set to 80%. Three cases were considered: MWW alone, 50% penetration of FWD, and 100% penetration of FWD. The per capita contribution of FWD are 30 gTS/day (dry basis) (Leverenz and Tchobanoglous, 2013). The per capita wastewater flow was 280 L/day. FW flowrates were 189 m^3/d (0.05 MGD) for 50% penetration of FWD and 378.5 m^3/d (0.1 MGD) for 100% penetration of FWD.

All modeling was conducted at 20 °C. Chemical P removal was achieved using ferric chloride dosed before secondary clarification to achieve an effluent total P of 0.5 mg/L. Mesophilic anaerobic digester SRTs for the four processes were adjusted to 14–30 days in order to meet 55% volatile solids reduction (VSR). Partial nitrification was achieved through setting DO level in the aerobic tank at 0.5 mg/L.

2.2. Influent and FW characteristics

Influent municipal wastewater and food waste characteristics are determined from the literature.

FW characteristics were estimated based on seven previous studies as presented in Table A.2 (Chowdhury et al., 2016; Kim et al., 2015, 2017; Leverenz and Tchobanoglous, 2013; Orhon and Çokgör, 1997; Thomas, 2011; Yazdanpanah et al., 2018). FW COD was characterized as 43% soluble COD and 57% particulate COD. Readily biodegradable COD and slowly biodegradable COD was 41% and 51% of TCOD. Similarly, inert COD was 8% of TCOD. The volatile fraction of FW solids was 88%. TCOD/TKN ad TCOD/TP ratios are 52 and 270, respectively. VFA content was 2.4% of TCOD. The composition of segregated FW was also estimated for simulation based on 50% and 100% penetration of FWD (Tables A.3a and A.3b).

FW COD of 12,700 mg/L was estimated based on a total water usage of 75 gal/capita/day, water use for food waste disposers (approx. 1% of total water use), and typical COD loading from FWDs (35g/capita/d) (Leverenz and Tchobanoglous, 2013). Other parameters were estimated based on the different factors summarized in Table A.2. rbCOD, unbiodegradable SCOD, unbiodegradable PCOD of FW TCOD were estimated as 40%, 2.4%, and 6.3%, respectively. Similarly, municipal wastewater characteristics were adopted from the previous report (Leverenz and Tchobanoglous, 2013), showing 438 mgTCOD/L, 173 mg SCOD/L, 43 mgTN/L, 32 $mgNH_4$/L, and 8.3 mgTP/L with rbCOD fraction of 0.136 (Tables A.3a and A.3b).

2.3. Unit costs and cost analysis

Cost and energy analysis were conducted using the different factors summarized in Table A.4. Different unit costs were adopted from the literature and BioWin values. Oxygen transfer efficiency of ~12% was used.

Cost analysis was conducted based on total cost and net cost. Total cost includes biosolids disposal cost, the chemical cost for dewatering, the chemical cost for P removal, the chemical cost for P precipitation in anaerobic digesters (BNR processes only), and aeration cost. Net cost was estimated by deducting methane energy from total cost for CAS and by subtracting methane energy and glycerol cost from the total cost for MLE, A2O, and Bardenpho. Particularly, glycerol cost was based on the quantity of additional glycerol COD needed to achieve the same effluent TN for the plants without FW as the plants with FW. The required glycerol (kg/day) was estimated based on COD requirement for denitrification using a factor of 7 mgCOD/mgN for nitrate and 4 mgCOD/mgN for nitrite (Tchobanoglous et al., 2003) i.e. ($NO_3 \times 7$ mgCOD/$mgNO_3$ + $NO_2 \times 4$ mgCOD/$mgNO_2$)/1000 g/kg × 37,854 m^3/d/1.217 mg COD/mg glycerol.

3. Results and discussion

3.1. Simulation at typical operating conditions (DO of 2 mg/L)

Simulation results for CAS, MLE, A2O and Bardenpho at a DO of 2 mg/L with chemical P removal are summarized in Table 1. It is interesting to note that for all four processes, the major contributors to overall operational costs are biosolids disposal, iron for P removal, and aeration cost. For the CAS, the three aforementioned costs accounted for 18%, 49%, and 30%, respectively. Effluent BOD and nitrogen concentrations for CAS with/without FW were 3–4 mgBOD/L and 36 mgTN/L. Compared with lack of FW, aeration demand for FW addition scenario increased by 11%–22% while iron dosage was close. Methane production and digested solids production also increased by 19%–38% and 6%–11%, respectively. Total operational cost increased by 5%–11% while net operational cost slightly decreased with FW addition by 2%–4%.

The MLE process achieved effluent BOD concentrations of 3 mg/L with and without FW, while effluent TN concentrations were 15–22 mg/L without FW, with a decrease of 17%–33% with FW addition. FW addition increased air demand by 7%–15% and also showed little change for iron dosage. Methane production and biosolids production also increased respectively by 19%–38% and 5%–11% with FW addition. The carbon credit from FW contribution to denitrification was 800–1600 kg glycerol/day. Operational cost analysis showed that total operational costs slightly increased by 4%–9% with FW addition while net operational cost considering energy production via methane, and cost saving of external carbon decreased by 32%–63%.

A2O simulation showed that effluent BOD and TN levels were 3–4 mg/L and 14–15 mg/L for the scenarios with and without FW addition, indicating a very close effluent quality. With addition of FW, air demand increased by less than 10%, and iron dosage decreased significantly by 44%–62%. Methane production and biosolids production were 19%–38% and 5%–10% higher with FW than without respectively. FW carbon credit for denitrification was 170–320 kg/d as glycerol. Total operational cost was similar within 4% for the cases with and without FW addition while net operational cost substantially decreased by 26%–46%, indicating that FW addition contributed a significant operational cost reduction.

Bardenpho scenarios estimated that effluent BOD and TN levels were 3–4 mg/L and 9–11 mg/L, respectively. NOx level decreased from 8 to 6 mg/L with FW addition. The segregated FW addition increased air demand by 6%–12% and decreased iron dosage by 34%–64%, relative to no FW. Methane and biosolids generation increased by 19%–38% and 5%–10%, respectively. Carbon credit for FW contribution to denitrification was 280–530 kg/day as glycerol. FW addition decreased total operational cost by 5%–9% and net operational cost by 30%–57%, indicating a significant operational cost saving through FW addition.

It is interesting to note that the overall operating costs of A2O and Bardenpho processes were ~$1900/day, as compared with ~ $2300 for the CAS. The major factor contributing to the reduction of overall operating costs for BNR was the cost of iron required for P removal, which is high due to the high raw wastewater P concentration of 8.3 mg/L.

Overall, the segregated FW impact on WWTP with mainstream DO of 2 mg/L showed that net operational cost savings were pronounced for BNR processes ranging from 26% to 63% with the higher value for FWD 100% penetration rate.

3.2. Partial nitrification (mainstream DO of 0.5 mg/L)

A summary of simulation results for partial nitrification conditions is presented in Table 2. CAS simulation showed similar effluent concentrations of BOD and TN for the scenarios with/without FW. With FW, air demand increased by 11–22% and iron dosage was similar. FW

Table 2
Summary of simulation results for partial nitrification (Temp 20 °C, DO 0.5 mg/L).

Parameters	Unit	CAS Value			Increment (%)		MLE Value			Increment (%)	
		FW0%	FW50%	FW100%	FW50%	FW100%	FW0%	FW50%	FW100%	FW50%	FW100%
Effluent quality											
BOD	mg/L	3	4	4	8	17	3	3	3	7	14
Total N	mg/L	33	33	32	−1	−1	19	15	11	−21	−41
Amm-N	mg/L	0.7	0.6	0.6	−17	−17	0.7	0.7	0.7	1	0
NO3	mg/L	29	29	29	0	−1	16	12	8	−26	−51
NO2	mg/L	0.4	0.3	0.3	−33	−33	0.5	0.5	0.5	2	0
NOx	mg/L	30	29	29	−1	−2	16	12	8	−25	−50
Aeration and chemical demands											
Air demand	m^3/min	180	200	220	11	22	156	167	179	7	15
Aeration Power	kW	182	202	222	11	22	157	169	181	7	15
Fe dosage	kg/day	704	710	718	1	2	708	716	727	1	3
Biosolids Management											
Primary sludge	kg/day	4867	4867	4867	0	0	4867	4867	4867	0	0
TWAS	kg/day	4392	4732	5077	8	16	4149	4447	4729	7	14
MLSS	mg/L	2283	2455	2628	8	15	2190	2305	2446	5	12
Digested biosolids	kg/day	5904	6236	6560	6	11	5782	6087	6393	5	11
Methane production	m^3/day	2088	2484	2880	19	38	2017	2397	2770	19	37
Glycerol dosage											
Required for full denitrification	kg/day	NA	NA	NA	NA	NA	3443	2569	1716	NA	NA
Glycerol saved	kg/day	NA	NA	NA	NA	NA	0	874	1727	NA	NA
Operational cost											
Biosolids Disposal	$/day	590	623	655	6	11	577	608	639	5	11
Chemical Cost for dewatering	$/day	104	110	115	6	11	102	107	112	5	11
Chemical Cost for P removal	$/day	775	781	790	1	2	779	788	800	1	3
Chemical for P precipitation in AD	$/day	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Aeration blower cost	$/day	437	485	533	11	22	377	405	434	7	15
Methane energy	$/day	740	880	1020	19	38	714	849	981	19	37
Supplemental carbon to achieve same effluent TN (glycerol)	$/day	NA	NA	NA	NA	NA	0	385	761	NA	NA
Total operational cost	$/day	1906	1998	2093	5	10	1835	1909	1985	4	8
Net operational cost	$/day	1166	1119	1073	−4	−8	1121	674	243	−40	−78
Effluent quality											
BOD	mg/L	4	4	4	10	20	4	4	4	8	14
Total N	mg/L	13	13	12	−5	−8	8	7	7	−13	−23
Amm-N	mg/L	1.9	1.9	1.9	−2	−2	1.2	1.2	1.1	−2	−3
NO3	mg/L	0.1	0.1	0.1	20	20	0.0	0.0	0.0	0	0
NO2	mg/L	9	8	8	−8	−14	5	4	3	−23	−41
NOx	mg/L	9	8	8	−8	−14	5	4	3	−22	−40
Aeration and chemical demands											
Air demand	m^3/min	129	139	150	8	16	130	140	150	7	15
Aeration Power	kW	130	141	151	8	16	131	141	151	7	15
Fe dosage	kg/day	142	92	70	−35	−51	227	113	45	−50	−80
Biosolids Management											
Primary sludge	kg/day	4508	4508	4508	0	0	4220	4220	4220	0	0
TWAS	kg/day	3514	3782	4092	8	16	3471	3622	3845	4	11
MLSS	mg/L	1865	2004	2164	7	16	1838	1916	2032	4	11
Digested biosolids	kg/day	5724	6072	6382	6	11	5561	5847	6140	5	10
Methane production	m^3/day	2063	2455	2845	19	38	2041	2428	2816	19	38
Glycerol dosage											
Required for full denitrification	kg/day	1141	1053	980	NA	NA	654	508	391	NA	NA
Glycerol saved	kg/day	0	88	161	NA	NA	0	146	263	NA	NA
Operational cost											
Biosolids Disposal	$/day	572	606	637	6	11	555	584	613	5	10
Chemical Cost for dewatering	$/day	101	107	112	6	11	97	102	107	5	10
Chemical Cost for P removal	$/day	156	101	77	−35	−51	250	125	50	−50	−80
Chemical for P precipitation in AD	$/day	273	273	273	0	0	273	273	273	0	0
Aeration blower cost	$/day	312	338	363	8	16	315	339	363	7	15
Methane energy	$/day	731	870	1008	19	38	723	860	997	19	38
Supplemental carbon to achieve same effluent TN (glycerol)	$/day	0	39	71	NA	NA	0	64	116	NA	NA
Total operational cost	$/day	1413	1424	1462	1	3	1490	1422	1406	−5	−6
Net operational cost	$/day	682	516	384	−24	−44	767	498	293	−35	−62

addition enhanced methane production by 19%–38%, but increased disposed dry solids by 6%–11%. With FW addition, total operational cost increased by 5%–10%, and net operational cost decreased by 4%–8%.

For the MLE process, effluent TN decreased from 19 to 11 mg/L with FW addition. Air demand increased by 7%–15% with no significant change in iron dosage. Methane production and biosolids generation increased by 19%–37% and 5%–11%, respectively. FW addition increased total operational cost by 4%–8% and decreased net operational cost by 40%–78%.

Partial nitrification for A2O process resulted in effluent BOD and TN concentrations of 4 and 12–13 mg/L, respectively. Compared with MWW alone scenarios, FW addition cases showed higher air demand by 8%–16% and lower iron dosage by 35%–51%. Methane production and biosolids generation increased by 19%–38% and 6%–11%, respectively. The carbon credit for denitrification was 88–160 kg/day as glycerol. Operational cost analysis showed that total operational cost slightly increased by less than 3% while net operational cost significantly

Table 3
Summary of simulation results for sidestream (Temp 20 °C, DO 2 mg/L).

Parameters	Unit	CAS Value			Increment (%)		MLE Value			Increment (%)	
		FW0%	FW50%	FW100%	FW50%	FW100%	FW0%	FW50%	FW100%	FW50%	FW100%
Effluent quality											
BOD	mg/L	4	4	4	8	16	3	3	3	6	13
Total N	mg/L	33	33	33	0	0	19	15	11	−20	−40
Amm-N	mg/L	0.5	0.5	0.5	0	0	0.5	0.5	0.5	0	−2
NO3	mg/L	29	29	29	0	−1	16	12	8	−24	−48
NO2	mg/L	0.1	0.1	0.1	0	0	0.1	0.1	0.1	0	−8
NOx	mg/L	30	29	29	0	−1	16	12	9	−24	−47
Aeration and chemical demands											
Air demand	m^3/min	223	247	272	11	22	202	217	232	7	15
Aeration Power	kW	231	256	281	11	22	209	224	240	7	15
Fe dosage	kg/day	697	705	714	1	3	708	716	727	1	3
Biosolids Management											
Primary sludge	kg/day	4867	4867	4867	0	0	4867	4867	4867	0	0
TWAS	kg/day	4430	4765	5104	8	15	4141	4437	4722	7	14
MLSS	mg/L	2302	2472	2642	7	15	2185	2300	2442	5	12
Digested biosolids	kg/day	5893	6222	6548	6	11	5778	6083	6384	5	10
Methane production	m^3/day	2083	2478	2873	19	38	2014	2392	2769	19	38
Glycerol dosage											
Required for full denitrification	kg/day	NA	NA	NA	NA	NA	3519	2673	1853	NA	NA
Glycerol saved	kg/day	NA	NA	NA	NA	NA	0	847	1667	NA	NA
Operational cost											
Biosolids Disposal	$/day	589	621	654	6	11	577	608	637	5	10
Chemical Cost for dewatering	$/day	104	109	115	6	11	102	107	112	5	10
Chemical Cost for P removal	$/day	767	776	786	1	3	779	788	800	1	3
Chemical for P precipitation in AD	$/day	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Aeration blower cost	$/day	553	614	675	11	22	502	538	576	7	15
Methane energy	$/day	738	878	1018	19	38	713	847	981	19	38
Supplemental carbon to achieve same effluent TN (glycerol)	$/day	NA	NA	NA	NA	NA	0	373	735	NA	NA
Total operational cost	$/day	2041	2153	2266	5	11	2003	2090	2179	4	9
Net operational cost	$/day	1303	1275	1248	−2	−4	1289	869	464	−33	−64
Effluent quality											
BOD	mg/L	3	4	4	11	22	3	3	4	11	20
Total N	mg/L	14	13	13	−5	−10	10	9	8	−12	−22
Amm-N	mg/L	0.9	0.9	0.9	0	0	0.5	0.5	0.5	−2	−4
NO3	mg/L	10	10	9	−8	−15	7	6	5	−17	−32
NO2	mg/L	0.4	0.3	0.3	−3	−3	0.2	0.2	0.2	−11	−21
NOx	mg/L	11	10	9	−7	−14	7	6	5	−17	−32
Aeration and chemical demands											
Air demand	m^3/min	196	209	222	6	13	200	213	225	6	12
Aeration Power	kW	201	214	227	6	13	205	218	230	6	12
Fe dosage	kg/day	310	169	117	−45	−62	403	273	163	−32	−60
Biosolids Management											
Primary sludge	kg/day	4508	4508	4508	0	0	4220	4220	4220	0	0
TWAS	kg/day	3727	3893	4134	4	11	3695	3831	3990	4	8
MLSS	mg/L	1972	2059	2184	4	11	1948	2014	2103	3	8
Digested biosolids	kg/day	5741	6040	6341	5	10	5590	5879	6167	5	10
Methane production	m^3/day	2052	2442	2835	19	38	2029	2421	2809	19	38
Glycerol dosage											
Required for full denitrification	kg/day	2299	2126	1967	NA	NA	1578	1310	1068	NA	NA
Glycerol saved	kg/day	0	173	332	NA	NA	0	268	510	NA	NA
Operational cost											
Biosolids Disposal	$/day	573	603	633	5	10	558	587	616	5	10
Chemical Cost for dewatering	$/day	101	106	112	5	10	98	103	108	5	10
Chemical Cost for P removal	$/day	341	215	144	−37	−58	444	300	179	−32	−60
Chemical for P precipitation in AD	$/day	273	273	273	0	0	273	273	273	0	0
Aeration blower cost	$/day	483	514	546	6	13	493	523	553	6	12
Methane energy	$/day	727	865	1004	19	38	719	858	995	19	38
Supplemental carbon to achieve same effluent TN (glycerol)	$/day	0	76	146	NA	NA	0	118	225	NA	NA
Total operational cost	$/day	1831	1773	1774	−3	−3	1894	1811	1758	−4	−7
Net operational cost	$/day	1105	831	624	−25	−44	1175	836	538	−29	−54

declined by 24%–44% with FW addition.

The Bardenpho system with partial nitrification with/without FW addition achieved effluent BOD of 4 mg/L and TN of 7–8 mg/L. With FW addition, air demand increased by 7%–15% while Fe dosage was significantly reduced by 50%–80%. Similarly, methane and biosolids production were increased by 19%–38% and 5%–10%, respectively. Carbon credit as glycerol was estimated as 150–260 kg/day. The estimated total operational cost slightly decreased by 6%, and net operational cost substantially deceased by 35%–62% with FW addition. Simulation for partial nitrification with FW addition showed net operational cost savings for BNR processes of 24%–78% compared with the cases with MWW only. Upon comparing the data of Table 1 (DO of 2 mg/L) and Table 2 (DO of 0.5 mg/L), it is evident that operating at low DO achieved 25%, 28%, 35%–37%, and 36%–38% reduction in

Table 4
Summary of simulation results for sidestream (Temp 20 °C, DO 0.5 mg/L).

Parameters	Unit	CAS					MLE				
		Value			Increment (%)		Value			Increment (%)	
		FW0%	FW50%	FW100%	FW50%	FW100%	FW0%	FW50%	FW100%	FW50%	FW100%
Effluent quality											
BOD	mg/L	4	4	4	8	16	3	3	3	6	13
Total N	mg/L	30	29	29	−2	−3	16	12	8	−26	−48
Amm-N	mg/L	0.7	0.7	0.7	0	0	0.7	0.7	0.7	−1	−3
NO3	mg/L	26	25	25	−2	−4	13	9	5	−32	−61
NO2	mg/L	0.4	0.4	0.4	0	0	0.4	0.4	0.4	−3	−5
NOx	mg/L	26	26	25	−2	−4	13	9	5	−32	−60
Aeration and chemical demands											
Air demand	m^3/min	167	185	203	11	22	146	156	167	7	14
Aeration Power	kW	174	193	212	11	22	152	163	174	7	15
Fe dosage	kg/day	697	705	714	1	3	708	716	727	1	3
Biosolids Management											
Primary sludge	kg/day	4867	4867	4867	0	0	4867	4867	4867	0	0
TWAS	kg/day	4436	4771	5111	8	15	4144	4440	4718	7	14
MLSS	mg/L	2306	2475	2646	7	15	2187	2302	2440	5	12
Digested biosolids	kg/day	5894	6224	6550	6	11	5780	6085	6385	5	10
Methane production	m^3/day	2087	2481	2876	19	38	2014	2394	2766	19	37
Glycerol dosage											
Required for full denitrification	kg/day	NA	NA	NA	NA	NA	2865	1949	1132	NA	NA
Glycerol saved	kg/day	NA	NA	NA	NA	NA	0	916	1734	NA	NA
Operational cost											
Biosolids Disposal	$/day	589	622	654	6	11	577	608	638	5	10
Chemical Cost for dewatering	$/day	104	109	115	6	11	102	107	112	5	10
Chemical Cost for P removal	$/day	767	776	786	1	3	779	788	800	1	3
Chemical for P precipitation in AD	$/day	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Aeration blower cost	$/day	418	462	508	11	22	364	390	417	7	15
Methane energy	$/day	739	879	1019	19	38	713	848	980	19	37
Supplemental carbon to achieve same effluent TN (glycerol)	$/day	NA	NA	NA	NA	NA	0	404	764	NA	NA
Total operational cost	$/day	1906	2002	2099	5	10	1865	1942	2021	4	8
Net operational cost	$/day	1166	1123	1081	−4	−7	1152	690	277	−40	−76
Effluent quality											
BOD	mg/L	4	4	4	10	20	4	4	4	7	14
Total N	mg/L	12	12	11	−5	−9	8	7	6	−13	−22
Amm-N	mg/L	2	2	2	−1	−2	1	1	1	−2	−2
NO3	mg/L	0.1	0.1	0.1	20	20	0.03	0.03	0.03	0	0
NO2	mg/L	8	8	7	−9	−17	4	3	3	−25	−43
NOx	mg/L	8	8	7	−9	−16	4	3	3	−24	−43
Aeration and chemical demands											
Air demand	m^3/min	125	135	145	8	16	126	135	145	8	15
Aeration Power	kW	128	139	149	8	16	129	139	149	7	15
Fe dosage	kg/day	137	88	69	−37	−50	227	142	51	−38	−78
Biosolids Management											
Primary sludge	kg/day	4508	4508	4508	0	0	4220	4220	4220	0	0
TWAS	kg/day	3509	3734	4091	6	17	3471	3670	3856	6	11
MLSS	mg/L	1862	2000	2164	7	16	1838	1941	2037	6	11
Digested biosolids	kg/day	5720	6026	6337	5	11	5558	5849	6140	5	10
Methane production	m^3/day	2067	2455	2846	19	38	2042	2430	2817	19	38
Glycerol dosage											
Required for full denitrification	kg/day	1050	958	880	NA	NA	562	425	323	NA	NA
Glycerol saved	kg/day	0	93	170	NA	NA	0	137	239	NA	NA
Operational cost											
Biosolids Disposal	$/day	571	602	633	5	11	555	584	613	5	10
Chemical Cost for dewatering	$/day	101	106	111	5	11	97	102	107	5	10
Chemical Cost for P removal	$/day	151	96	76	−37	−50	250	156	56	−38	−78
Chemical for P precipitation in AD	$/day	273	273	273	0	0	273	273	273	0	0
Aeration blower cost	$/day	308	333	358	8	16	310	334	357	7	15
Methane energy	$/day	732	870	1008	19	38	723	861	998	19	38
Supplemental carbon to achieve same effluent TN (glycerol)	$/day	0	41	75	NA	NA	0	60	105	NA	NA
Total operational cost	$/day	1447	1460	1510	1	4	1508	1472	1431	−2	−5
Net operational cost	$/day	715	549	428	−23	−40	785	551	328	−30	−58

aeration energy costs for CAS, MLE, A2O, and Bardenpho, respectively. The aforementioned benefit occurred simultaneously with lower effluent TN concentrations.

3.3. Emerging BNR (sidestream N removal process with mainstream DO of 2 mg/L)

A summary of simulation results for sidestream N removal process at a mainstream DO of 2 mg/L is presented in Table 3. CAS employed with sidestream process showed that the estimated effluent quality of BOD and TN was close respectively at 4 mg/L and 33 mg/L with/without FW addition. With FW addition, air demand increased by 11%–22% with an insignificant change of iron dosage. FW addition increased methane and biosolids production by 19%–38% and 6%–11%, respectively. The estimated total operational cost increased by less than 11% with a decrease for the net operational cost by less than 4%.

MLE process simulation achieved similar effluent BOD levels for the scenarios with/without FW supplementation but effluent TN levels decreased by 20%–40% with the addition of FW. With FW addition, air demand increased by 7%–15% and iron dosage was minimally changed. Methane production and biosolids generation with FW addition increased by 19%–38% and 5%–10%, respectively, compared to no FW. The carbon credit was 850–1670 kg/day. The estimated total operational cost increased by 4%–9% with FW addition while the net operational cost decreased by 33%–64%.

A2O simulation showed that effluent BOD and TN levels were 3–4 mg/L and 13–14 mg/L, respectively. With FW, air demand increased by 6%–13% with a significant drop for iron dosage by 45%–62%. Relative to no FW, methane and biosolids generation increased by 19%–38% and 5%–10%, respectively. Addition of FW can save 170–330 kg/day of glycerol. Total operational costs were close for scenarios with and without FW addition within 3% while net operational cost dropped by 25%–44% with the addition of FW.

The simulation for Bardenpho process employed with sidestream process resulted in BOD concentrations of 3–4 mg/L and TN of 8–10 mg/L, with an increasing air demand by 12%–22% and a decreasing iron dosage by 32%–60% upon FW addition. Methane and digested biosolid production increased by 7%–13% and 5%–10%, respectively. Glycerol dose declined by 270–510 kg/day through FW addition. Moreover, compared with a case without FW addition, FW addition scenarios showed a slight drop for total operational cost by 4%–7% and a substantial decline in net operational cost by 29%–54%.

3.4. Emerging BNR (sidestream N removal process with mainstream partial nitrification)

Results of simulation for sidestream processes with mainstream partial nitrification are summarized in Table 4. Effluent BOD and TN levels for CAS was 4 mg/L and 29–30 mg/L, respectively, similar for cases with/without FW supplementation. FW addition scenarios increased air demand by 11%–22% with an insignificant increase in Fe dosage. With FW, methane production also increased by 19%–38% with an increase in biosolids production by 6%–11%. With FW addition, total operational cost increased by 5%–10% while net operational cost decreased by less than 7%.

The MLE process achieved effluent BOD and TN levels of 3 mg/L and 8–16 mg/L with a significant decrease in TN levels by 26%–48% with FW addition. FW supplementation cases increased air demand by 7%–14% relative to no FW, with an insignificant change in Fe consumption. Methane and biosolids production were higher by 19%–37% and 5%–10%, respectively. Compared with municipal wastewater alone, FW addition increased total operational cost by 4%–8% with a significant reduction in net operational cost (40%–76%).

Similarly, effluent BOD and TN concentrations for the A2O process were 4 mg/L and 11–12 mg/L. Air demand increased by 8%–16% with a decrease in Fe dosage by 37%–50%, with FW addition. Methane and biosolids production were higher by 19%–38% and 10%, respectively, compared to no FW. FW addition increased total operational cost by less than 4% and decreased net operational cost by 23%–40%.

Moreover, Bardenpho process simulation estimated that effluent BOD and TN concentrations were 4 mg/L and 6–8 mg/L, respectively. Air demand increased by 8%–15% while Fe dosage decreased by 38%–78%. Methane and biosolids production increased by 19%–38%. FW addition cases decreased total operational cost by 5% and net operational cost substantially by 30%–58%.

Wastewater treatment simulations for the impact of FW on emerging BNR processes showed that FW addition reduces net operational cost by 25%–64% for mainstream DO of 2 mg/L and 23%–76% for mainstream partial nitrification, for the three BNR processes, with negligible benefits (< 7%) for the CAS.

3.5. Overall impact of FW addition on WWTP performance

Summary of the relative change in performance with FW addition to results without FW addition at a mainstream DO of 2 mg/L are presented in Fig. 1. Systems with a typical operation of DO of 2 mg/L showed that FW addition increased effluent BOD levels by < 0.6 mg/L

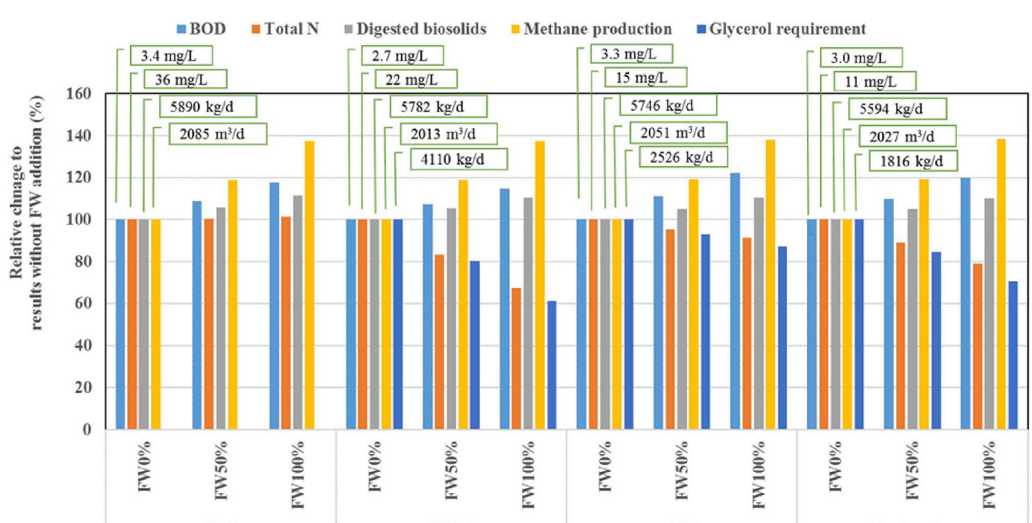

Fig. 1. Impact of FW addition on WWTP performance (typical operational condition).

for CAS, < 0.4 mg/L for MLE, < 0.7 mg/L for A2O, and < 0.6 mg/L for Bardenpho while TN levels were close for CAS and decreased by 4–7 mg/L for MLE, < 1.3 mg/L for A2O, and 1.2–2.3 mg/L for Bardenpho. Similarly, with FW addition, digested sludge production and methane production increased by 277–669 kg/d (5%–11%) and 382–783 m^3/d (19%–38%), respectively for the four systems while glycerol requirement decreased by 800–1589 kg/d (19%–39%) for MLE, 172–322 kg/d (7%–13%) for A2O, and 277–531 kg/d (15%–29%) for Bardenpho (Fig. 1).

Systems employing mainstream partial nitrification also showed that FW addition increased effluent BOD by 0.2–0.7 mg/L for the four systems and decreased effluent TN by < 0.5 mg/L for CAS, 4–7.8 mg/L for MLE, 0.6–1.1 mg/L for A2O, and 1.1–1.9 mg/L for Bardenpho (Fig. 2). In addition, biosolids production and methane production increased with FW addition by 286–657 kg/d (5%–11%) and 380–707 m^3/d (19%–38%), respectively while glycerol requirement decreased by 874–1727 kg/d (25%–50%) for MLE, 88–161 kg/d (8%–14%) for A2O, and 146–263 kg/d (22%–40%) for Bardenpho.

As shown in Fig. 3, sidestream processes combined with mainstream DO of 2 mg/L also showed that FW supplement increased effluent BOD levels by 0.2–0.7 mg/L for the four systems and decreased effluent TN levels by 3.8–7.5 mg/L for MLE, 0.7–1.3 mg/L for A2O, and 1.2–2.2 mg/L for Bardenpho with no change for CAS. FW addition increased biosolids and biogas production by 289–655 kg/d (5%–11%) and 379–780 m^3/d (19%–38%), respectively whereas glycerol demand decreased with FW inclusion by 847–1667 kg/d (24%–47%) for MLE, 173–332 kg/d (8%–14%) for A2O, and 268–510 kg/d (17%–32%) for Bardenpho.

Sidestream processes with main partial nitrification (Fig. 4) showed that FW addition scenarios increased effluent BOD by 0.2–0.7 mg/L for all four systems and decreased TN levels by 0.5–1 mg/L for CAS, 4.2–7.9 mg/L for MLE, 0.7–1.2 mg/L for A2O, and 1–1.7 mg/L for Bardenpho. Biosolids production and biogas production increased with FW addition by 291–656 kg/d (5%–11%) and 379–789 m^3/d (19%–38%), respectively while FW supplement reduced glycerol requirement by 916–1734 kg/d (32%–61%) (MLE), 93–170 kg/d (9%–16%) (A2O), and 137–239 kg/d (24%–43%) (Bardenpho).

Overall, FW addition in BNR processes decreased effluent TN by 3.6–7.9 mg/L for MLE, 0.6–1.3 mg/L for A2O, and 1–2.3 mg/L for Bardenpho, indicating that FW addition enhances nutrient removal.

In order to maintain the clarity of Figs. 1–4 above, the impact of FW on effluent TP was not shown. Effluent TP concentrations for CAS and MLE without chemical P removal were 6.4–6.5 mg/L with and without FW addition at a mainstream DO of 2 mg/L and 0.5 mg/L. In contrast, effluent TP for A2O at a mainstream DO of 2 mg/L was 3.2 mg/L without FW addition and 1.3–1.9 mg/L with FW supplementation. Similarly, A2O employing mainstream partial nitrification achieved effluent TP of 1.3 mg/L without FW and 0.8–1 mg/L with FW addition. Without iron addition, effluent TP for the Bardenpho process with a mainstream DO of 2 mg/L was 4.1 mg/L without FW addition and 1.6–2.8 mg/L with FW supplement. Bardenpho system with mainstream partial nitrification showed effluent TP of 1.7 mg/L for MWW alone and 0.7–1 mg/L for FW addition. It indicates that P removal for CAS and MLE without chemical addition was close with and without FW addition while A2O and Bardenpho increased P removal with FW addition by 0.4–1.9 mg/L and 0.7–2.5 mg/L, respectively.

This trend was similar for the four systems with sidestream N removal. Effluent TP concentrations for CAS and MLE with sidestream N removal were 6.3–6.5 mg/L at mainstream DOs of 2 mg/L and 0.5 mg/L, with and without FW addition. A2O employing sidestream process showed effluent TP of 1.2–3 mg/L at a mainstream DO of 2 mg/L and 0.8–1.3 mg/L for mainstream partial nitrification with the higher level obtained from simulation without FW. Similarly, Bardenpho systems also showed 3.9 mg/L (without FW) and 1.5–2.6 mg/L (with FW) at a DO level of 2 mg//L and 1.6 mg/L (without FW) and 0.8–1 mg/L (with FW) for partial nitrification.

Effluent TN concentrations for the four processes were close in all scenarios with and without chemical P removal with a difference of less than 0.2 mg/L.

3.6. Comparison of total operational cost and net operational cost between the different scenarios

The scrutiny of operational cost data indicated that the total operational costs breakdown for the different systems with and without FW addition are: 20%–31% for aeration, 35%–42% for iron dosage, and 29%–32% for biosolids disposal for non-EBPR systems i.e. CAS and MLE. For A2O and Bardenpho the breakdown showed 21%–33% for aeration, 4%–25% for iron dosage, 14%–19% for chemical P removal in anaerobically digested sludge, and 29%–44% for biosolids disposal, indicating these parameters are the major factors for total operational costs.

The impact of the different systems on the costs for aeration, chemical P removal, and biosolids treatment varied between system types and operational conditions. As presented in Table A.5, compared with the operational condition of DO 2 mg/L with and without FW addition,

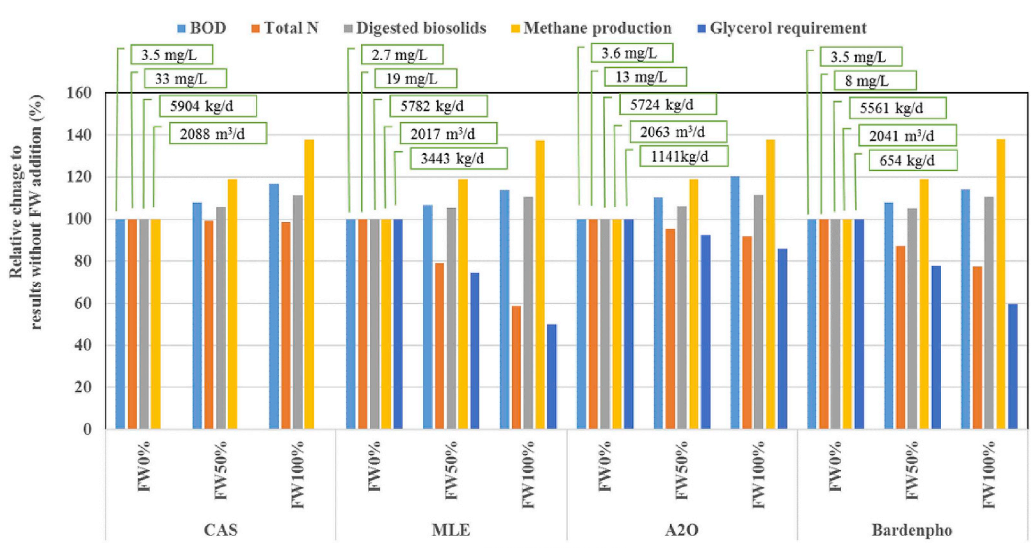

Fig. 2. Impact of FW addition on WWTP performance (partial nitrification condition).

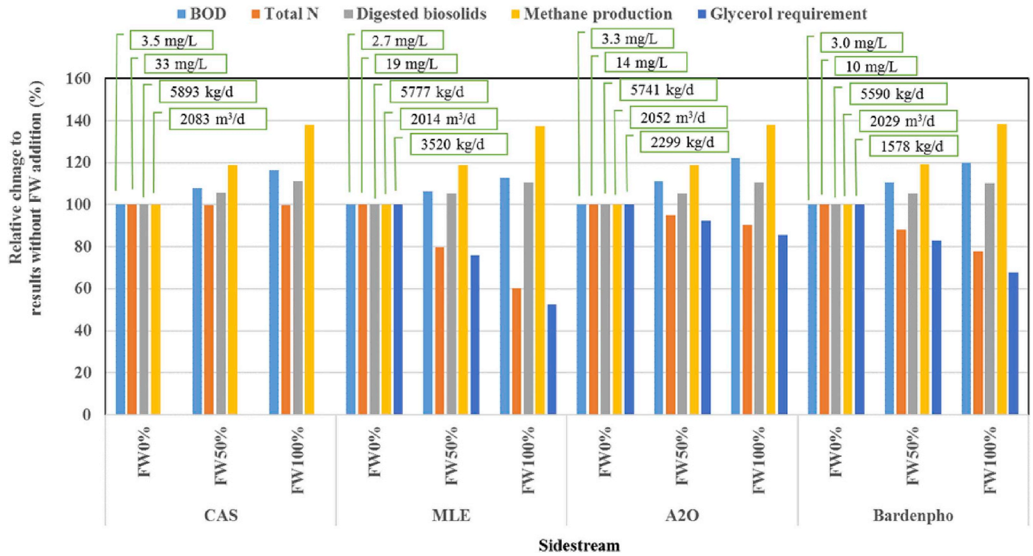

Fig. 3. Impact of FW addition on WWTP performance (sidestream with mainstream DO 2 mg/L).

partial nitrification conditions saved iron dosage cost minimally but reduced aeration cost by 25%–28% for CAS and MLE. However, for A2O and Bardenpho, partial nitrification reduced iron dosing and aeration costs by 47%–71% and 35%–38%, respectively. In contrast, sidestream N treatment, compared with the typical operational condition of DO of 2 mg/L, decreased aeration cost by 3%–5% for the four processes, with insignificant change in iron dosage cost for CAS and MLE and 4%–7% lower iron dosage for A2O and Bardenpho, indicating low cost savings through sidestream treatment. Furthermore, the combination of partial nitrification and sidestream treatment, relative to the typical operational condition (DO 2 mg/L), decreased aeration cost by 28%–31% for CAS and MLE, and 36%–39% for A2O and Bardenpho, but affected insignificant iron cost change for CAS and MLE and 47%–67% lower for A2O and Bardenpho. Biosolids disposal cost was close for all different operational conditions for the four processes. Thus, it can be suggested that partial nitrification processes can effectively decrease the total operational costs for CAS and BNR processes.

Similarly, methane energy costs for partial nitrification, sidestream, and the combination of the two with/without FW addition were similar to the values from the typical conditions of DO of 2 mg/L for the four processes. In contrast, compared with a DO of 2 mg/L, carbon credit cost with FW addition for partial nitrification, sidestream, and the combination of the two with/without FW addition increased by 5%–14% for MLE, decreased by 46%–55% for A2O and Bardenpho employed with partial nitrification with/without sidestream. The carbon credit for A2O and Bardenpho with sidestream treatment only was similar to the baseline operation at a DO of 2 mg/L (Table A5).

Using CAS scenario without FW addition at a mainstream DO of 2 mg/L as a baseline, relative percentage of total operational cost for the different scenarios involving partial nitrification and sidestream N treatment with and without FW is presented in Fig. 5. Compared with CAS with MWW alone, most scenarios for MLE, A2O and Bardenpho showed a lower total operational cost by up to 31% while CAS cases showed from an increase by 11% to a decrease by 7%. As apparent from Fig. 5a, for CAS plants partial nitrification achieves about 7% reduction in total operational costs, sidestream treatment for nitrogen increased total operating costs by close to 0%–11% with and without FW, and the combination achieves −7% to 3%. Particularly, the increased total operational costs for sidestream were due to the increased biosolids disposal and aeration costs by 5% and 16%, respectively compared with CAS with MWW alone.

It is also evident from Fig. 5 that the total and net operating costs for

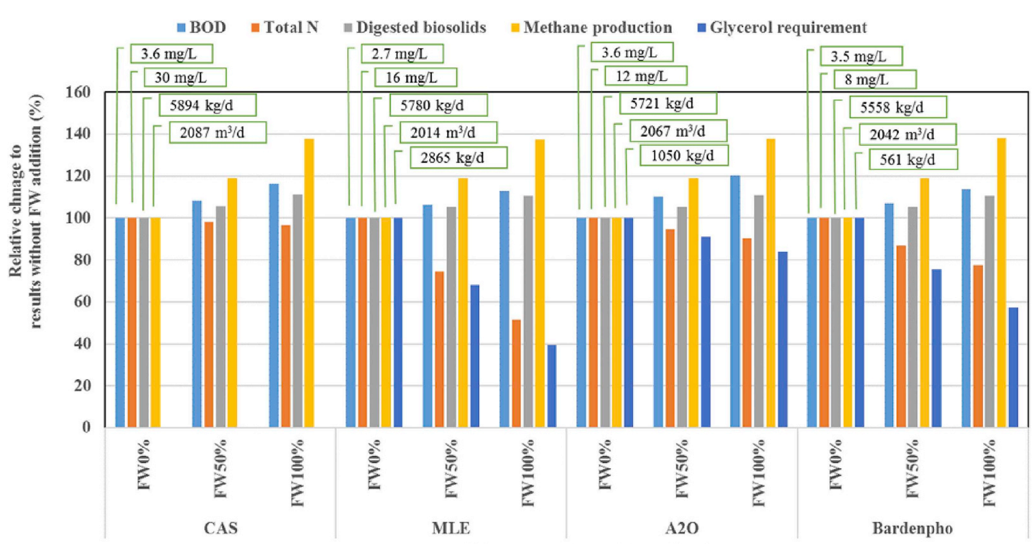

Fig. 4. Impact of FW addition on WWTP performance (sidestream with mainstream partial nitrification).

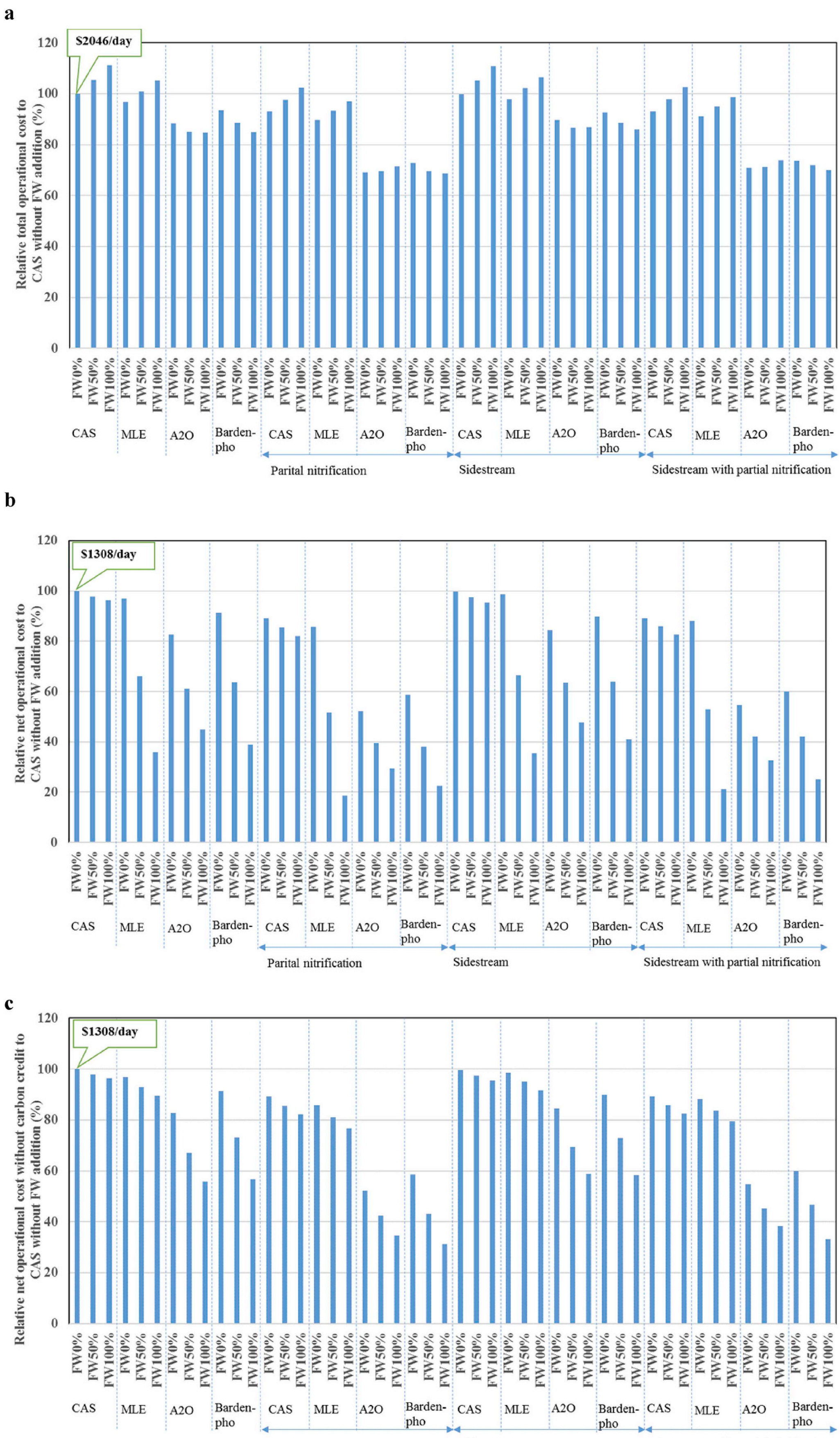

Fig. 5. Comparison of total operational cost and net operational cost between different scenarios.

Table 5
A summary of different PT removal efficiencies on impact of FW (a) MLE (b) A2O (c) Bardenpho.

(a) Parameters	Unit	MLE FW0% Value			Increment (%)		MLE FW50% Value			Increment (%)		MLE FW100%Value			Increment (%)	
		PT45%	PT65%	PT85%	PT45%	PT85%	PT45%	PT65%	PT85%	PT45%	PT85%	PT45%	PT65%	PT85%	PT45%	PT85%
Effluent quality																
BOD	mg/L	3	3	2	12	−12	3	3	3	11	−11	3	3	3	10	−10
Total N	mg/L	21	22	23	−5	8	17	18	20	−6	8	14	15	16	−7	10
Amm-N	mg/L	0.5	0.5	0.5	0	2	0.5	0.5	0.5	0	0	0.5	0.5	0.5	0	2
NO3	mg/L	18	19	21	−6	10	14	15	17	−7	11	10	12	13	−9	14
NO2	mg/L	0.1	0.1	0.1	0	0	0.1	0.1	0.1	−7	0	0.1	0.1	0.1	0	8
NOx	mg/L	18	19	21	−6	10	14	15	17	−7	11	11	12	13	−9	14
Aeration and chemical demands																
Air demand	m^3/min	241	215	192	12	−11	257	231	208	11	−10	274	248	224	11	−10
Aeration Power	kW	244	218	194	12	−11	260	234	210	11	−10	277	250	226	11	−10
Fe dosage	kg/day	747	708	675	6	−5	753	716	681	5	−5	787	727	691	8	−5
Biosolids Management																
Primary sludge	kg/day	3369	4867	6364	−31	31	3369	4867	6364	−31	31	3369	4867	6364	−31	31
TWAS	kg/day	4938	4149	3342	19	−19	5245	4447	3643	18	−18	5570	4729	3930	18	−17
MLSS	mg/L	2609	2187	1761	19	−19	2722	2303	1886	18	−18	2883	2445	2031	18	−17
Digested biosolids	kg/day	5519	5782	6028	−5	4	5823	6086	6337	−4	4	6157	6391	6641	−4	4
Methane production	m^3/day	1746	2013	2269	−13	13	2122	2395	2653	−11	11	2496	2770	3030	−10	9
Glycerol dosage																
Required for full denitrification	kg/day	3863	4110	4502	−6	10	3065	3309	3666	−7	11	2285	2520	2865	−9	14
Glycerol saved	kg/day	0	0	0	0	0	799	800	835	0	4	1579	1589	1636	−1	3
Operational cost																
Biosolids Disposal	$/day	551	577	602	−5	4	582	608	633	−4	4	615	638	663	−4	4
Chemical Cost for dewatering	$/day	97	102	106	−5	4	102	107	111	−4	4	108	112	117	−4	4
Chemical Cost for P removal	$/day	822	779	743	6	−5	829	788	750	5	−5	865	800	761	8	−5
Chemical for P precipitation in AD	$/day	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Aeration blower cost	$/day	585	523	466	12	−11	624	561	503	11	−10	665	601	542	11	−10
Methane energy	$/day	619	713	804	−13	13	751	848	940	−11	11	884	981	1073	−10	9
Supplemental carbon to achieve same effluent TN (glycerol)	$/day	NA	NA	NA	NA	NA	352	353	368	0	4	696	701	721	−1	3
Total operational cost	$/day	2055	1980	1917	4	−3	2138	2065	1998	4	−3	2254	2152	2083	5	−3
Net operational cost	$/day	1437	1267	1114	13	−12	1034	863	690	20	−20	674	470	288	43	−39

(b) Parameters	Unit	A2O FW0% Value			Increment (%)		A2O FW50% Value			Increment (%)		A2O FW100% Value			Increment (%)	
		PT45%	PT65%	PT85%	PT45%	PT85%	PT45%	PT65%	PT85%	PT45%	PT85%	PT45%	PT65%	PT85%	PT45%	PT85%
Effluent quality																
BOD	mg/L	4	3	3	13	−14	4	4	3	12	−12	4	4	4	11	−11
Total N	mg/L	14	15	16	−4	5	14	14	15	−4	5	13	14	14	−4	5
Amm-N	mg/L	0.9	0.9	0.9	−1	0	0.9	0.9	0.9	0	1	0.9	0.9	0.9	0	1
NO3	mg/L	11	11	12	−6	8	10	11	11	−7	8	9	10	11	−8	8
NO2	mg/L	0.4	0.4	0.4	0	0	0.3	0.4	0.4	−3	0	0.3	0.4	0.4	−3	0
NOx	mg/L	11	12	13	−6	7	10	11	12	−7	8	10	10	11	−7	8
Aeration and chemical demands																
Air demand	m^3/min	228	205	184	11	−10	241	218	196	11	−10	254	231	209	10	−10
Aeration Power	kW	231	207	186	11	−10	244	220	198	11	−10	257	234	212	10	−10
Fe dosage	kg/day	356	330	306	8	−7	227	210	192	8	−8	151	140	124	8	−11

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Table 5 (*continued*)

(b) Parameters	Unit	A2O FW0% Value			Increment (%)		A2O FW50% Value			Increment (%)		A2O FW100% Value			Increment (%)	
		PT45%	PT65%	PT85%	PT45%	PT85%	PT45%	PT65%	PT85%	PT45%	PT85%	PT45%	PT65%	PT85%	PT45%	PT85%
Biosolids Management																
Primary sludge	kg/day	3129	4508	5883	−31	30	3129	4508	5883	−31	30	3129	4508	5883	−31	30
TWAS	kg/day	4653	3755	2867	24	−24	4794	3913	3036	23	−22	5015	4145	3273	21	−21
MLSS	mg/L	2463	1986	1515	24	−24	2537	2069	1605	23	−22	2650	2190	1729	21	−21
Digested biosolids	kg/day	5532	5746	5958	−4	4	5840	6039	6252	−3	4	6152	6343	6553	−3	3
Methane production	m^3/day	1809	2051	2260	−12	10	2196	2444	2656	−10	9	2582	2835	3051	−9	8
Glycerol dosage																
Required for full denitrification	kg/day	2367	2526	2718	−6	8	2187	2354	2537	−7	8	2037	2204	2380	−8	8
Glycerol saved	kg/day	NA	NA	NA	NA	NA	180	172	181	4	5	330	322	337	3	5
Operational cost																
Biosolids Disposal	$/day	552	574	595	−4	4	583	603	624	−3	4	614	633	654	−3	3
Chemical Cost for dewatering	$/day	97	101	105	−4	4	103	106	110	−3	4	108	112	115	−3	3
Chemical Cost for P removal	$/day	392	363	337	8	−7	250	231	212	8	−8	166	154	137	8	−11
Chemical for P precipitation in AD	$/day	273	273	273	0	0	273	273	273	0	0	273	273	273	0	0
Aeration blower cost	$/day	554	498	447	11	−10	584	529	476	11	−10	617	561	508	10	−10
Methane energy	$/day	641	727	801	−12	10	778	866	941	−10	9	915	1004	1081	−9	8
Supplemental carbon to achieve same effluent TN (glycerol)	$/day	NA	NA	NA	NA	NA	79	76	80	4	5	146	142	149	3	5
Total operational cost	$/day	1868	1808	1755	3	−3	1793	1741	1695	3	−3	1778	1733	1687	3	−3
Net operational cost	$/day	1227	1081	955	13	−12	936	800	674	17	−16	718	587	458	22	−22

(c) Parameters	Unit	Bardenpho FW0% Value			Increment (%)		Bardenpho FW50% Value			Increment (%)		Bardenpho FW100% Value			Increment (%)	
		PT45%	PT65%	PT85%	PT45%	PT85%	PT45%	PT65%	PT85%	PT45%	PT85%	PT45%	PT65%	PT85%	PT45%	PT85%
Effluent quality																
BOD	mg/L	3	3	3	1	3	3	3	3	0	1	4	4	4	−1	3
Total N	mg/L	10	11	12	−11	10	9	10	11	−11	11	8	9	10	−12	11
Amm-N	mg/L	0.5	0.5	0.8	0	47	0.5	0.5	0.5	2	0	0.5	0.5	0.5	4	2
NO3	mg/L	7	8	9	−14	7	6	7	8	−17	15	5	6	7	−19	17
NO2	mg/L	0.2	0.2	0.5	−5	163	0.2	0.2	0.2	0	6	0.2	0.2	0.2	−6	6
NOx	mg/L	7	8	9	−14	11	6	7	8	−16	15	5	6	7	−19	17
Aeration and chemical demands																
Air demand	m^3/min	232	210	187	11	−11	244	222	201	10	−10	257	235	213	9	−9
Aeration Power	kW	235	212	189	11	−11	247	225	203	10	−10	259	237	216	9	−9
Fe dosage	kg/day	445	431	445	3	3	293	283	283	3	0	162	156	159	4	2
Biosolids Management																
Primary sludge	kg/day	2929	4220	5507	−31	30	2929	4220	5507	−31	30	2929	4220	5507	−31	30
TWAS	kg/day	4607	3733	2889	23	−23	4704	3839	3006	23	−22	4829	3972	3145	22	−21
MLSS	mg/L	2430	1968	1525	23	−23	2481	2024	1584	23	−22	2545	2093	1653	22	−21
Digested biosolids	kg/day	5399	5594	5784	−3	3	5695	5871	6064	−3	3	5999	6160	6342	−3	3
Methane production	m^3/day	1777	2027	2242	−12	11	2163	2421	2653	−11	10	2546	2807	3054	−9	9
Glycerol dosage																
Required for full denitrification	kg/day	1555	1816	1983	−14	9	1284	1539	1775	−17	15	1042	1285	1506	−19	17
Glycerol saved	kg/day	NA	NA	NA	NA	NA	271	277	207	−2	−25	513	531	477	−3	10
Operational cost																
Biosolids Disposal	$/day	539	559	578	−3	3	569	586	606	−3	3	599	615	633	−3	3
Chemical Cost for dewatering	$/day	94	98	101	−3	3	100	103	106	−3	3	105	108	111	−3	3

(*continued on next page*)

Table 5 (*continued*)

(c) Parameters	Unit	Bardenpho FW0% Value			Increment (%)		Bardenpho FW50% Value			Increment (%)		Bardenpho FW100% Value			Increment (%)	
		PT45%	PT65%	PT85%	PT45%	PT85%	PT45%	PT65%	PT85%	PT45%	PT85%	PT45%	PT65%	PT85%	PT45%	PT85%
Chemical Cost for P removal	$/day	489	475	489	3	3	322	311	311	3	0	179	172	175	4	2
Chemical for P precipitation in AD	$/day	273	273	273	0	0	273	273	273	0	0	273	273	273	0	0
Aeration blower cost	$/day	563	509	454	11	−11	593	539	488	10	−10	623	569	517	9	−9
Methane energy	$/day	629	718	794	−12	11	766	858	940	−11	10	902	994	1082	−9	9
Supplemental carbon to achieve same effluent TN (glycerol)	$/day	NA	NA	NA	NA	NA	120	122	91	−2	−25	226	234	210	−3	10
Total operational cost	$/day	1959	1913	1895	2	−1	1856	1812	1783	2	−2	1778	1737	1709	2	−2
Net operational cost	$/day	1330	1195	1101	11	−8	970	832	752	17	−10	650	508	417	28	−18

the three BNR process for MWW only are significantly lower than the CAS primarily due to significant reduction in iron costs. In addition, net operational cost comparison showed that the lowest net operational cost compared with CAS with MWW alone was observed for BNR partial nitrification systems with FW 100% and without sidestream N treatment, indicating that FW addition benefit is maximized for BNR processes. Similarly, the lowest net operational cost without carbon credit relative to CAS with no FW was also observed for A2O and Bardenpho employed with partial nitrification with FW 100%.

Using aeration energy and methane production data, the net change in aeration energy (kWh/d) and methane energy (kWh/d) for FW added scenarios relative to the cases without FW addition was calculated according to Eqs (1)–(5). The positive and negative net energy values in Fig. A.2 indicate energy consumption and recovery, respectively.

$$\text{Aeration energy (kWh/d)} = \text{aeration (kW)} \times 24\,\text{h} \quad (1)$$

$$\text{Net aeration energy (kWh/d)} = \text{aeration energy for FWadded} - \text{aeration energy for 0\%FW} \quad (2)$$

$$\text{Methane energy (kWh/d)} = \text{methane } m^3/d \times 35.8\,\text{MJ}/m^3 \times 1\,\text{kWh}/3.6\,\text{MJ} \quad (3)$$

$$\text{Net methane energy (kWh/d)} = \text{methane energy for FWadded} - \text{methane energy for 0\%FW} \quad (4)$$

$$\text{Net total energy impact (kWh/d)} = \text{net aeration energy} + \text{net methane energy} \quad (5)$$

It is evident from Fig. 1, that as expected, irrespective of the secondary biological treatment technology, the additional energy from anaerobic digestion of FW solids more than offsets the marginal increase in aeration demand, thus implying that addition of FW is almost inevitable if energy-neutrality is goal. The addition of FW can generate up to an additional 0.21 kWh/m^3 based on the 100% penetration of FW grinders scenario.

Simulation at DO 2 mg/L estimated that the net total energy gain for FW addition, calculated by summing up the net change in aeration energy and methane energy, ranged from 3300 to 6500 kWh/d for CAS, 3450 to 6800 kWh/d for MLE, 3600 to 7200 kWh/d for A2O, and 3600 to 7200 kWh/d for Bardenpho (Fig. A.2a).

The estimated net total energy gain for FW separate addition for partial nitrification systems was 3500–6900 kWh/d for CAS, 3500–6900 kWh/d for MLE, 4300–7900 kWh/d for A2O, and 3600–7200 kWh/d for Bardenpho (Fig. A.2b).

The estimated net total energy gain for FW addition for systems with sidestream N removal was 3300–6600 kWh/d for CAS, 3400–6800 kWh/d for MLE, 3600–7200 kWh/d for A2O, and 3600–7200 kWh/d for Bardenpho (Fig. A.3c).

The estimated net total energy gain for FW addition for sidestream with mainstream partial nitrification was 3500–6900 kWh/d for CAS, 3500–6900 kWh/d for MLE, 3600–7300 kWh/d for A2O, and 3600–7200 kWh/d for Bardenpho, indicating the energy gain for all cases (Fig. A.3d).

3.7. Sensitivity of FW impact to carbon diversion efficiency

In order to examine sensitivity of FW impact, carbon diversion scenarios were simulated for the three BNR systems i.e. MLE, A2O, and Bardenpho operating at a DO of 2 mg/L without sidestream N removal, using different primary treatment suspended solids removal efficiencies (PTE) i.e. 45%, 65%, and 85% (Table 5).

Effluent BOD levels for MLE process were 2–3 mg/L for scenarios with different FW addition and PTEs. Effluent TN levels increased by 2–3 mg/L as PTE increased from 45% to 85%. Since all previous analysis had been performed at a PTE of 65%, this represents the baseline for comparative assessment of primary treatment efficiencies. Compared with PTE of 65%, PTE 45% increased air demand by 12%

while PTE 85% decreased it by 11%. A similar tendency was seen with iron dosage. PTE of 45% scenarios increased aeration energy by 11%–12% and MLSS by 18%–19% while methane production and digested biosolids decreased by 10%–13% and 4%–5%, respectively. PTE of 85% for the MLE cases increased digested biosolids by 4% and methane production by 9%–13% while aeration energy and MLSS decreased by 10%–11% and 17%–19%, respectively. Relative to MWW alone scenarios, the estimated net total energy gain for FW separate addition considering both the net change in aeration energy and methane energy was 3300–6700 kWh/d for PTE of 45%, and 3500–6900 kWh/d for PTE of 85%, indicating marginally higher energy gain for higher PTE.

Similarly, for the A2O, PTE of 45% scenarios increased MLSS by 21%–24% and aeration energy by 10%–11%, and decreased methane production by 9%–12% and biosolids production by less than 4%. In contrast, PTE of 85% for A2O decreased aeration energy by 10% and MLSS by 21%–24% whereas methane production and biosolids production increased by 8%–10% and less than 4%, respectively. The estimated net total energy gain for FW separate addition relative to the MWW alone scenarios was 3500–7100 kWh/d for PTE of 45%, and 3600–7200 kWh/d for PTE of 85%, indicating the slightly higher energy gain for higher PTE.

Similar patterns were also observed for Bardenpho. Compared with a PTE of 65%, PTE of 45% scenarios increased aeration energy (9%–11%) and MLSS (22%–23%), and decreased methane production (9%–12%) and biosolids production (< 3%). PTE of 85% cases increased methane production (9%–11%) and biosolids (3%) while aeration energy (9%–11%) and MLSS (21%–23%) were decreased. The estimated net total energy gain for separate FW addition relative to the MWW alone scenarios was 3500–7100 kWh/d for PTE of 45%, and 3800–7400 kWh/d for PTE of 85%, indicating the higher energy gain for higher PTE.

Compared with MWW alone cases, the segregated FW scenarios for MLE, A2O, and Bardenpho at a mainstream DO of 2 mg/L without sidestream N removal decreased net operational costs by 24%–53% (PTE of 45%), 26%–63% (PTE of 65%), and 29%–74% (PTE of 85%). This indicates not only the positive impact of the segregated FW on net operational costs for carbon diversion but also that higher carbon diversion efficiencies affected more pronounced savings in net operational costs. It should be noted that generally as PTE increased, iron requirement decreased due to the particulate phosphorus in the influent contributing to roughly 50% of the total P. While in all 3 cases, energy generation increased with PTE increasing, the glycerol credit decreased as PTE increased, due to a lower ratio of rbCOD:N in the primary effluent.

4. Conclusions

This study presented that FW addition to MLE, A2O, and Bardenpho processes decreased effluent nitrogen by 3.6–7.9 mg/L, 0.6–1.3 mg/L and 1–2.3 mg/L, respectively. Generally, for non-bio P processes such as CAS and MLE, TP removal without chemical addition was close with and without FW addition. On the other hand, A2O and Bardenpho increased P removal with FW addition by 0.4–1.9 mg/L and 0.7–2.5 mg/L, respectively. Similarly, FW supplementation reduced net operational costs for a 37,854 m^3/d (10 MGD) plant by 26%–63% for a mainstream DO of 2 mg/L, 24%–78% for partial nitrification processes (DO of 0.5 mg/L), 29%–54% for sidestream with mainstream DO of 2 mg/L, and 23%–76% for sidestream with mainstream partial nitrification processes (DO of 0.5 mg/L). Total energy benefit considering both the net change in aeration energy and methane energy also increased by 3300–7200 kWh/d for CAS, MLE, A2O, and Bardenpho for mainstream DO 2 mg/L, 3500–7900 kWh/d for partial nitrification processes, 3300–7200 kWh/d for sidestream nitrogen removal with mainstream DO of 2 mg/L, and 3500–7300 kWh/d for sidestream nitrogen removal with mainstream shortcut nitrification. Carbon diversion scenarios using PTE of 45%, 65%, and 85% showed that net operational costs decreased as PTE increased, indicating a more pronounced beneficial FW impact on net operational costs with increasing carbon diversion efficiency. Relative to the MWW alone cases, the estimated net total energy gain for FW separate addition considering both the net change in aeration energy and methane energy was 3300–7100 kWh/d for a PTE of 45% and 3500–7400 kWh/d for a PTE of 85%, indicating an increasing energy gain with higher PTE.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2019.02.065.

References

Battistoni, P., Fatone, F., Passacantando, D., Bolzonella, D., 2007. Application of food waste disposers and alternate cycles process in small-decentralized towns: a case study. Water Res. 41, 893–903.
Bolzonella, D., Micolucci, F., Battista, F., Cavinato, C., Gottardo, M., Piovesan, S., Pavan, P., 2019. Producing Biohythane from Urban Organic Wastes. Waste and Biomass Valorization.
Bolzonella, D., Pavan, P., Battistoni, P., Cecchi, F., 2003. The under sink garbage grinder: a friendly technology for the environment. Environ. Technol. 24, 349–359.
Chowdhury, M.M.I., Kim, M., Haroun, B.M., Nakhla, G., Keleman, M., 2016. Flocculent settling of food wastes. Water Environ. Res. 88, 660–664.
Dai, X., Duan, N., Dong, B., Dai, L., 2013. High-solids anaerobic co-digestion of sewage sludge and food waste in comparison with mono digestions: stability and performance. Waste Manag. 33, 308–316.
Iacovidou, E., Ohandja, D.-G., Voulvoulis, N., 2012. Food waste co-digestion with sewage sludge – realising its potential in the UK. J. Environ. Manag. 112, 267–274.
Kim, M., Chowdhury, M.M.I., Nakhla, G., Keleman, M., 2015. Characterization of typical household food wastes from disposers: fractionation of constituents and implications for resource recovery at wastewater treatment. Bioresour. Technol. 183, 61–69.
Kim, M., Chowdhury, M.M.I., Nakhla, G., Keleman, M., 2017. Synergism of co-digestion of food wastes with municipal wastewater treatment biosolids. Waste Manag. 61, 473–483.
Koch, K., Plabst, M., Schmidt, A., Helmreich, B., Drewes, J.E., 2016. Co-digestion of food waste in a municipal wastewater treatment plant: comparison of batch tests and full-scale experiences. Waste Manag. 47, 28–33.
Leverenz, H., Tchobanoglous, G., 2013. Energy Balance and Nutrient Removal Impacts of Food Waste Disposers on Wastewater Treatment. Final Report to InSinkErator.
Nghiem, L.D., Koch, K., Bolzonella, D., Drewes, J.E., 2017. Full scale co-digestion of wastewater sludge and food waste: bottlenecks and possibilities. Renew. Sustain. Energy Rev. 72, 354–362.
Orhon, D., Çokgör, E.U., 1997. COD fractionation in wastewater characterization—the state of the art. J. Chem. Technol. Biotechnol. 68, 283–293.
Tang, J., Wang, X.C., Hu, Y., Ngo, H.H., Li, Y., Zhang, Y., 2017. Applying fermentation liquid of food waste as carbon source to a pilot-scale anoxic/oxic-membrane bioreactor for enhancing nitrogen removal: microbial communities and membrane fouling behaviour. Bioresour. Technol. 236, 164–173.
Tchobanoglous, G., Burton, F.L., Stensel, H.D., Metcalf, Eddy, I., Burton, F., 2003. Wastewater Engineering: Treatment and Reuse. McGraw-Hill Education.
Thomas, P., 2011. The effects of food waste disposers on the wastewater system: a practical study. Water Environ. J. 25, 250–256.
Xu, F., Li, Y., Ge, X., Yang, L., Li, Y., 2018. Anaerobic digestion of food waste – challenges and opportunities. Bioresour. Technol. 247, 1047–1058.
Yazdanpanah, A., Ghasimi, D.S.M., Kim, M.G., Nakhla, G., Hafez, H., Keleman, M., 2018. Impact of trace element supplementation on mesophilic anaerobic digestion of food waste using Fe-rich inoculum. Environ. Sci. Pollut. Control Ser. 25, 29240–29255.
Zhang, C., Su, H., Baeyens, J., Tan, T., 2014. Reviewing the anaerobic digestion of food waste for biogas production. Renew. Sustain. Energy Rev. 38, 383–392.
Zheng, X., Zhou, W., Wan, R., Luo, J., Su, Y., Huang, H., Chen, Y., 2018. Increasing municipal wastewater BNR by using the preferred carbon source derived from kitchen wastewater to enhance phosphorus uptake and short-cut nitrification-denitrification. Chem. Eng. J. 344, 556–564.

Modeling the Impact of Food Wastes on Wastewater Treatment Plants

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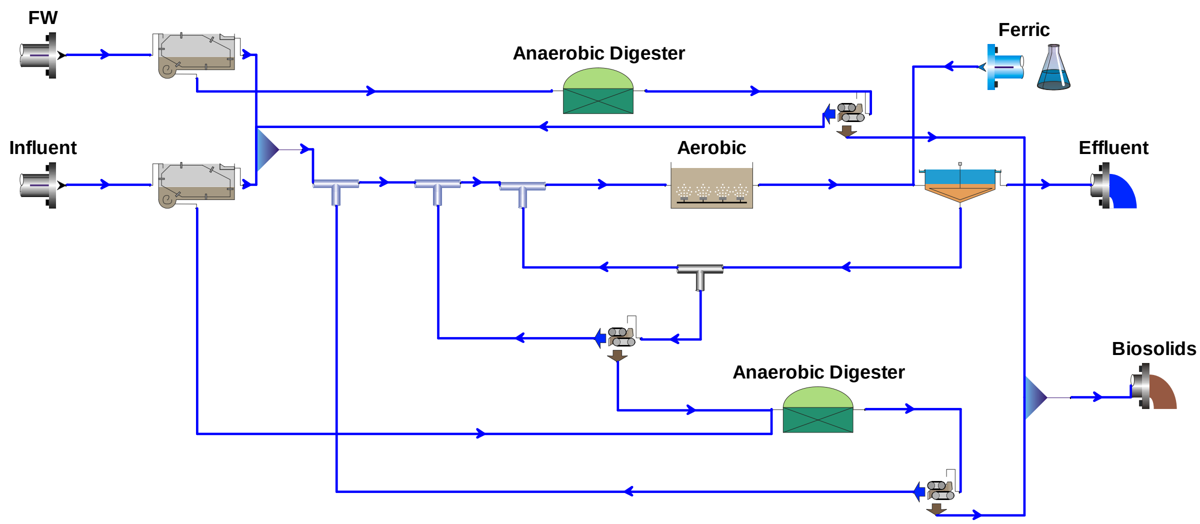
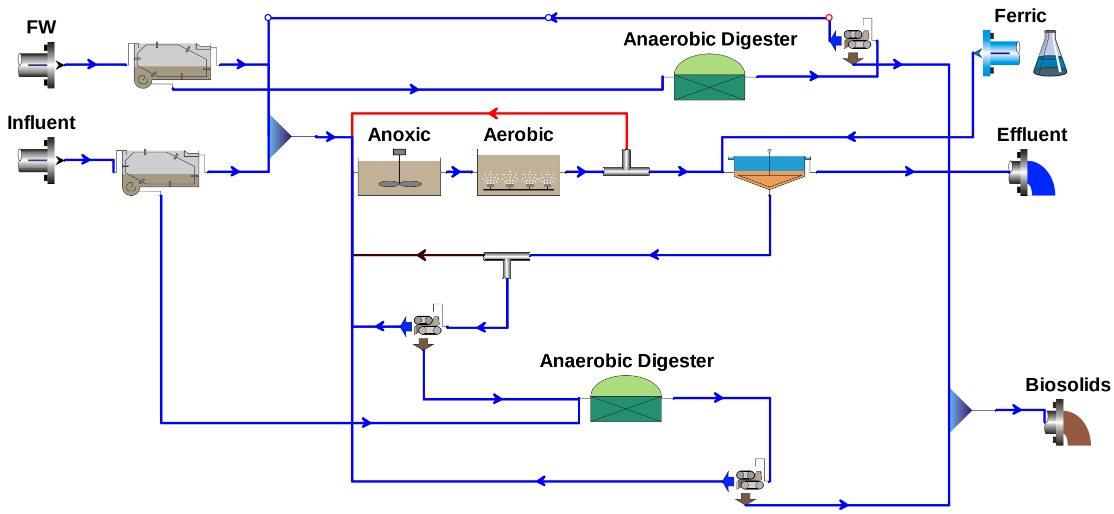
Supporting Information

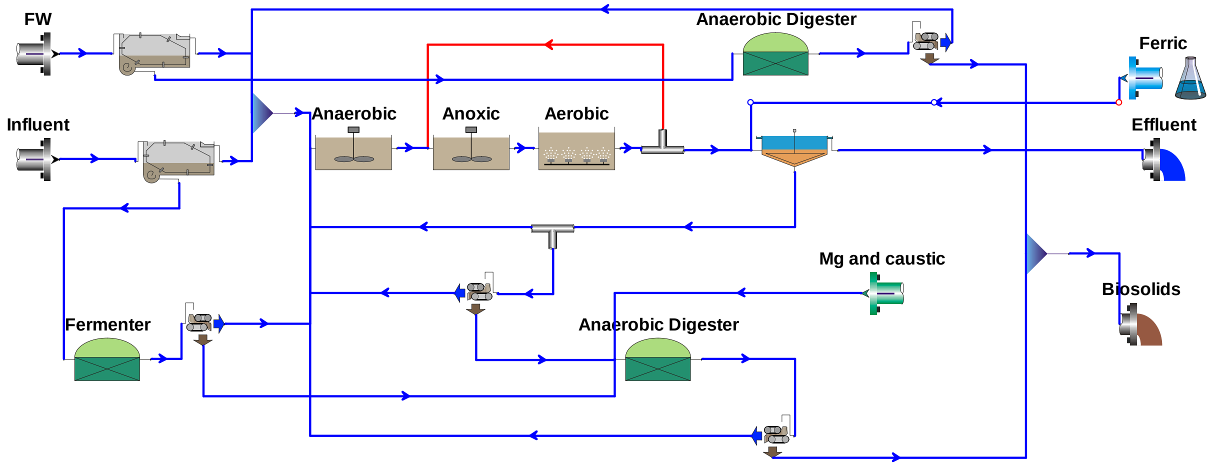
a. CAS + Separate FW + Chemical P



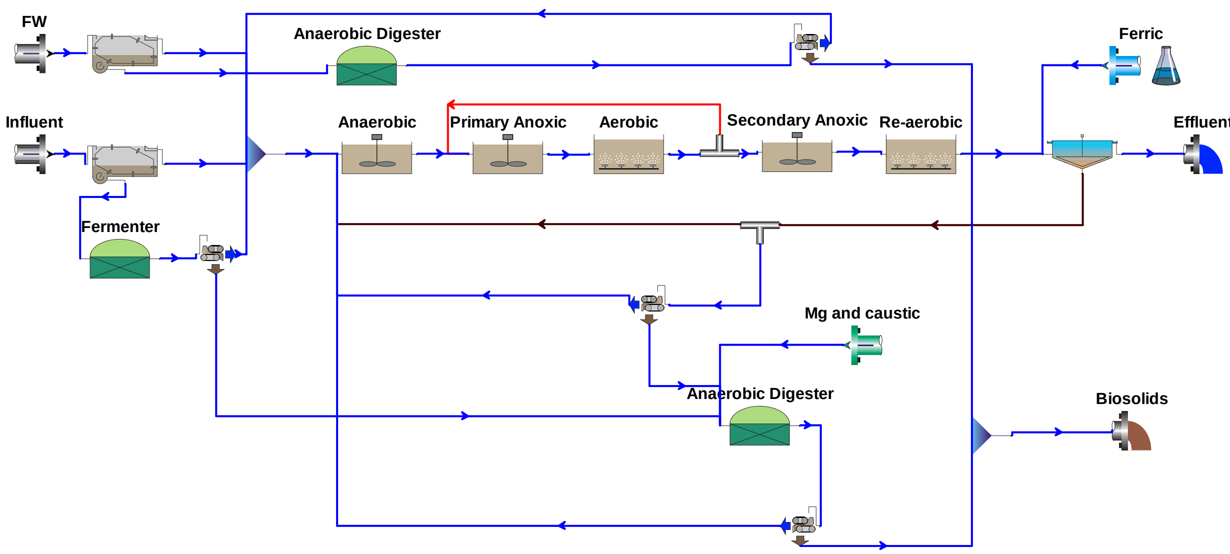
b. MLE + Separate FW + Chemical P



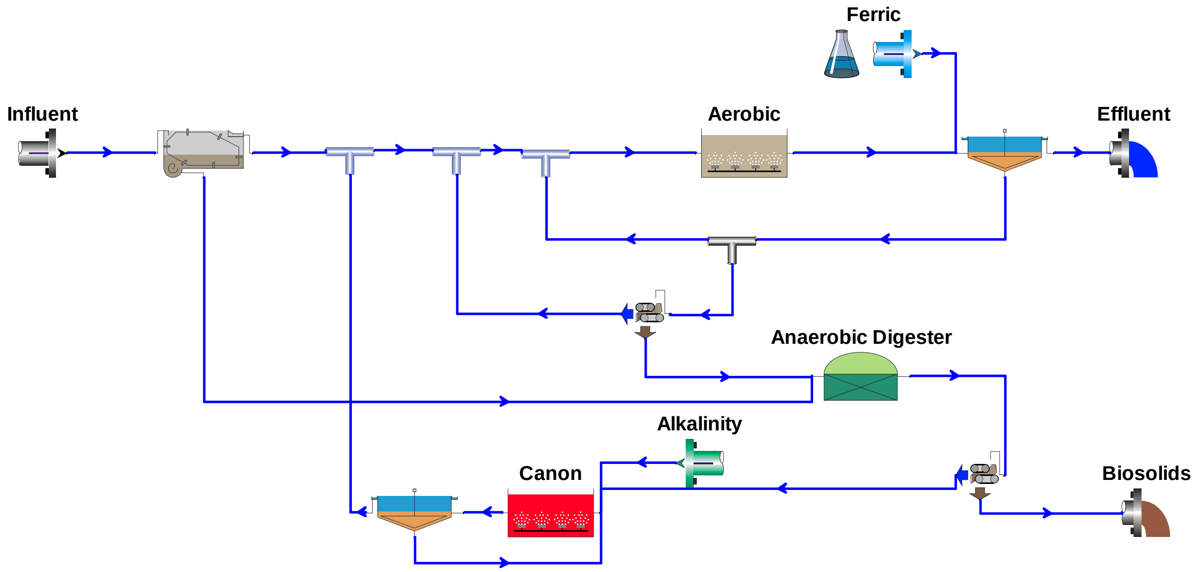
c. A2O + Separate FW + Chemical P



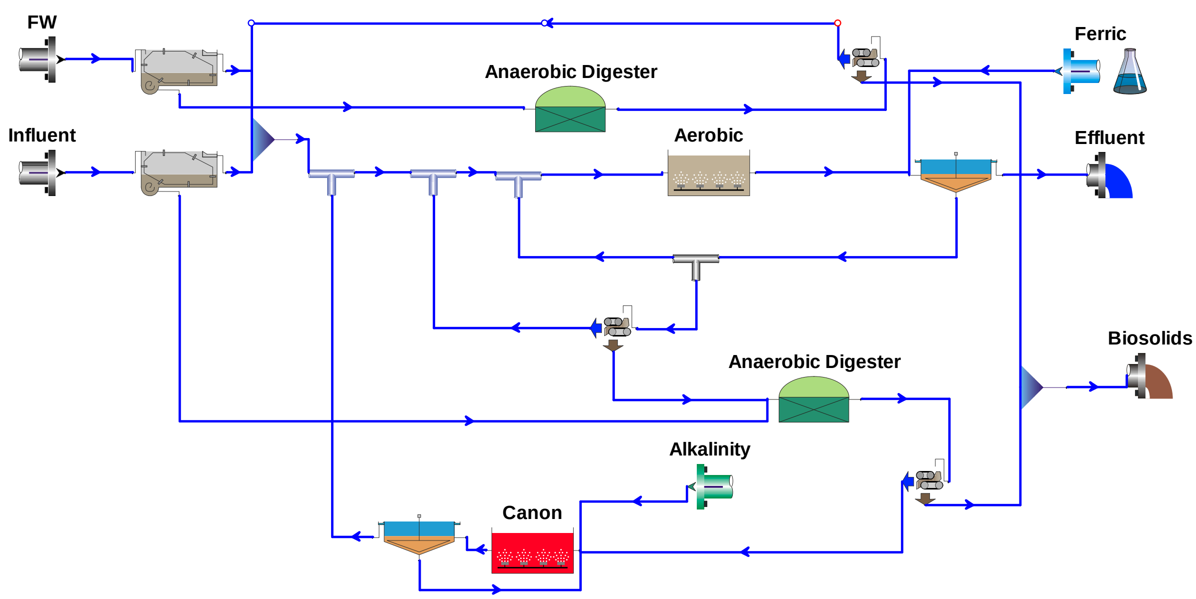
d. Bardenpho + Separate FW + Chemical P



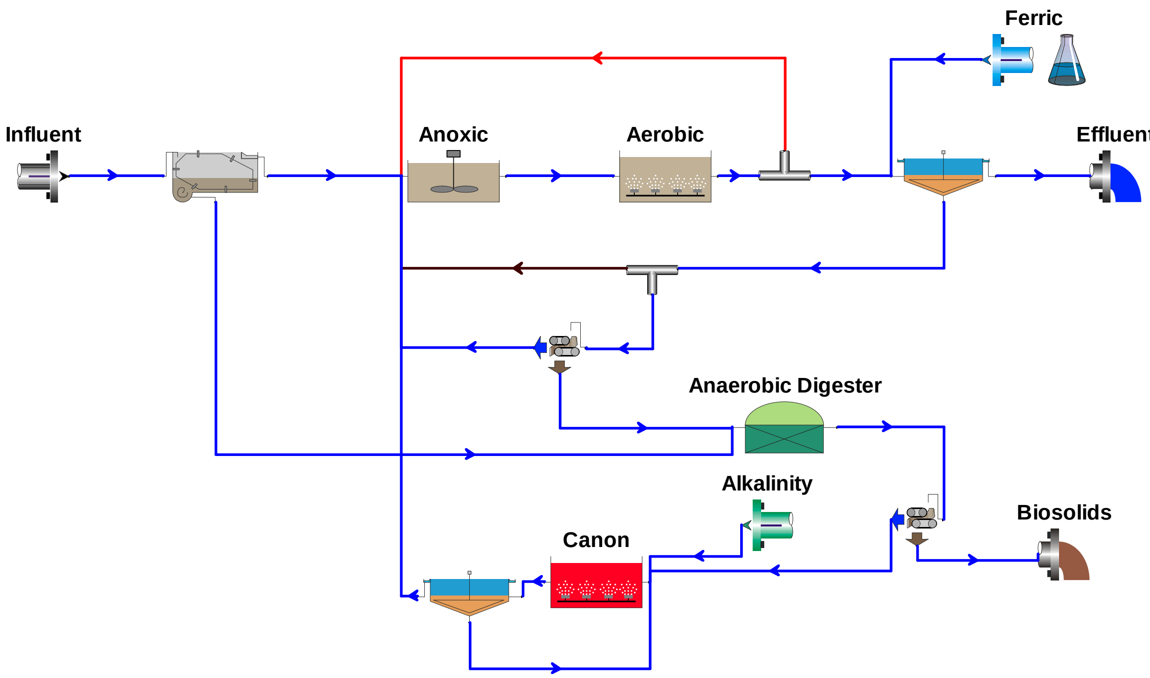
e. CAS + Chemical P + Canon



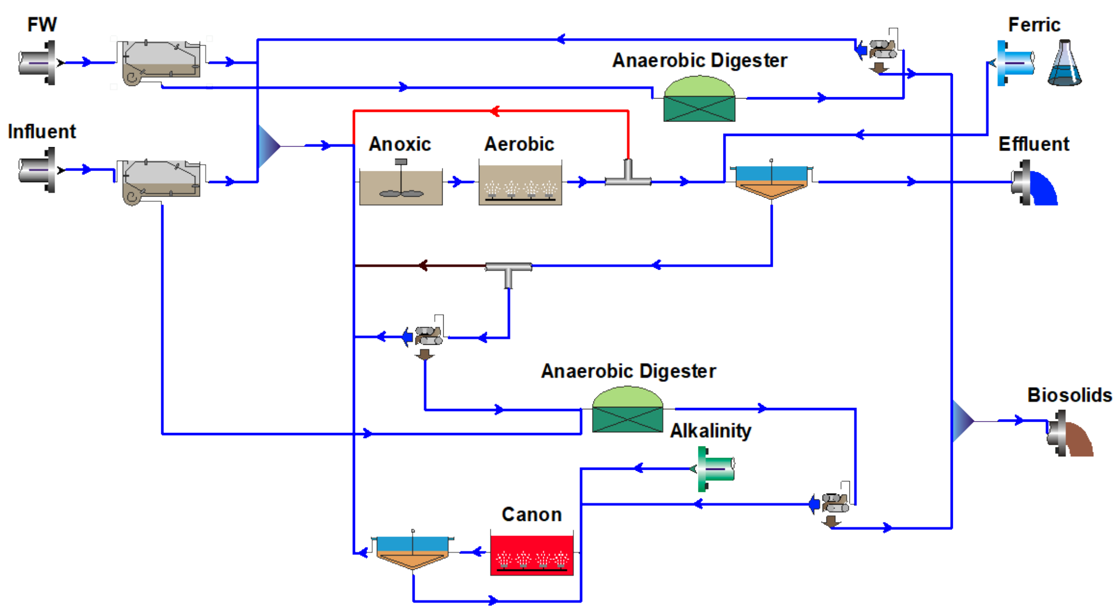
f. CAS + Separate FW + Chemical P + Canon



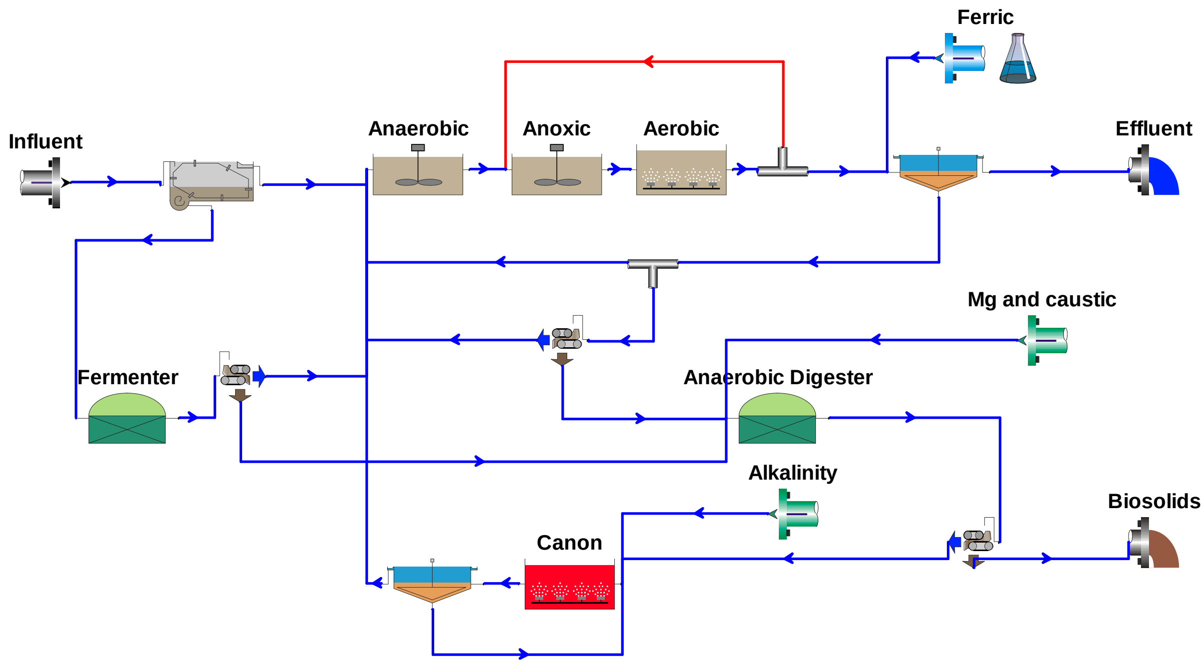
g. MLE + Chemical P + Canon



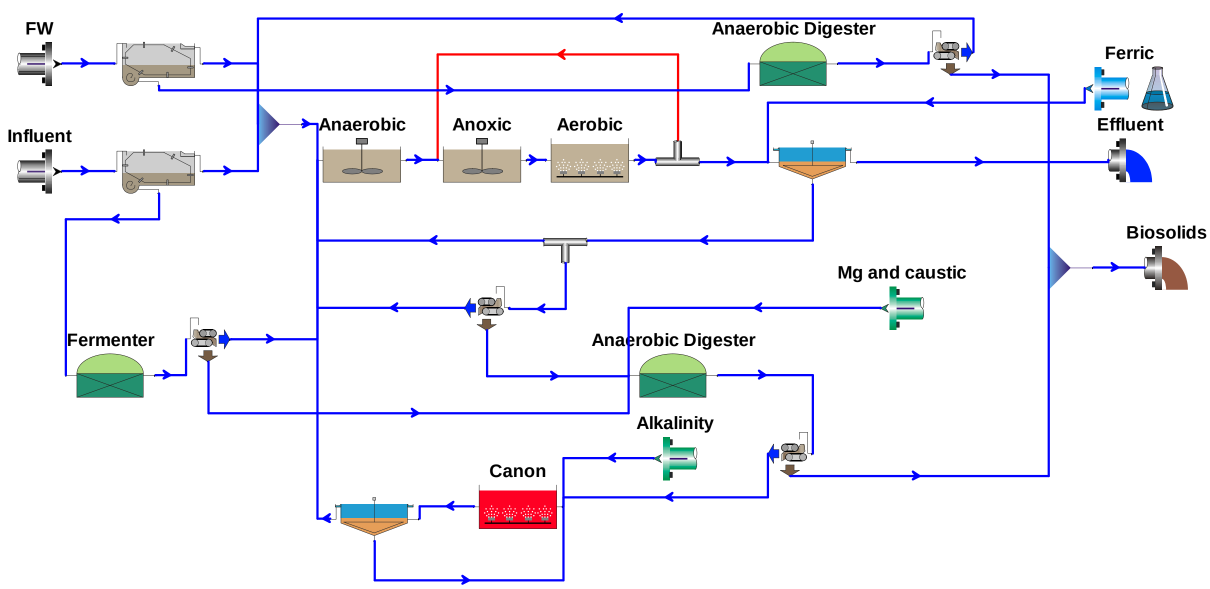
h. MLE + Separate FW + Chemical P + Canon



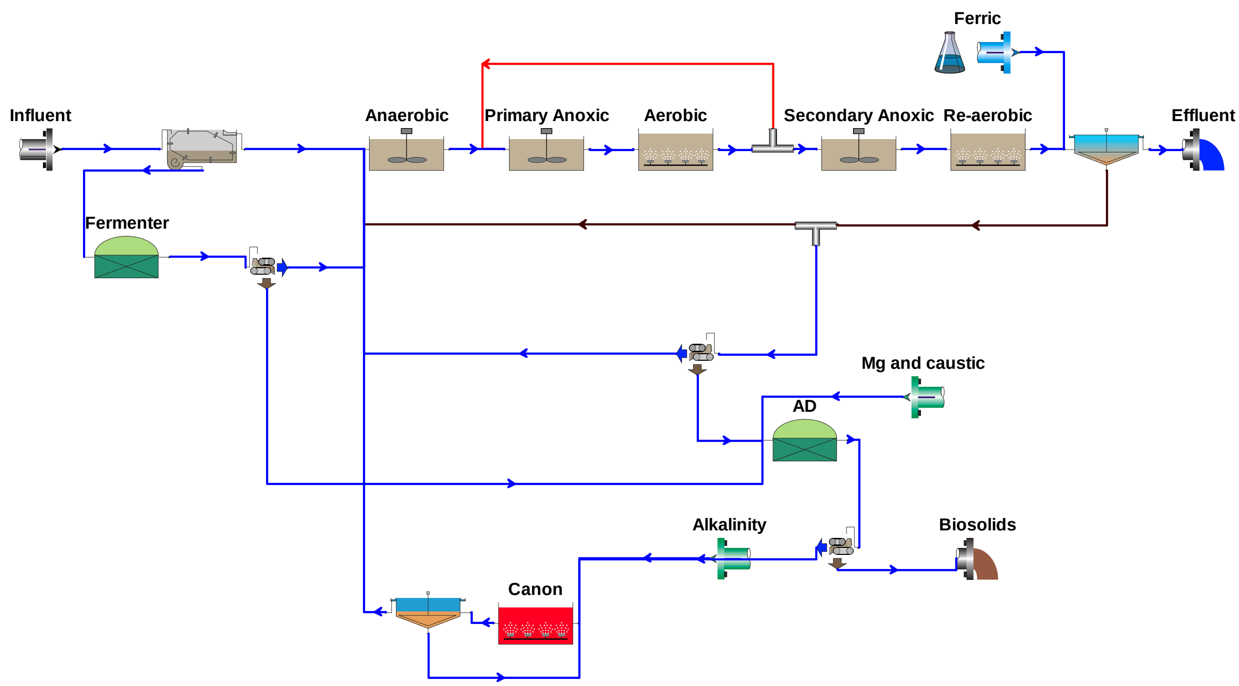
i. A2O + Chemical P + Canon



j. A2O + Separate FW + Chemical P + Canon



k. Bardenpho + Chemical P + Canon



l. Bardenpho + Separate FW + Chemical P + Canon

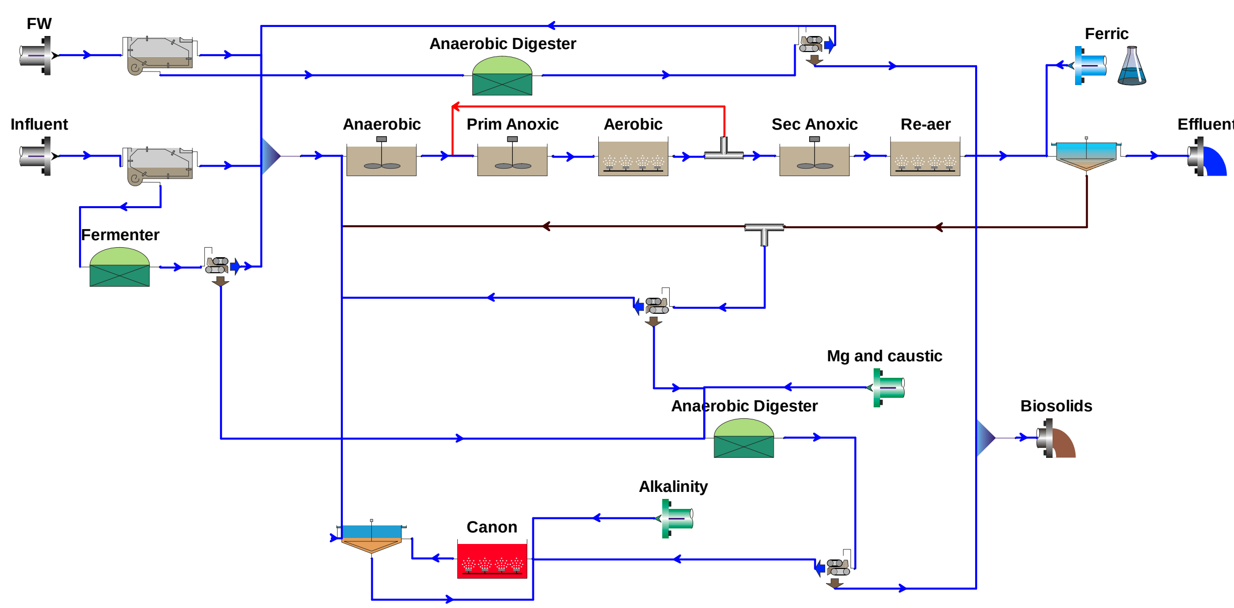
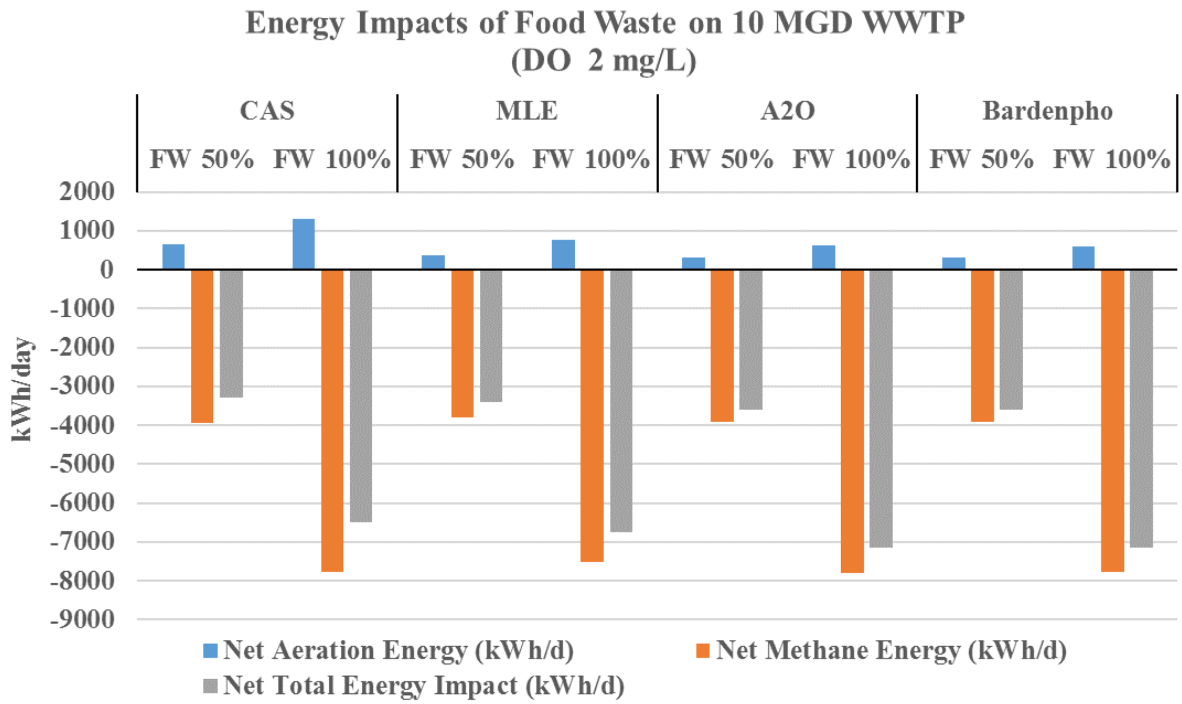


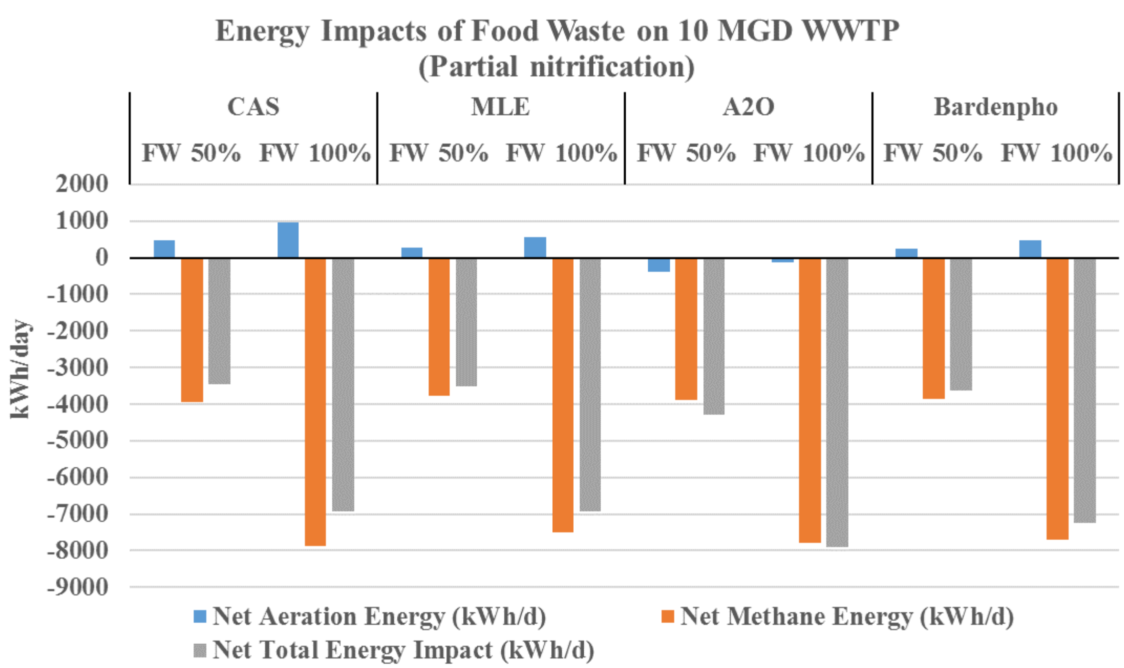
Figure A.1. Process Flow Diagrams

a.



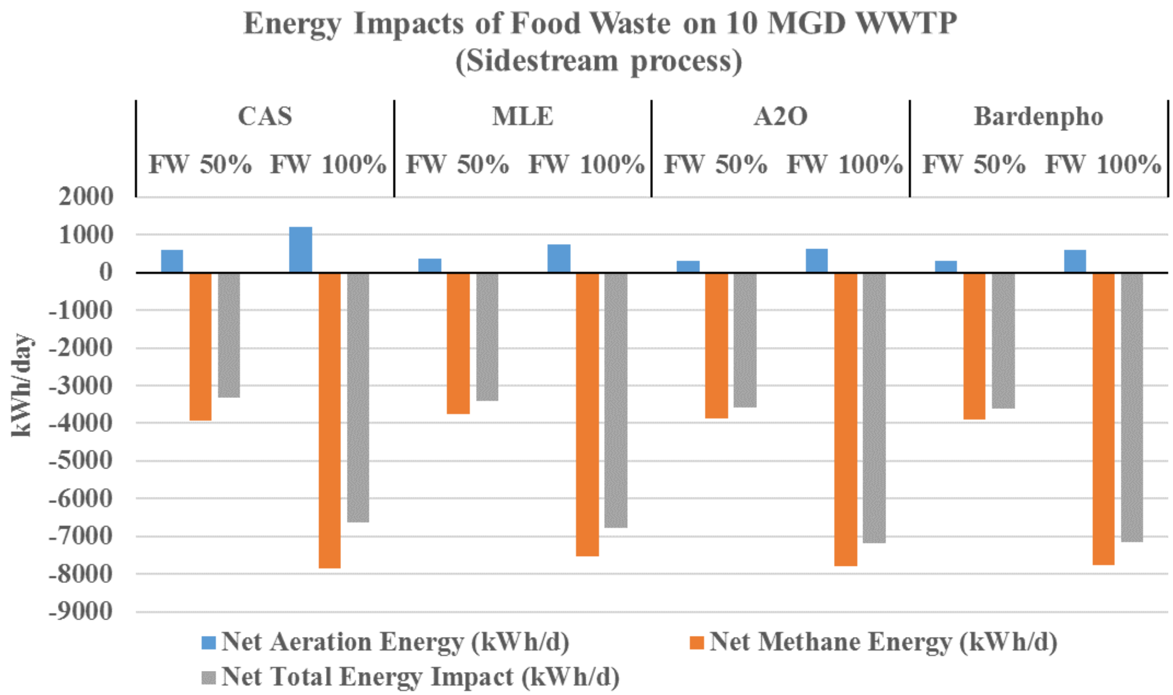
Energy Impacts of Food Waste on 10 MGD WWTP
(DO 2 mg/L)
CAS
MLE
A2O
Bardenpho
FW 50%
FW 100%
2000
1000
0
-1000
-2000
-3000
-4000
-5000
-6000
-7000
-8000
-9000
kWh/day
Net Aeration Energy (kWh/d)
Net Methane Energy (kWh/d)
Net Total Energy Impact (kWh/d)

b.



Energy Impacts of Food Waste on 10 MGD WWTP
(Partial nitrification)
CAS
MLE
A2O
Bardenpho
FW 50%
FW 100%
2000
1000
0
-1000
-2000
-3000
-4000
-5000
-6000
-7000
-8000
-9000
kWh/day
Net Aeration Energy (kWh/d)
Net Methane Energy (kWh/d)
Net Total Energy Impact (kWh/d)

c.



d.

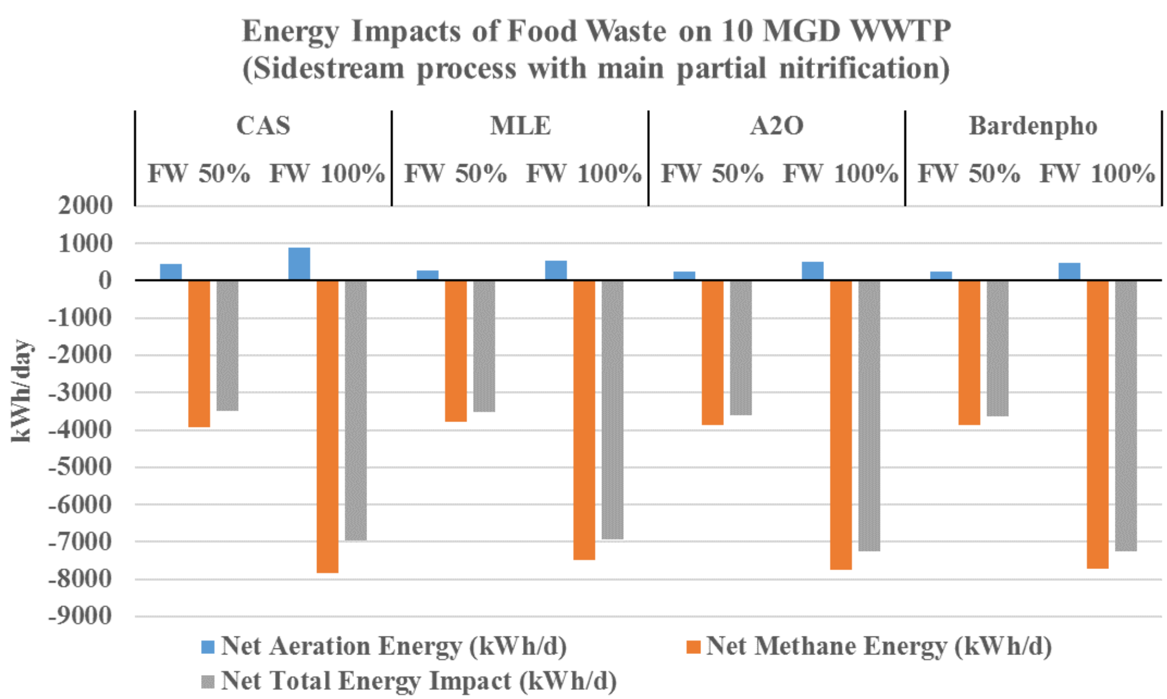


Figure A.2. Energy impacts of food waste on WWTP (37854 m3/d or 10 MGD)

Table A.1. Simulated systems and HRT (Tchobanoglous et al., 2003)

Process type	Bioreactor HRT (hrs)	SRT (days)	Recycle rate (%)	Internal recycle rate (%)
CAS	Aerobic (12)	9.8	100	NA
MLE	Anoxic (3.8) Aerobic (14)	16	100	500
A2O	Anaerobic (4.8) Anoxic (3.6) Aerobic (7.2)	13	100	300
Bardenpho	Anaerobic (1.2) Anoxic1 (3.6) Aerobic1 (7.2) Anoxic2 (3.6) Aerobic2 (2.4)	13	100	300

Table A.2. FW characterization

COD fractionation	Value	Literature
Total influent COD, C_T (= $S_T + X_T$)	1	(Orhon and Çokgör, 1997)
Soluble influent COD, S_T (= $S_S + S_H + S_I$)	0.43	
Readily biodegradable COD, S_S	0.41	
Influent rapidly hydrolysable COD, S_H		
Influent soluble inert COD, S_I	0.02	
Total influent particulate COD, X_T (=$X_S + X_H + X_I$)	0.57	
Influent slowly biodegradable COD, X_S	0.51	
Active heterotrophic biomass, X_H	0	
Influent particulate inert COD, X_I	0.06	
Relationship between wastewater parameters	**Value**	**Literature**
VS/TS	0.88	(Leverenz and Tchobanoglous, 2013; Yazdanpanah et al., 2018; Zahan et al., 2016)
VSS/TSS	0.95	(Chowdhury et al., 2016; Kim et al., 2015, 2017; Leverenz and Tchobanoglous, 2013; Yazdanpanah et al., 2018)
TSS/TS	0.64	(Leverenz and Tchobanoglous, 2013; Yazdanpanah et al., 2018)
TCOD/TS	1.5	(Leverenz and Tchobanoglous, 2013; Yazdanpanah et al., 2018; Zahan et al., 2016)
TCOD/TKN	51.5	(Chowdhury et al., 2016; Kim et al., 2015, 2017; Leverenz and Tchobanoglous, 2013; Yazdanpanah et al., 2018)
TCOD/TP	270	(Kim et al., 2015, 2017; Leverenz and Tchobanoglous, 2013; Thomas, 2011; Yazdanpanah et al., 2018)
TCOD/TBOD	1.72	(Leverenz and Tchobanoglous, 2013; Thomas, 2011)
NH4/TKN	0.03	(Zahan et al., 2016)
VFA/TCOD	0.024	(Kim et al., 2015, 2017)
SCOD/TCOD	0.43	(Chowdhury et al., 2016; Kim et al., 2015, 2017)

Table A.3a. Detailed characterization of different wastewaters (mg/L)

mg/L	MWW (Flowrate 37854 m^3/d or 10 MGD)	FW (Flowrate 189 m^3/d or 0.05 MGD for 50% penetration and 378.5 m^3/d or 0.1 MGD for 100% penetration)
TCOD	438	12700
SCOD	173	5422
PCOD	265	7278
S_I	28	300
X_I	57	803
TBOD	213	7055
SBOD	103	3623
PBOD	110	3432
rbCOD	60	5061
VFA	12	302
TSS	198	4804
VSS	166	4570
TKN	43	246
SON	5.5	119
$STKN_I$	0.9	4.9
PON	5.5	119
NH_4	32	7.6
TP	8.3	47.1
SP	4.1	0.0

TCOD – total chemical oxygen demand, SCOD – soluble COD, PCOD – particulate COD, rbCOD – readily biodegradable COD, S_I-soluble inert COD, X_I – particulate inert COD, TBOD – total biochemical oxygen demand, SBOD – soluble biochemical oxygen demand, PBOD – particulate BOD, VFA – volatile fatty acids, TSS – total suspended solids, VSS – volatile suspended solids, TKN – total kjeldahl nitrogen, SON – soluble organic nitrogen, $STKN_I$-soluble inert TKN, PON – particulate organic nitrogen, TP-total phosphorus, SP-soluble phosphorus,

Table A.3b. Fractionation of different wastewater

	MWW	FW separate
Fbs - Readily biodegradable (including Acetate) [gCOD/g of total COD]	0.136	0.398
Fac - Acetate [gCOD/g of readily biodegradable COD]	0.202	0.06
Fxsp - Non-colloidal slowly biodegradable [gCOD/g of slowly degradable COD]	0.7	0.99
Fus - Unbiodegradable soluble [gCOD/g of total COD]	0.065	0.024
Fup - Unbiodegradable particulate [gCOD/g of total COD]	0.13	0.063
Fna - Ammonia [gNH3-N/gTKN]	0.744	0.031
Fnox - Particulate organic nitrogen [gN/g Organic N]	0.5	0.5
Fnus - Soluble unbiodegradable TKN [gN/gTKN]	0.02	0.02
FupN - N:COD ratio for unbiodegradable part. COD [gN/gCOD]	0.035	0.035
Fpo4 - Phosphate [gPO4-P/gTP]	0.497	0
FupP - P:COD ratio for unbiodegradable part. COD [gP/gCOD]	0.011	0.011
FZbh - OHO COD fraction [gCOD/g of total COD]	0.02	0.02
FZbm - Methylotroph COD fraction [gCOD/g of total COD]	1.00E-04	1.00E-04
FZaob - AOB COD fraction [gCOD/g of total COD]	1.00E-04	1.00E-04
FZnob - NOB COD fraction [gCOD/g of total COD]	1.00E-04	1.00E-04
FZaao - AAO COD fraction [gCOD/g of total COD]	1.00E-04	1.00E-04
FZbp - PAO COD fraction [gCOD/g of total COD]	1.00E-04	1.00E-04
FZbpa - Propionic acetogens COD fraction [gCOD/g of total COD]	1.00E-04	1.00E-04
FZbam - Acetoclastic methanogens COD fraction [gCOD/g of total COD]	1.00E-04	1.00E-04
FZbhm - H2-utilizing methanogens COD fraction [gCOD/g of total COD]	1.00E-04	1.00E-04
FZe - Endogenous products COD fraction [gCOD/g of total COD]	0	0

Table A.4. Unit costs for cost analysis

Parameter	Unit cost (US$)
Ferric for chemical P removal	$2.29/kgFe [1]
Chemical addition for P removal of the anaerobic digesters for BNR systems	$0.288/L of chemical [2]
Methane energy	$10/GJ [3] (35.8MJ/$m^3$ methane)
Glycerol required for denitrification	$0.44/kg [4]
Polymer costs for sludge dewatering	$3.5/kg polymer and 5 kg polymer per 1000 kg dry solids [5]
Sludge disposal cost	$100/dry ton solids [6]
Electricity	$0.1/kwh [7]

Sources

[1] https://assets.lawrenceks.org/assets/agendas/cc/2011/12-06-11/UT_Chemical_Unit_Price_Comparison_2009-2012.pdf

[2] BioWin value

[3] Canadian Biogas Association, Renewed Gas Financial Tool, March 2017

[4] Environmental operating solutions – personal communication

[5] https://www.mi-wea.org/docs/Session%206%20-%20Polymers.pdf

[6] Communication with Lystek International.

[7] https://www.eia.gov/electricity/monthly/epm_table_grapher.php?t=epmt_5_6_a

Table A.5. Relative change of different operational costs for the different processes compared with the typical operation at DO 2 mg/L.

	Partial nitrification											
	CAS			**MLE**			**A2O**			**Bardenpho**		
Cost change (%)	**FW0%**	**FW50%**	**FW100%**	**FW0%**	**FW50%**	**FW100%**	**FW0%**	**FW50%**	**FW100%**	**FW0%**	**FW50%**	**FW100%**
Biosolids Disposal	0.2	0.2	0.0	0.0	0.0	0.0	-0.4	0.6	0.6	-0.6	-0.4	-0.3
Chemical Cost for dewatering	0.2	0.2	0.0	0.0	0.0	0.0	-0.4	0.6	0.6	-0.6	-0.4	-0.3
Chemical Cost for P removal	0.3	0.0	-0.4	0.0	0.0	0.0	-57	-56	-50	-47	-60	-71
Chemical for P precipitation in AD							0.0	0.0	0.0	0.0	0.0	0.0
Aeration blower cost	-25	-25	-25	-28	-28	-28	-37	-36	-35	-38	-37	-36
Methane energy	0	0	0	0	0	0	1	0	0	1	0	0
Supplemental carbon to achieve same effluent TN (glycerol)					9	9		-49	-50		-47	-51

	Sidestream											
	CAS			**MLE**			**A2O**			**Bardenpho**		
Cost change (%)	**FW0%**	**FW50%**	**FW100%**	**FW0%**	**FW50%**	**FW100%**	**FW0%**	**FW50%**	**FW100%**	**FW0%**	**FW50%**	**FW100%**
Biosolids Disposal	0.1	0.0	-0.2	-0.1	0.0	-0.1	-0.1	0.0	0.0	-0.1	0.1	0.1
Chemical Cost for dewatering	0.1	0.0	-0.2	-0.1	0.0	-0.1	-0.1	0.0	0.0	-0.1	0.1	0.1
Chemical Cost for P removal	-0.8	-0.6	-0.9	0	0	0	-6	-7	-6	-7	-4	4
Chemical for P precipitation in AD							0	0	0	0	0	0
Aeration blower cost	-5	-5	-5	-4	-4	-4	-3	-3	-3	-3	-3	-3
Methane energy	0	0	0	0	0	0	0	0	0	0	0	0
Supplemental carbon to achieve same effluent TN (glycerol)					6	5		1	3		-3	-4

	Partial nitrification + Sidestream											
	CAS			**MLE**			**A2O**			**Bardenpho**		
Cost change (%)	**FW0%**	**FW50%**	**FW100%**	**FW0%**	**FW50%**	**FW100%**	**FW0%**	**FW50%**	**FW100%**	**FW0%**	**FW50%**	**FW100%**
Biosolids Disposal	0.1	0.0	-0.1	0.0	0.0	-0.1	-0.4	-0.2	-0.1	-0.6	-0.4	-0.3
Chemical Cost for dewatering	0.1	0.0	-0.1	0.0	0.0	-0.1	-0.4	-0.2	-0.1	-0.6	-0.4	-0.3
Chemical Cost for P removal	-0.8	-0.6	-0.9	0.0	0.0	0.0	-58	-58	-51	-47	-50	-67
Chemical for P precipitation in AD							0	0	0	0	0	0
Aeration blower cost	-28	-28	-29	-30	-30	-31	-38	-37	-36	-39	-38	-37
Methane energy	0	0	0	0	0	0	1	0	0	1	0	0
Supplemental carbon to achieve same effluent TN (glycerol)					14	9		-46	-47		-51	-55

References

Chowdhury, M.M.I., Kim, M., Haroun, B.M., Nakhla, G., Keleman, M., 2016. Flocculent Settling of Food Wastes. Water Environment Research 88, 660-664.

Kim, M., Chowdhury, M.M.I., Nakhla, G., Keleman, M., 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.M.I., Nakhla, G., Keleman, M., 2017. Synergism of co-digestion of food wastes with municipal wastewater treatment biosolids. Waste Management 61, 473-483.

Leverenz, H., Tchobanoglous, G., 2013. Energy Balance and Nutrient Removal Impacts of Food Waste Disposers on Wastewater Treatment. Final report to InSinkErator.

Orhon, D., Çokgör, E.U., 1997. COD Fractionation in Wastewater Characterization—The State of the Art. Journal of Chemical Technology & Biotechnology 68, 283-293.

Tchobanoglous, G., Burton, F.L., Stensel, H.D., Metcalf, Eddy, I., Burton, F., 2003. Wastewater Engineering: Treatment and Reuse. McGraw-Hill Education.

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

Yazdanpanah, A., Ghasimi, D.S.M., Kim, M.G., Nakhla, G., Hafez, H., Keleman, M., 2018. Impact of trace element supplementation on mesophilic anaerobic digestion of food waste using Fe-rich inoculum. Environmental Science and Pollution Research 25, 29240-29255.

Zahan, Z., Othman, M.Z., Rajendram, W., 2016. Anaerobic Codigestion of Municipal Wastewater Treatment Plant Sludge with Food Waste: A Case Study. BioMed Research International 2016, 13.