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A review on recent disposal of hazardous sewage sludge via anaerobic digestion and novel composting

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A R T I C L E I N F O

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A B S T R A C T

The high investment cost required by modern treatment technologies of hazardous sewage sludge such as incineration and anaerobic digestion have discouraged their application by many developing countries. Hence, this review elucidates the status, performances and limitations of two low-cost methods for biological treatment of hazardous sewage sludge, employing vermicomposting and black soldier fly larvae (BSFL). Their performances in terms of carbon recovery, nitrogen recovery, mass reduction, pathogen destruction and heavy metal stabilization were assessed alongside with the mature anaerobic digestion method. It was revealed that vermicomposting and BSFL were on par with anaerobic digestion for carbon recovery, nitrogen recovery and mass reduction. Thermophilic anaerobic digestion was found superior in pathogen destruction because of its high operational temperature. Anaerobic digestion also had proven its ability to stabilize heavy metals, but no conclusive finding could confirm similar application from vermicomposting or BSFL treatment. However, the addition of co-substrates or biochar during vermicomposting or BSFL treatment may show synergistic effects in stabilizing heavy metals as demonstrated by anaerobic digestion. Moreover, vermicomposting and BSFL valorization had manifested their potentialities as the low-cost alternatives for treating hazardous sewage sludge, whilst producing value-added feedstock for biochemical industries.

1. Introduction

In general, conventional wastewater treatment plant produces two types of sewage sludge – primary and secondary sludge. Primary sludge is the solid accumulated from the physical separation process and consists of gravitational precipitates. Activated sludge, also known as secondary sludge, is the byproduct generated from the biological treatment

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plant, containing high amount of active microbes (Liew et al., 2021). In raw form, both of them are highly inhabited by pathogens and bacteria such as *E. coli*, *Salmonella* spp. and *Enterococcus* (Al-Gheethi et al., 2018).

Nomenclature

BSF	Black soldier fly
BSFL	Black soldier fly larvae
BOD	Biochemical oxygen demand
COD	Chemical oxygen demand
EPS	Extracellular polymeric substances
FAME	Fatty acid methyl ester
MEC	Microbial electrolysis cell
MPN	Most probable number
sCOD	Soluble chemical oxygen demand
SRT	Sludge residence time
TOC	Total organic carbon
TS	Total solids
VFA	Volatile fatty acids
VS	Volatile solid

The raw sludge is also loaded with high organic matter and heavy metals as shown in Table 1. The direct application of sewage sludge into soil could potentially cause leaching of organic matter and heavy metals into the soil and eventually seep into the groundwater. If it is left untreated, the sludge will undergo anaerobic digestion naturally and generate an unpleasant smell due to the release of volatile organic and inorganic compounds such as NH_3 and H_2S, being the most dominant odor gases (Zhu et al., 2016).

To overcome these hazards, numerous sludge treatment technologies have been introduced. They can be classified into three main categories, namely, chemical treatment, thermal treatment and biological treatment. Generally, the performances of each sludge treatment method can be assessed based on five criteria: (i) carbon recovery, (ii) nitrogen recovery, (iii) volume/weight reduction, (iv) pathogen destruction and (v) heavy metal stabilization. The advantages and limitations for each treatment are summarized as shown in Table 2. Chemical treatment is a simple treatment method that only requires pH monitoring after chemical addition. However, this method increases the dry weight of residue sludge due the additional weight contributed by the added chemicals. With chemical treatment alone, it is also impossible to recover the carbon in the form of biofuel or biomass. Thermal treatments such as incineration, pyrolysis, gasification, and supercritical water oxidation have higher weight reduction ability and are able to recover the carbon for conversion into biofuels (Liew et al., 2021; Ge et al., 2020; Lam et al., 2019). Meanwhile, biological treatment employs living organisms such as bacteria, worms, and insect larvae to accelerate the decomposition of organic matters in sewage sludge. It requires less energy to operate and could also recover carbon in different ways. It also has the ability to retain the nitrogen in the treated sludge as compared with thermal treatment where most of the nitrogen would be volatilized and removed.

Among the many technologies, incineration and anaerobic digestion are the most matured method that are exploited. The adoption rate of these technologies is still very low and it is only concentrated in developed countries such as Germany, Netherland, United States and China. This could be attributed to the high capital cost involved. For example, an incineration plant with a processing capacity of 200,000 ton/year requires a capital cost of around €122 million (Hogg, 2017). Owing to this reason, the implementation of these modern treatment methods is still very much lagging in developing countries. Therefore, to promote a wider adoption of more sustainable sewage sludge treatment technologies, the focus should be allocated on researching and introducing a low-cost sludge treatment method that can be affordably adopted globally.

Currently, there are two potentially low-cost biological treatment methods, namely vermicomposting and black soldier fly larvae (BSFL) that could be further explored to tackle the issue of sludge management. The employment could eliminate the need of installing high-temperature equipment as required by thermal treatment, thereby reducing the process complexity and capital cost (Kacprzak et al., 2017; Mayer et al., 2020). Vermicomposting has demonstrated the ability to decompose the sewage sludge three times faster than that naturally being decomposed during the disposal (Domínguez et al., 2000). In New Zealand, a large-scale application of vermicomposting to treat sewage sludge and co-substrate has been successfully established (Quintern, 2014). Meanwhile, the BSFL assimilate the organic wastes such as food wastes and animal manure, and convert the waste nutrition into insect biomass, containing high lipid and protein contents (Gold et al., 2018). Therefore, the BSFL treatment has also been deployed industrially to produce high protein poultry feed through feeding with organic wastes.

This review elucidates the current status, limitations and area of improvements for employing the biological treatments to reduce and convert sewage sludge. The performances of biological treatments, namely, anaerobic digestion, vermicomposting and BSFL valorization, are assessed with respect to carbon recovery, nitrogen recovery, mass reduction and sludge stabilization. Comparisons are made between the two low-cost solutions and the more matured anaerobic digestion method. This provides useful information to evaluate the feasibility of applying vermicomposting or BSFL treatment as a low-cost alternative to support a more sustainable sludge management process.

Table 1
Characteristics of raw sewage sludge.

Parameter	Value	Reference
pH	7.2 ± 0.7	(Metcalf et al., 2013; Gupta and Garg, 2008; Ruiz-Espinoza et al., 2012; Awasthi et al., 2020a; Koupaie et al., 2018; Babel et al., 2009; Khwairakpam and Bhargava, 2009)
Total dry solids (% of TS)	4.4 ± 3.2	(Metcalf et al., 2013; Gupta and Garg, 2008; Ruiz-Espinoza et al., 2012; Koupaie et al., 2018; Babel et al., 2009)
Volatile solids (% of TS)	58.4 ± 19	(Metcalf et al., 2013; Gupta and Garg, 2008; Ruiz-Espinoza et al., 2012; Koupaie et al., 2018; Babel et al., 2009)
TOC (% of TS)	31.7 ± 10.2	(Awasthi et al., 2020a; Babel et al., 2009; Khwairakpam and Bhargava, 2009; Hait and Tare, 2011)
Total Nitrogen (% of TS)	1.93 ± 0.81	(Metcalf et al., 2013; Awasthi et al., 2020a; Babel et al., 2009; Khwairakpam and Bhargava, 2009; Hait and Tare, 2011)
Total Phosphorus (% of TS)	1.66 ± 1.57	(Metcalf et al., 2013; Gupta and Garg, 2008; Babel et al., 2009; Khwairakpam and Bhargava, 2009)
Lead (mg/kg)	102 ± 138	(Babel et al., 2009; Tomasi Morgano et al., 2018; Liu et al., 2018; Seggiani et al., 2012; Abu Bakar et al., 2008)
Zinc (mg/kg)	1400 ± 986	(Babel et al., 2009; Tomasi Morgano et al., 2018; Liu et al., 2018; Seggiani et al., 2012; Abu Bakar et al., 2008)
Chromium (mg/kg)	225 ± 118	(Babel et al., 2009; Tomasi Morgano et al., 2018; Liu et al., 2018; Seggiani et al., 2012)
Copper (mg/kg)	299 ± 328	(Tomasi Morgano et al., 2018; Liu et al., 2018; Seggiani et al., 2012; Abu Bakar et al., 2008)
Nickel (mg/kg)	94 ± 75	(Babel et al., 2009; Tomasi Morgano et al., 2018; Liu et al., 2018; Seggiani et al., 2012; Abu Bakar et al., 2008)
Fecal coliform (MPN/g)	$1.33 \times 10^7 \pm 1.31 \times 10^7$	(Hait and Tare, 2011; Scaglia et al., 2014; Bina et al., 2004; Liu et al., 2017)
Salmonella (MPN/g)	495 ± 854	(Hait and Tare, 2011; Scaglia et al., 2014; Bina et al., 2004)

Table 2
Comparison among sewage sludge treatment methods (Liew et al., 2021; Bina et al., 2004; Kacprzak et al., 2017; Mayer et al., 2020; Schnell et al., 2020; Sanger et al., 2001; Teoh and Li, 2020; Massé et al., 2007).

Treatment mode	Carbon Recovery	Nitrogen Recovery	Mass Reduction	Pathogen Destruction	Heavy Metal Stabilization	Advantage	Limitation
Chemical Treatment	–	+	–	+	+	- Easiest mode of operation. - Inhibition of pathogens at pH 12. - High nitrogen retention in treated sludge	- Increasing weight of sludge residue. - High operating cost due to the use of excessive chemicals.
Thermal Treatment	+	–	+	+	+	- Carbon recovery through generation of biogas, bio-oil or biochar. - Highest volume/weight reduction. - Short treatment time. - Complete destruction of pathogens.	- High investment cost and complex technology. - High energy requirement and consumption. - Emission of pollutants, requiring air pollution control unit. - Some methods have not been proven in mass-scale. - Low nitrogen retention due to the volatilization at high temperature.
Biological Treatment	+	+	+	V	V	- Carbon recovery ability varies for different type of biological treatments. ▪ Anaerobic Digestion: Biogas ▪ Vermicomposting: Worms ▪ BSFL: Biodiesel/Larvae - High nitrogen retention in solid residue.	- Long treatment duration. - Anaerobic digestion can stabilize heavy metal and pathogens, but inconclusive finding was reported for vermicomposting and BSFL. - Anaerobic digestion requires high investment cost and has a complex operation.

+ : Effective; -: Ineffective; V: Varies

2. Anaerobic digestion

2.1. Overview

Anaerobic digestion is a biological process that uses microbes to convert biodegradable organic matters into biogas in an oxygen-free condition (Cao and Pawłowski, 2012). It has been extensively used as a treatment method for both organic wastes and sewage sludge. The wastewater treatment plant at Italy, for instance, uses anaerobic digestion to process around 380 m^3/d of sewage sludge, while generating about 6100 m^3/d of biogas (Rittmann et al., 2008). In general, anaerobic digestion system requires a lower capital cost as compared with thermal treatment plant such as incinerator, which could cost at least 3 times more investment than anaerobic digestion (Mayer et al., 2020). However, the reaction time needed for anaerobic digestion is significantly longer as opposed to other non-biological methods.

Anaerobic digestion consists of four successive biochemical processes, namely, hydrolysis, acidogenesis, acetogenesis and methanogenesis. The first step is hydrolysis by which large complex organic polymers such as polysaccharides, proteins, starches, free oil and grease are hydrolyzed into simpler soluble substituents such as amino acids, long chain fatty acids and simple sugars. The bacteria species involved in such hydrolysis include *Clostridium, Cellulomonas, Bacteroides, Succinivibrio, Prevotella, Ruminococcus, Fibrobacter, Firmicutes, Erwinia, Acetovibrio, Microbispora* species, etc (Zhen et al., 2017). Subsequently, during acidogenesis, the acidogenic bacteria would further break down the by-products from the initial hydrolysis to shorter chain volatile fatty acids (VFA), ammonia, CO_2 and H_2S. This is completed with the help of *Peptoccus, Clostridium, Lactobacillus, Geobacter, Bacteroides, Eubacterium, Phodopseudomonas, Desulfovibrio, Desulfobacter, Sarcina* species, etc (Zhen et al., 2017). As the third step ensues, long chain organic acids are then decomposed into mainly acetic acid, H_2 and CO_2. The key bacteria involved in acetogenesis are *Syntrophobacter, Syntrophus, Pelotomaculum, Syntrophomonas,* and *Syntrophothermus* species (Cai et al., 2016). Lastly, methane is produced through the mechanism of acetoclastic and hydrogenotrophic methanogenesis. Acetoclastic methanogenesis is facilitated by *Methanosarcina* and *Methanosaeta* species through the conversion of acetate and water into methane. Meanwhile, methane can also be formed through hydrogenotrophic methanogenesis that reacts CO_2 with H_2 in the presence of *Methanobacterium* and *Methanoculleus* species (Zhen et al., 2017; Cai et al., 2016; Guo et al., 2015; Scherer and Neumann, 2013).

The duration allocated for sewage sludge to undergo anaerobic digestion, also known as sludge retention time (SRT) usually spans from 10 to 20 days. This is because most works agreed that 90% of methane yield could be obtained during the first 14 days of treatment as shown in Fig. 1 (Appels et al., 2008). A comprehensive work had investigated the effect of SRT on anaerobic digestion of sewage sludge (Nges and Liu, 2010). The SRT studied in the work ranged from 3 to 35 days. During the early stage of digester operation (SRT $<$5 days), acidogenesis culminates VFA, driving the system pH to a low value, which is unfavorable for methanogenesis. After 5 days, VFA reduces and creates a more favorable environment for methanogenesis to occur as shown in Fig. 1. Consequently, the accumulative gas production increases exponentially from 62 mL/g VS_{added} (Day 3) to 340 mL/g VS_{added} (Day 15).

2.2. Mass reduction

The ability of anaerobic digestion is reducing the mass of sewage sludge is normally reported in terms of volatile solids (VS) and total solids (TS) reduction. According to APHA standard methods (APHA, AWWA, 2005), VS refers to the organic matter that could be lost when heated up to 550 °C. Meanwhile, TS is defined as the residual material left in vessel after undergoing evaporation at 105 °C. Generally, VS is considered as the biodegradable content in sewage sludge while TS is the combination of biodegradable and non-biodegradable matter. Anaerobic digestion is able to reduce the VS content of sludge by 35–60%, depending on the operational conditions and whether a pre-treatment step is applied (Gebreeyessus and Jenicek, 2016). Meanwhile, TS reduction is around 28–30% according to the work of Chi et al. Chi et al. (2010).

2.3. Carbon recovery through biogas generation

During anaerobic digestion, VS will be degraded by different groups of bacteria, which subsequently produce biogas as the end product. Cao and Pawłowski (2012) reported that 0.8–1.2 m^3 of biogas were obtained with every kg of VS being reduced. This range was quite close to the

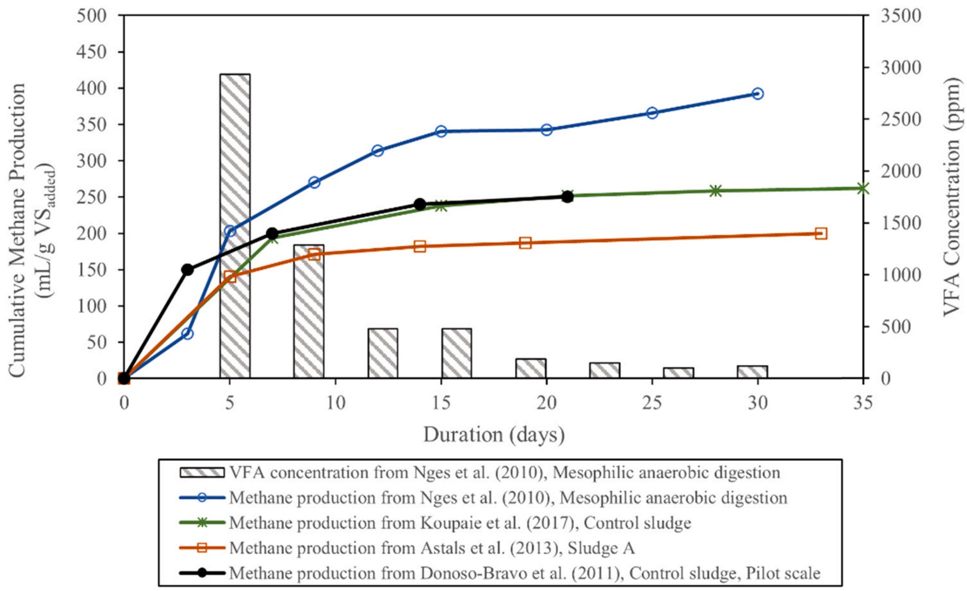

Fig. 1. Methane production and VFA concentration against SRT during anaerobic digestion processes (Nges and Liu, 2010; Hosseini Koupaie et al., 2017; Astals et al., 2013; Donoso-Bravo et al., 2011).

value reported by Appels et al. (2008), Cao and Pawłowski (2012), who stated that biogas production was around 0.75–1.12 m^3/kg VS reduced. The biogas produced is made up of 65–70 vol% of methane, followed by CO_2 at 30–35 vol% and the remaining being contributed by fractions of water vapor, H_2S and H_2 (Appels et al., 2008). The calorific value of biogas ranges between 15.9 and 27.8 MJ/Nm^3. The removal of other unwanted gaseous species is usually conducted to obtain a higher quality biogas that can be used as a fuel for engines, compressors, boilers, and vehicles. The CO_2 and H_2S can be removed via the use of activated carbon as an adsorbent, water-scrubbing system or membrane separation process (Appels et al., 2008; Degrève et al., 2001). Cryogenic separation could also be used to produce high purity methane gas (97 wt %) but at a high cost. Instead of separation, the formation of H_2S could also be inhibited with the addition of Fe^{3+} salts (Appels et al., 2008). The remaining residue after the anaerobic digestion is known as digestate. Since there is still carbon content left in digestate, energy recovery is possible through other methods (Cao and Pawłowski, 2012). However, the leftover is usually highly resistant to degradation, therefore thermal treatment methods such as pyrolysis, and gasification are normally used to treat the digestate.

Cao and Pawłowski had evaluated the energy efficiencies of anaerobic digestion and pyrolysis processes on primary and activated sludges (Cao and Pawłowski, 2012). The routes of evaluation are shown in Fig. 2. Strategy 1 involved the application of anaerobic digestion to produce biogas. Strategy 1b was an extension of strategy 1, whereby pyrolysis of the digestate was conducted to further produce bio-oil and biochar. Meanwhile, Strategy 2 involved the application of pyrolysis to produce bio-oil and biochar. For better comparison of the main products from the anaerobic digestion and pyrolysis, first, a comparison of anaerobic digestion biogas and pyrolysis bio-oil was performed. In the case of 100 kg of primary sludge (VS content: 84 wt%), the energy output was higher at 1573.2 MJ in the form of biogas produced from

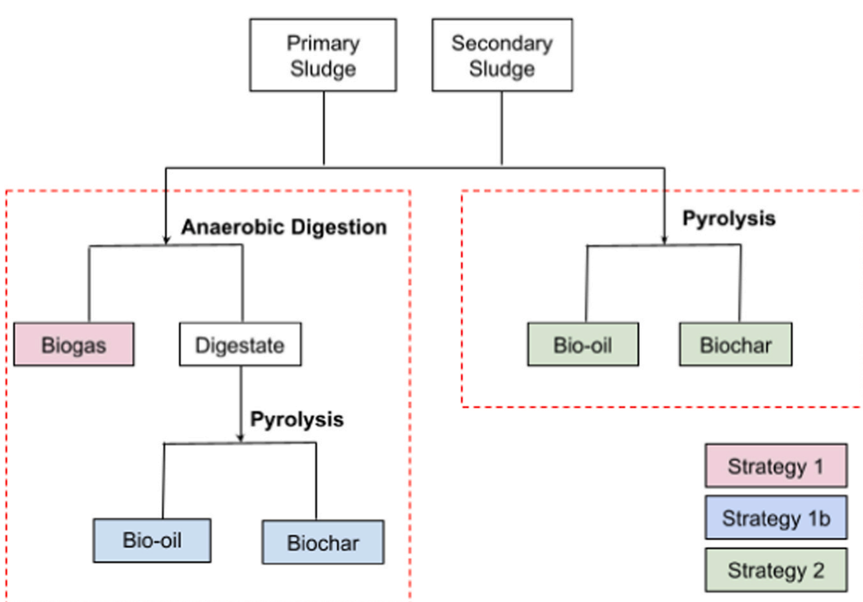

Fig. 2. Flowchart of different strategies in valorizing sewage sludge (Cao and Pawłowski, 2012).

anaerobic digestion as compared with 1554 MJ obtained in the form of pyrolysis bio-oil. Meanwhile, the 100 kg of activated sludge (VS content: 69 wt%) showed a lower energy output of 629.3 MJ in the form of biogas produced from anaerobic digestion as compared with 1147 MJ obtained in the form of pyrolysis bio-oil. This showed that anaerobic digestion was capable of generating more energy output for sewage sludge with high VS content (>84%). It also reiterates the importance of high VS content in producing more biofuel for both treatment methods. However, Cao and Pawłowski (Cao and Pawłowski, 2012) stated that the energy recovery from anaerobic digestion could be further enhanced from valorizing the residual digestate through using pyrolysis to produce bio-oil and biochar via Strategy 1b, generating an additional 581 MJ of energy output from the produced bio-oil and biochar. Hence, once integrated with a subsequent pyrolysis process (Strategy 1 +1b), anaerobic digestion showed a higher energy recovery efficiency for both high and low VS content sludges, namely, primary and activated sludges.

2.4. Nitrogen recovery

Digestate, the residual of anaerobic digestion, is commonly used as a soil fertilizer as it contains a high content of nutrients (N,P,K) and organic carbon (Hermassi et al., 2018). The N content is mainly derived from decomposition of proteins in sewage sludge. In the work of Massé et al. (2007) where swine manure was subjected to anaerobic digestion, nearly 100% of N content was retained in the digestate after the process. The amount of ammoniacal nitrogen in digestate had increased by 36.5 wt%, signifying the occurrence of conversion of nitrogenous compound into ammoniacal nitrogen. As the digestate was allowed to settle, the settled sludge cake carried 31.5 wt% of ammoniacal nitrogen, while the supernatant that was usually returned to the wastewater treatment plant contained 67.1 wt% of ammoniacal nitrogen. On the other hand, 74.5 wt% of the P content was retained in the digestate, with 24.95 wt% in the supernatant and 46.4 wt% in the settled digestate. Through the addition of magnesium at a Mg/N/P molar ratio of 1:1:1, 63–64 wt% of NH_4^+ and PO_4^{3-} could be recovered as struvite ($MgNH_4PO_4$) and bobierrite ($Mg_3(PO_4)_2$) (Hermassi et al., 2018). Other alternatives that show promising efficiency in recovering N content include air stripping, thermal drying, hydrophobic and vacuum membranes (van der Hoek et al., 2018).

2.5. Heavy metal stabilization

Besides, anaerobic digestion had been reported with ability to reduce the bioavailability of heavy metals in sewage sludge by converting them from the readily exchangeable form into the more stable form that is harder to leach into the surrounding environment (Hong et al., 2007). Generally, the metals could be classified into five different forms including exchangeable (F1), acid-soluble (carbonate bound, F2), oxidizable (organic bound, F3), reducible (Fe/Mn bound, F4) and residual (F5). F1 was the least stable form of heavy metals, as it could be readily leached into water and subsequently, being uptake by plants and animals. This is then followed by F2 that could be readily available under an acidic environment. These two forms of metals (F1 + F2) were the main contributors to the bioavailability and toxicity effect of sludge. Meanwhile, the F3, F4 and F5 forms were relatively stable in soil and had the lowest bioavailability (Hong et al., 2009; Lim et al., 2013).

However, for anaerobic digestion, Zhu et al. (2014) introduced one new fraction which is the water-soluble fraction. This refers to the heavy metals which were solubilized in the supernatant of the digestate, meanwhile F1 to F5 were used to categorize the heavy metals in the settled digestate. When taking into consideration of all the six heavy metal forms including the water-soluble heavy metal concentration in the supernatant, it was reported that most of the heavy metals tested were within China's control standards for pollutants in sludge for agricultural use (GB4284–84), except for Cu, Zn and Cd (Zhu et al., 2014). However, putting the more stable F3-F5 aside, the toxicity should only be judged by F1, F2 and the water-soluble forms. In this scenario, only Zn did not meet the standard required for acidic soil. Although the settled digestate could not be applied on acidic soil (pH<6.5) in China, it still meets a more lenient limit demarcated by the same standard GB4284–84 for the application on alkali soil (pH > 6.5). In addition, Zhang and Wang (2020) also showed that the addition of Fe-Mn modified biochar could facilitate the immobilization heavy metals. In their work, the bioavailable form (F1 + F2) of Cr, Ni and Cd reduced by 49.76%, 31.46%, and 47.69% respectively when 0.12 g of Fe-Mn modified biochar was added to every 1 g of sludge (Zhang and Wang, 2020). They attribute this to the additional active sites and functional groups of Fe-Mn modified biochar to interact with the heavy metals.

2.6. Pathogen reduction

The operating conditions of anaerobic digestion can be divided into mesophilic and thermophilic conditions. Under mesophilic condition, the temperature is in the range of 30–38 °C, while that required by thermophilic condition is higher from 50 to 57 °C (Cao and Pawłowski, 2012). Since thermophilic anaerobic digestion operates at higher temperature, it consumes more energy. However, higher temperature does improve the effectiveness in reducing pathogen. As consequence, the residual from anaerobic digestion, known as digestate, meets the pathogen standard of Class A biosolid as set by US EPA (Gebreeyessus and Jenicek, 2016). Being classified as Class A biosolid, the digestate produced from thermophilic anaerobic digestion can be suitably applied to both agricultural land and public access land such as private lawn and home garden. Meanwhile, the mesophilic anaerobic digestion can only produce a digestate that meets the criteria as Class B biosolid (Al-Gheethi et al., 2018), which is inferior as compared with Class A biosolid. It contains more pathogen and is not able to meet the more stringent standard. The application of Class B biosolid is limited to only agricultural land (USEPA, 1994). Generally, the amount of pathogen reduction increases with digestion duration. The most drastic reduction happened between day 3 and day 6, recording a 4 $\log_{10}$ reduction. Subsequently, from day 6 to day 60, only 1 $\log_{10}$ reduction was achieved (Scaglia et al., 2014). The content of *Salmonella* spp. reduced from 81 MPN/g to nil in 3 days for a thermophilic anaerobic digestion, while it took 13 days to achieve a similar result for a mesophilic anaerobic digestion, proving that the thermophilic anaerobic digestion is more superior in pathogen reduction. In both anaerobic digestion processes, no living eggs of helminths was detected (Scaglia et al., 2014).

2.7. Pre-treatment

To enhance the biogas yield from anaerobic digestion, pre-treatment of sewage sludge is often carried out. Different pre-treatment methods have been tested and proved effective, including thermal pre-treatment, chemical pre-treatment, enzymatic pre-treatment and mechanical pre-treatment. The prime objective of pre-treatment is to disintegrate the sludge in advance. This is because the sewage sludge is an unique biomass that contains extracellular polymeric substances (EPS) (Tyagi and Lo, 2011). The EPS is a gel-like structure that binds together the microbes and holds plenty of nutrients within it. It is made up of 58% of protein and 9% of polysaccharides (Mikkelsen and Keiding, 2002). By disrupting the EPS structure, more biodegradable nutrients could be released, and this enhances the potential of anaerobic digestion in extracting more biogas.

The effectiveness of pre-treatment is normally measured in terms of soluble chemical oxygen demand (sCOD) and the solubilization ratio. The sCOD increases when more bio-degradable organic matter is released into the soluble phase (Hosseini Koupaie et al., 2017). This could be in the form of intracellular biopolymers such as reducible sugars, nuclei acids, lipids or even extracellular biopolymers such as disintegrated EPS (Hosseini Koupaie et al., 2017). Although the total chemical oxygen demand (COD) in the digester might remain unchanged, the solubility ratio will increase when the sCOD increases. A higher solubility ratio signifies that the pre-treated sewage sludge is more easily degraded and could produce a higher biogas yield.

Many different pre-treatment methods have been studied to improve the biodegradability of sewage sludge. For instance, the microwave pre-treatment of activated sludge at a heating duty of 336 kJ/kg total solid increased the sCOD content by 214% and subsequently 50% increase in the biogas production was observed during the mesophilic anaerobic digestion (Appels et al., 2013). A pilot scale study of thermal pre-treated activated sludge at 65 ^{o}C for 20 min showed a 2- to 3-fold increase in the solubility ratio and as expected, an increment of 30–40% in biogas production. Similarly, NaOH and H_2SO_4 had been tested for the chemical pre-treatment of sewage sludge. Results showed that an alkaline pre-treatment regime enhanced the sCOD more than an acidic pre-treatment. This could be due to refractory compound formation when acid is used for sludge pre-treatment, causing a reduction in the outcomes of the digester as refractory compounds do not degrade easily. In their work, a pH at 10 released the highest amount of methane, followed by that released at pH 5 and then pH 2 (Tulun and Bilgin, 2019).

2.8. Co-digestion

Co-digestion involves the anerobic digestion of sewage sludge with another organic waste. Co-digestion of sewage sludge has shown improved VS reduction, enhanced biogas yield and improved in heavy metal stabilisation. In the context of anaerobic digestion, an optimum C/N ratio for anaerobic digestion is usually between 20:1 and 30:1 (Monnet, 2003). Sewage sludge that has a low carbon-to-nitrogen (C/N) ratio within 5:1 – 16:1 is thus considered nutrient deficient for anaerobic digestion to thrive (Rynk: Robert, 1992). Low C/N ratio will lead to excess ammonia formation, thereby increasing the system pH above 8.5 which is not conducive for methanogens (Monnet, 2003). Therefore, by deploying a co-substrate, the missing nutrients in sewage sludge can be counterbalanced, resulting in improved C/N ratio for a better anaerobic digestion performance (Chow et al., 2020).

In a large-scale plant, the co-digestion of sewage sludge with organic fraction of municipal solid waste has shown positive effect on biogas production. In Frutigen, Switzerland, a full-scale study to validate the feasibility of co-digesting organic fraction of municipal solid waste with sewage sludge was conducted (Edelmann et al., 2000). Organic municipal solid waste was collected from regional supermarket chains and food wastes from hospitals, shredded and added for co-digestion. Experimental result shows that the biogas production increases by 27% with the addition of 20% solid waste into the system, without affecting the process stability.

Crude glycerol, a by-product from biodiesel production, has also been used as a co-substrate due to its high carbon content (BOD of 97080 mg/L) (Chow et al., 2015). When 1 vol% of the crude glycerol was added into an anaerobic co-digestion system, an increase of 115% in the methane yield was recorded (Fountoulakis et al., 2010). However, any higher dosage than that would reduce the alkalinity of the digester and thus culminating in the accumulation of VFA which inhibited methanogens (Razaviarani et al., 2013). Next, free oil and grease can also be co-digested with sewage sludge at optimum ratio (50:50 in terms of VS). Any higher ratio would result in an excessive accumulation of long chain fatty acids in the biomass, limiting the mass transfer and resulting in aggregation and clogging (Chow et al., 2020). In addition, co-digestion of different sludges is also possible. This is shown in the work of Babel and Sae-Tang (Babel et al., 2009), who mix sewage sludge and brewery sludge for co-digestion. As a result, the mixed digestate finally meets the pollutant limit of biosolid as set by US EPA.

In general, for an effective co-digestion, an optimum mixing ratio must be investigated as it varies according to the type of co-substrates envisaged for use. Furthermore, a proper mixing is also very important to produce a homogenous feed. Although co-digestion has shown its effectiveness in facilitating sewage sludge anaerobic digestion, it looks more like a symbiosis relationship. For instance, the VS reduction and biogas yield improves because the co-substrate is often more biodegradable. Furthermore, all the co-substrate discussed earlier has a lower heavy metal concentration, thereby creating a dilution effect for the digestate to achieve EPA standard. It is highly possible that the digestion of co-substrate alone without sewage sludge will bring more economic value.

2.9. Microbial electrolysis cell (MEC)

Lately, MEC has been paired up with anaerobic digestion to produce a higher methane yield and stabilize the anaerobic digestion system (Zhen et al., 2017; Yu et al., 2018). In MEC, exoelectrogenic bacteria consume the organic matter anaerobically and generate electrons at the anode. The electrons then travel to the cathode side with the help of an external voltage supply. At the cathode side, electrons are used by methanogens to convert CO_2 and acetic acid into methane gas. Liu et al. (2016) conducted a MEC-anaerobic digestion process of activated sludge at 0.8 V and obtained a methane production rate of 91.8 g/m^3, which is 3 times higher than that obtained from the typical anaerobic digestion process (30.6 g/m^3). This was supported by the increased population of methanogens. They reported that the highest increment was *Methanobacterium* (hydrogenotrophic bacteria), followed by *Methanosaeta* (acetrophic bacteria) and *Methanospirillum* (hydrogenotrophic bacteria) species. They also observed that the degradation rate of organic matter was higher in MEC-anaerobic digestion with a lower remaining content of polysaccharides and proteins. This accelerated rate of degradation was attributed to the growth of exoelectrogenic bacteria *Geobacter* sp., which could anaerobically decompose a variety of organic substrates (Yu et al., 2018).

Researchers had shown that the effectiveness of MEC could be affected by the external voltage applied. In the work of Choi et al. (2017), different voltage supplies (0.5, 0.7, 1 and 1.5 V) were tested on the MEC of mixed substrate containing anaerobic digestion effluent and growth medium. Results showed that methane production improved with increasing voltage from 0.5 to 1 V. However, the methane production decreased at 1.5 V. Likewise, Linji et al. (2013) also reported that 0.8 V was the optimum voltage for their MEC experiment on activated sludge. Despite of the potential of MEC, so far only lab-scale MEC had been proven successful.

3. Vermicomposting

3.1. Overview

Vermicomposting, also known as vermistabilization, combines the use of earthworms and microorganisms to accelerate the organic waste degradation process. The earthworms will ingest, grind, and digest the organic waste with the help of microflora in their gut, thereby modifying the physical, biological, and chemical status of the organic waste. The excretion of earthworms is known as vermicast, is in the form of a homogenous, stable, fine, humified and microbially active organic matter. Besides, it contains nutrients such as K and P in a more readily available form for use as fertilizer to grow plants (Gupta and Garg, 2008; Huang et al., 2013; Liu et al., 2005). Furthermore, the growth of earthworm populations could also be used as a high-protein feed for farmed animals such as chickens. All these can be achieved at a low cost, making this treatment method economical and ecological to be implemented, especially in developing countries. However, vermicomposting requires an even longer treatment time as compared to anaerobic digestion, and it is not capable of generating any biofuel too. The experimental duration for vermicomposting in most studies spans from 30 days to 60 days (Gupta and Garg, 2008; Khwairakpam and Bhargava, 2009; Hait and Tare, 2011; Huang et al., 2013; Liu et al., 2005).

Vermicomposting of sewage sludge has reached the industrial-scale application, but it is only limited to co-substrate basis, where sewage sludge is mixed with pulp mill solid. In Tasman, New Zealand, the commercial operation of vermicomposting was running based on the mixture of sewage sludge and paper mill sludge. As of 2008, the vermicomposting system was processing 2000 tons of pulp mill solid and 900 tons of municipal sewage sludge per year (Quintern, 2014). The end product was a good quality vermicast that met the criteria as New Zealand biosolid (Quintern and Morley, 2017). Nonetheless, it should be highlighted that the raw sewage sludge contained a comparatively low heavy metal concentration.

3.2. Mass reduction

Based on literature review, mass reduction in terms of VS or TS reduction is less emphasized on. The main objective of vermicomposting is to stabilize the sewage sludge by reducing the organic matter content in the sludge, thus preventing the growth of bacteria later on. This reduction of organic matter can be measured through the total organic carbon (TOC). The TOC reduces as the duration of vermicomposting increases. This is because part of the organic carbon is used up by earthworms and bacteria for respiration, converting it into CO_2 and reducing the volatile solids in the biomass (Hait and Tare, 2011). In the work of Khwairakpam and Bhargava, (2009), the TOC in sewage sludge was reduced by 10–25% in pure cultures and 10–17% in mixed cultures after 45 days. The highest TOC reduction recorded was 25.46% when pure culture of *P. excavates* was used. Meanwhile, control sewage sludge without earthworm only recorded a TOC reduction of 8.7%. Pure culture refers to the use of only one species of earthworm during vermicomposting, while mixed culture uses two or more species of earthworm. TOC reduction of 20–30% is commonly agreed by the experimental values in several studies when vermicomposting is deployed on sewage sludge (Gupta and Garg, 2008; Huang et al., 2013), which is about 2.5–3 times higher TOC reduction that the control without earthworm.

3.3. Carbon recovery through earthworm growth

In vermicomposting, earthworm ingests the carbon content in the substrate to grow and populate. Consequently, the weight of earthworms increases along with the vermicomposting duration. After 45 days of vermicomposting, the weights of earthworms are reported to increase 9.09–28.57% (Khwairakpam and Bhargava, 2009). Gupta and Garg (2008) showed that the total weight of earthworms increased steadily to a peak before losing weight due to exhausted nutrients in the substrates. For sewage sludge, the nutrient exhaustion point happened around day 35.

3.4. Nitrogen recovery

After vermicomposting of sewage sludge, the concentration of total N in the vermicast always increases due to the reduction in substrate dry mass as water is lost through evaporation and respiration via earthworm (Gupta and Garg, 2008; Huang et al., 2013). According to the work of Antoniou et al. (1990), the vermicomposting of straw pallet and cattle manure mixture showed that 76.3–79.4% of original total N content in the substrate mixture could be recovered in the vermicast. This high N content retention could be attributed to the low pH profile (5.5–7.2) throughout the vermicomposting period because high pH above 7.5 favors the formation and volatilization of NH_3 (Khwairakpam and Bhargava, 2009).

Khwairakpam and Bhargava (2009) also reported that the concentration of NH_4^+ was always higher than NO_3^- throughout the 45 days of vermicomposting. However, the concentration of NH_4^+ declined sharply between the 30th and 45th days, whereas NO_3^- started to increase. The rapid reduction of NH_4^+ at the end could be attributed to the increment of pH from an average of 5.5–7.2. The increment of pH would unfavour ammonia protonation (Oonincx DGAB et al., 2015; Pan et al., 2018). Concurrently, this could also be due to the formation of NO_3^- as it had been reported that the nitrifying bacteria only started to colonize at the later stage of composting (Bernal et al., 2009). Furthermore, the growth rate of nitrifying bacteria was also reported to be higher as the substrate approached pH 7–8 from both acidic and alkaline conditions (Watson et al., 1981).

3.5. Heavy metal stabilization

In terms of heavy metals stabilization, inconclusive findings are reported thus far. Khwairakpam and Bhargava (2009) found that the final concentration of Cu in vermicast remained unchanged, whereas Pb, Zn and Mn were reduced by around 30%, 20% and 50%, respectively. The decrease in heavy metals could be due to the accumulation of heavy metals in the chloragosomal tissue in the body of earthworm (Fischer and Molnár, 1992). It has been suggested that when earthworm was placed in heavy-metals contaminated substrate, a cation-exchange mechanism would take place in the chloragosomal tissue of earthworm; for instance, the Pb would displace Ca in the chloragosomal (Fischer and Molnár, 1992; Fischer, 1977). This could be due to the self-protective mechanism of earthworm that could prevent heavy metal in its body from entering the haemoglobin pathway (Morgan and Morgan, 1989).

A similar finding was reported by Huang et al. (2013), exhibiting that the heavy metal concentrations reduction was ranging from 9.8% to 20.5% after 30 days of vermicomposting with two different types of earthworm. In contrast, heavy metal concentrations were found to increase by more than 50% for Fe, Cu and Zn in another work that tested municipal plant primary sludge and cow dung as the feeds for vermicomposting (Gupta and Garg, 2008). Hartenstein and Hartenstein (Hartenstein and Hartenstein, 1981) explained that it could be due to the mineralization process that was accelerated in the presence of earthworm as the final heavy metal concentration was more than the concentration in the control (without earthworm).

Later, Liu et al. (2005) determined that the earthworms could absorb heavy metals into their body. This was conducted by measuring the heavy metal content in the earthworm's body before and after vermicomposting. However, there was also a maximum concentration that the earthworm could tolerate. From the experiment, this was determined to be around 500 mg/kg for Cu and 20 mg/kg for Cd in the sewage sludge. Beyond these lethal concentrations, earthworm's ability in absorbing heavy metals was reduced. Their work also showed that vermicomposting did not change the fraction of heavy metals from one form to another (Liu et al., 2005). For instance, the amounts of carbonate Cd were 10.53 wt% and 10.27 wt% with and without earthworms, respectively. Yet, when the sewage sludge and vermicast were separately added into two similar soil pots for planting Chinese cabbage, it was discovered that the Cu and Cd contents in the cabbage were lesser than that found in vermicast-raised cabbage. This proved that the vermicast could reduce the bioavailability of heavy metals, not by changing metal fractions, but by the uptake of heavy metals to itself and perhaps, changing the soil characteristics.

3.6. Pathogen reduction

Vermicomposting could also reduce most of the pathogens in sewage sludge (Khwairakpam and Bhargava, 2009). Up to 99.9% of the fecal coliform was reduced after a vermicomposting of 45 days, recording the final fecal coliform population of between 1300 and 2300 MPN/g. The reduction of pathogens is due to the antibacterial effect of the coelomic fluid released by earthworms during their physiological activities such as respiration and burrowing (Valembois et al., 1982). Although achieving a high reduction, the final fecal coliform still did not meet the pathogen requirement of US EPA Class A biosolid, which limits fecal coliform to be 1000 MPN/g. Nevertheless, the vermicast managed to meet USEPA Class B biosolid within just 15 days of vermicomposting. The authors also showed that a mixed culture of worms was more effective in destroying pathogen as compared with a pure culture (Khwairakpam and Bhargava, 2009). In another work, pathogen level after vermicomposting had met the US EPA Class A standard, but the sample was first pre-treated with aerobic composting (Hait and Tare, 2011).

3.7. Co-substrates

Domínguez et al. (2000) pointed out that sewage sludge was not a suitable substrate for vermicomposting due to its low C/N ratio. Co-substrate can thus be deployed to increase the C/N ratio. For example Gupta and Garg (2008) mix cow dung (C/N ratio: 69.3) and sewage sludge (C/N ratio: 16.1) together for vermicomposting. This resulted in an improved TOC reduction of 62.6% as compared with 30.5% for the control (only sewage sludge). This in turn translates to a 48% weight increment for the earthworms.

Instead of introducing a different co-substrate, Huang et al. (2013) utilized vermicast itself as a co-substrate to see its effect on sewage sludge vermicomposting. Interestingly, despite the low initial C/N ratio in sludge, the C/N ratio still showed a massive reduction from 9.72 to 5.16 (*E. foetida*) and 5.08 (*B. parvus*) in just 30 days as shown in Fig. 3. The growth rate and reproduction rate of both earthworm species also increased by 28–37%. At the same time, the total N, P, K increased as much as 15.4–37.5%, while heavy metal content was reduced by 9.8–20.5%. This could be due to the effect of increased microbial activity contributed by the addition of vermicast, which is rich and diverse in bacterial microflora.

4. Black soldier fly larval valorization

4.1. Overview

The research into black soldier fly, *Hermetia illucens* was initiated around the 1950 s due to its ability as a natural control to the disease-transmitting house fly, *Musca domestica*. Black soldier fly (BSF) population could reduce the population of *Musca domestica* by 94–100% (Sheppard, 1983). BSF is non-identical to the typical household fly that it is not a pest and does not transmit diseases. BSF is an insect that is normally found feeding on decomposed organic matters such as rotting vegetables, fruits, plant litter and manure in the tropics and warm temperate regions. It has a short lifespan of about 54 days from its first emergence from cocoon to the stage it reproduces and dies. As shown in Fig. 4, the life cycle begins with the female adults ovipositing about 320–620 eggs near a decomposing organic matter source (Tomberlin et al., 2009). The eggs will normally hatch after 4 days, and the young larvae then feeds on the decomposed organic matter. Next, the larvae will undergo stages of larval development from 1st until 5th instar stages. The duration of growth varies according to the quality of food sources it has access to, ranging from 15 to 52 days (Gold et al., 2018). Next, having accumulated enough proteins and lipids for the transformation into prepupae, the larvae then stop ingesting any matter. Its mouth part slowly turns into a hook-shaped structure that allows it to move to the dry and dark place and ensconce itself for pupation. This is the 6th instar stage and the last stage of the larvae before turning into a cocoon and then emerging as a fly after about a week. The fly usually mates after 2 days of pupal emergence (Tomberlin and Sheppard, 2002). Again, after mating, it takes another two days for the female to oviposit eggs, before dying of exhaustion at a maximum of 9 days (Booth and Sheppard, 1984; Raksasat et al., 2020; Samayoa et al., 2016).

Similar to vermicomposting, it uses a combination of BSFL and microbial population to speed up the decomposition process of organic matter. During their larval stage, BSFL ingests organic matter ferociously and stores them in the form of body fat (Craig Sheppard et al., 1994). This is crucial to sustain their development during prepupae and adult stage, as they no longer consume food at that two stage. As consequence, BSFL contains high contents of lipid and protein at 47% and 41%, respectively (Guo et al., 2021). Simultaneously, solid excrement, also known as frass is discharged by BSFL and it can be used as a fertilizer.

The high protein content of BSFL makes it suitable to be used as a part of the poultry diet. There had been many researches showing positive results of using BSFL as a feedstock meal for poultry, swine and fish (Li et al., 2016; Sprangers et al., 2017). Skrivanova et al. (2007), for instance, reported that a diet of BSFL could help in managing the microbiota in the small intestine of pigs and poultry, thereby fortifying the poultry's health. Meanwhile, Kim et al. (2020) reported that a diet of BSFL oil could improve the gut health and enhance the antioxidant capacity of broiler chickens. Recently, BSFL lipid is also extracted to produce biodiesel.

Numerous kinds of organic waste, also known as substrates have been administered to grow BSFL. This includes food waste, cow manure, chicken manure, coconut endosperm, soybean curd, vegetable waste, human feces and sewage sludge. Different kinds of substrates bring about significant effect on the growth and development of BSFL (Sprangers et al., 2017; Lalander et al., 2019; Leong et al., 2016; Liu et al., 2020a). The application of BSFL treatment has reached industrial scale for numerous organic wastes. For example, InnovaFeed in France collects agricultural by-products from a starch production plant to rear

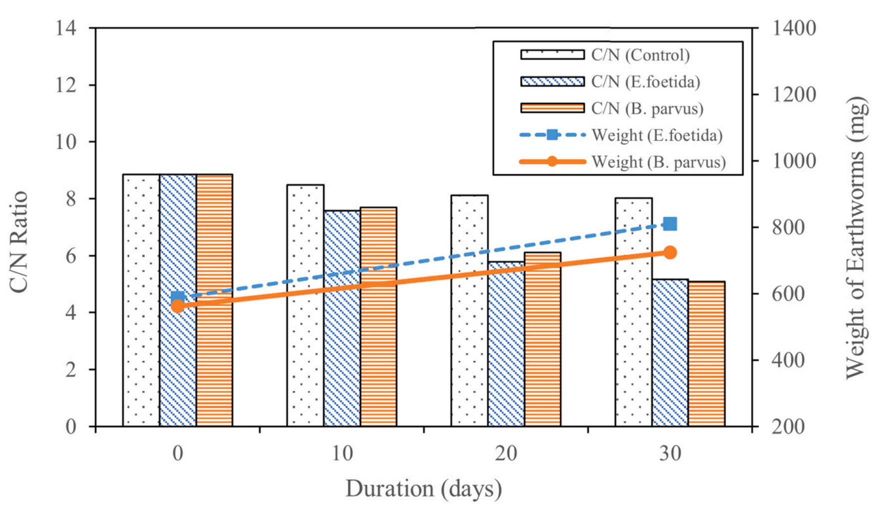

Fig. 3. Effects of vermicast addition as co-substrate on sewage sludge vermicomposting process (Huang et al., 2013).

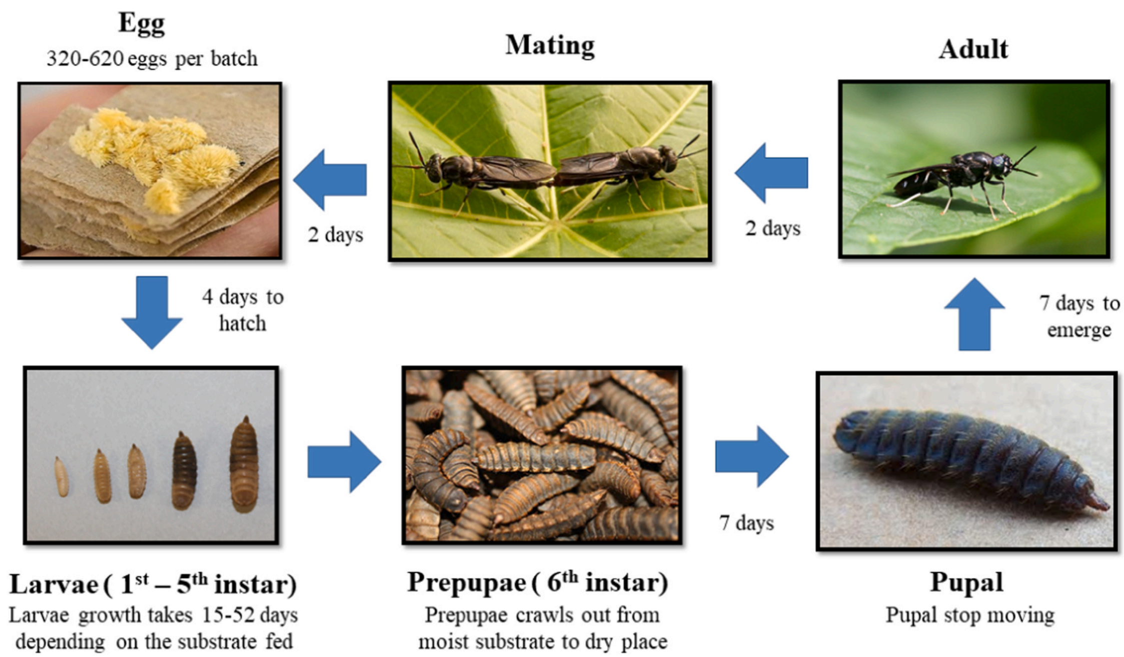

Fig. 4. Lifecycle of black soldier fly (Gold et al., 2018; Tomberlin et al., 2009; Tomberlin and Sheppard, 2002).

BSFL (PhI et al., 2020). Subsequently, BSFL is collected as poultry feedstock as well as for oil production. However, till date, limited studies on sewage sludge as BSFL substrate have been conducted. Hence, the discussion below regarding performance of BSFL treatment will be of a combination of sewage sludge and other substrates.

4.2. Mass reduction

Mass reduction for BSFL treatment is commonly expressed in terms of dry matter reduction. Lalander et al. (2019) employed the BSFL to treat three different types of sludges, namely primary sludge, activated sludge and digested sludge. Primary and activated sludges are both raw sludge that is untreated, whereas digested sludge refers to sludge that is pretreated via anaerobic digestion. BSFL treatment on primary sludge and activated sludge could achieve 63.3% and 49.2% dry weight reductions, respectively. Meanwhile, only 13.2% dry weight reduction was recorded for digested sludge. This is because for digested sludge, most of its VS has been released during anaerobic digestion, leaving the non-biodegradable matter in the sludge, resulting in low mass reduction. In the work of Raksasat et al. (2021), a dry mass reduction of 73.34% is recorded after 27 days. Meanwhile, only 2% dry matter reduction was reported by Cai et al. (2018). This could be due to the difference in drying temperature used by all three works. For instance, Lalander et al. Lalander et al. (2019) dry the residue at 80 ^{o}C for 48 h while Cai et al. (2018) dries the residue at 105 ^{o}C for 10 min, and then 60 ^{o}C for 2 days.

To enable comparison among BSFL treatments and also other methods such as anaerobic digestion, a reasonable temperature should be unanimously agreed and used for all future works. Since anaerobic digestion often measures TS reduction, the temperature range of 103–105 ^{o}C as recommended by APHA standard is suitable for the adoption (APHA, AWWA, 2005). Furthermore, according to the thermogravimetric chart of sewage sludge, the water loss happens at the temperatures below 120 ^{o}C with most water loss occurs at 87.6 ^{o}C (Gao et al., 2014). Hence, the drying temperature should be at least above 87.6 ^{o}C. Comparing with other substrates, the BSFL treatment could reduce the dry matters of animal manure by 42–56% (Sheppard et al., 2002), food waste by 55.3% and human feces by 47.7% (Lalander et al., 2019).

4.3. Carbon recovery through BSFL growth and biodiesel production

During BSFL treatment, carbon recovery is achieved through the weight gained by BSFL. Subsequently, the heavier BSFL is harvested to be used as a poultry feedstock or to be converted into biodiesel. The characteristic of substrate affects the growth of BSFL significantly. When organic waste such as food waste, vegetable waste and abattoir wastes were fed to BSFL, the wet weight of each prepupa was within the range of 212–252 mg (Lalander et al., 2019). When animal manure was used, the wet weight of prepupa reduced to 164 mg. However, when primary sludge and activated sludge were used, the prepupa weight dropped to 137 and 145 mg. Furthermore, the time taken for the first prepupa to emerge in the food waste was only 14 days as compared to 21 and 30 days for primary sludge and activated sludge (Lalander et al., 2019) as shown in Table 3. This indicates that sewage sludge is not a suitable substrate for BSFL rearing, and it could be due to the nutrients of sludge that are rather hard to be accessed because of the EPS structure that protected the microbial population (Dohányos et al., 2004; Ruffino et al., 2015). In addition, this shows that C/N ratio is not a good indicator to determine the quality of substrates. For instance, even though the activated sludge had a similar C/N ratio as compared with the mixture of abattoir, fruit, and vegetable waste, both substrates showed a big difference in the BSFL development properties (Lalander et al., 2019). Based on the substrate characteristic, Lalander et al. (2019) inferred that VS and total N were the two important parameters that determined the quality of substrates to suit the BSFL diet.

Similar result was reported by Leong et al. (2016), who reared BSFL with fresh sewage sludge. They reported a negative growth rate of −0.07 g/day, where the prepupae weight decreased after four days of rearing. Raksasat et al. (2021) in their work, reported a 14.38 mg prepupa dry weight, which was 3 times smaller than the larva reared in the mixed substrate of sewage sludge and palm kernel expeller. Basically, most studies do not show a positive result using sewage sludge as substrate to rear BSFL.

In order to produce biodiesel, body fat can be extracted from BSFL and then undergo transesterification process. Li et al. (2011) attained 93–96% of biodiesel yield from the crude fat extracted from BSFL through acid-catalyzed esterification of free fatty acids, followed by alkaline-catalyzed transesterification. The resultant biodiesel was of

Table 3
Larval development and process efficiency of BSFL treatment using different sewage sludges.

Type of Sewage Sludge	Sewage Sludge Properties		Process Efficiency		Larval Development	Reference
	Total Volatile Solid (% of DM)	Total Nitrogen (mg/g)	Biomass Conversation Rate (% of DM)	Material Reduction (% of DM)	Observation	
Primary Sludge	77.8	4.7	2.3	63.3	Individual prepupa final weight: 137 mg Time for first prepupa: 16–21 days Survival rate: 81.0%	(Lalander et al., 2019)
Secondary Sludge	77.1	4.2	2.2	49.2	Individual prepupa final weight: 145 mg Time for first prepupa: 30 days Survival rate: 76.2%	(Lalander et al., 2019)
Digested Sludge	63.2	6.8	0.2	13.2	Individual prepupa final weight: 70 mg Time for first prepupa: 39–42 days Survival rate: 39.0%	(Lalander et al., 2019)
Sewage Sludge	–	–	1.1	73.3	Individual prepupa final weight: 14 mg Duration of experiment: 27 days	(Raksasat et al., 2021)
Sewage Sludge	–	12.7–34.1	-0.8	1.0	Total prepupae weight gain: − 500 mg Duration of experiment: 30 days	(Cai et al., 2018)
Sewage Sludge	–	–	-0.09	5.4	Total prepupae weight gain: −400 mg Duration of experiment: 8 days	(Leong et al., 2016)

DM: Dry mass

good quality meeting the European biodiesel standard (EN14214) in terms of density, viscosity, cetane number and sulfur content. Furthermore, the BSFL biodiesel had a high percentage of saturated fatty acid methyl ester (FAME) of 67.6% as compared with crop-based biodiesels such as rapeseed biodiesel (4.3%) (Li et al., 2011). This is a highly sought-after quality as saturated FAME is more chemically stable than unsaturated FAME and hence, has a higher oxidative stability as opposed to the biodiesel with high unsaturated FAME.

Based on the work of Zheng et al., the final yield of biodiesel from restaurant waste is only 2.4%. The yield of larval to restaurant waste is 6.5%, yield of body fat to larval is 39.2%, and yield of biodiesel to body fat is 93.1%. In order to produce a breakthrough in BSFL biodiesel production, it is thus crucial to overcome the limitation of the first yield, which measures how much larval weight is gained from the organic waste. Once a healthy and heavy larva is obtained, body fat extraction and subsequent transesterification process generally grants a much higher yield.

To maximize biodiesel production, Wong et al. (2019) proposed to bring forward the harvesting session of BSFL from the 6th instar stage (prepupae) to the 5th instar. They discovered that the lipid content in BSFL at the 5th instar stage was higher than that observed at the 6th instar stage, i.e., 34.23 wt% versus 25.88 wt%, respectively. This is because in the 5th instar stage, the BSFL are still actively ingesting organic substrate to grow. However, reaching the 6th instar stage, the mouth part slowly turns into a hook-shaped beak to prepare it for ensconcing itself in the pupal stage. This transformation from the 5th to the 6th instar stages takes weeks, hence consuming the lipid content stored in its body. By harvesting the BSFL earlier, a significant improvement of 32% in BSFL lipid was collected for the transesterification process (Wong et al., 2019). Furthermore, an early collection of BSFL did not affect the FAME composition from BSFL, where C12:0 was still the dominant FAME.

Additionally, it is observed that early prepupation with lower larval weight happens when a high protein substrate is fed to BSFL as compared to the protein deficient substrate (Lalander et al., 2019). To maximize biodiesel production, an optimisation study should be performed to see if the shorter development time could counter the effect of lower larval weight. This is because the total biodiesel generated per day could be manipulated by both larval weight and days required for prepupae formation. Moreover, in-depth research should also be conducted to study the effect of feeding substrates with different protein contents on the BSFL lipid yields and FAME compositions to ensure that the quality of BSFL biodiesel is not affected by the early prepupae conversion.

4.4. Nitrogen recovery

When it comes to total N content, it had been calculated that 68 wt% of the initial N content would remain in the residual biomass after 24.3 days of BSFL treatment on dairy manure (Rehman et al., 2017). This could be apprehended as some of the nitrogen content were volatilized and some had been consumed by BSFL for growth. In another experiment of BSFL treatment for substrate sewage sludge and food waste, the concentration of NH_4^+ was observed to increase and peak at day 5 before reducing (Liu et al., 2020a). The increment of NH_4^+ in food waste was, however, more tremendous, reaching 1001 mg/kg while only 438 mg/kg for sludge sample. This can be attributed to the lower pH in food waste sample, which promoted a shift in equilibrium towards ammonia protonation to form NH_4^+ ions (Pan et al., 2018; Bernal et al., 2009). Liu et al. (2020a) also plotted the changes of NO_3^- content throughout the BSFL treatment. No significant change had occurred to the concentration of NO_3^- throughout the 10 days of experiment, with NO_3^- concentration hovering between 20 and 24 mg/kg for both food waste and sewage sludge sample. This observation where NO_3^- was consistently lesser than NH_4^+ was alike with the finding for vermicomposting. The conservation of nitrogen content in sewage sludge is very critical to produce high-quality frass.

4.5. Effects of heavy metal on BSFL and heavy metal stabilization

Studies regarding the fate of heavy metals during BSFL treatment are quite limited to date, especially on sewage sludge. This is because most BSFL studies focus on food waste, restaurant waste and animal manure that are not highly contaminated with heavy metals. Thus, the leftover frass is foreseen not to have any complication with the issue of heavy metals. Nonetheless, the effects of heavy metals on BSFL growth and survivability as well as the concentrations of heavy metals in the larval body and the frass after the BSFL treatment had been investigated.

To study the effect of heavy metals on the growth BSFL, Cai et al. (2018) has prepared seven artificial diets made up of bran and wheat, with different heavy metals concentration corresponding to the sludges collected from seven different municipal plants. Based on the principal component (PC) analysis conducted (Cai et al., 2018), the PC1 that explained 39.16% of the total variability showed that high concentrations of Pb, Ni, B and Hg could retard the growth of BSFL. Meanwhile, high concentrations of Cu, Zn, Cr, Cd and Hg could slightly harm the larvae survivability as exhibited by PC2 with 28.69% of total variability. Remarkably, the combined heavy metals did not have a drastic influence on BSFL as the survival rates for all samples including the control were still high, ranging within 90–100%, while the conversion rates (weight increment in BSFL/weight reduction in substrate) were quite constant at 9–11% (Cai et al., 2018). Despite that these results were obtained based on artificial diets with emulated heavy metal content, the BSFL was proven to be highly tolerant towards the heavy metals, thus having the potential to be employed for sewage sludge treatment.

Subsequently, Cai et al. (2018) prepared a mixture of substrate containing 3:1:1 of sewage sludge, chicken manure and bran to rear the BSFL. The frass from this sample was later tested and was reported to be able to meet the fertilizer standard in China, although no exact value was given. Yet, it was worthy to highlight that the initial heavy metal contents in this mixture were already lower than the concentration set by China fertilizer standard. In addition, it was also stated that the extracted larval oil from G7 contained lesser than 1% of the heavy metal content of its body, making biodiesel production from BSFL a viable approach (Cai et al., 2018).

Next, in the work of Wu et al. (2020), they investigated the fate of Cu and Cd when BSFL was fed with wheat bran that was added with traces of heavy metals. Although Cu is a recognized as a necessary trace element for the proper growth and health of animals, no statistically significant changes on larvae weight was observed when Cu and Cd concentration was increased until 400 mg/kg and 80 mg/kg substrate, respectively. As the heavy metal concentrations in feed increased, both Cu and Cd contents in the larval body increased. Nevertheless, despite the increase of Cu content in the larval body, the content of Cu in its body was remained at an average of 30%, while 70% of the remaining Cu was consistently excreted as feces, also known as frass. The ability for an insect to excrete heavy metals as feces and exuviae had been reported as a method for the insect to regulate its internal metal concentrations (Jiang et al., 2018). In contrast, the Cd content in larval body was at the highest of 90% during low Cd feed concentration. However, as the concentration of Cd in feed increased, the composition of Cd in the larvae body decreased drastically to slightly below 50%. Despite the drastic reduction of Cd content in larvae body itself, the Cd content in larvae body was still exceptionally high (50–90%), as compared to Cu (10–30%). This was aligned with the observation of other works that BSFL had a higher bioaccumulation capacity for Cd than other heavy metals, resulting in a high Cd content in the larval body (Biancarosa et al., 2018; Diener et al., 2015). A subsequent leaching test by Wu et al. (2020) showed that residual form (F5) of Cu and Cd was always the dominant form of heavy metal in the frass for all samples. Nonetheless, the concentration of Cu in residual form seems to reach a plateau after substrate's Cu concentration exceeds 100 mg/kg. When concentration of Cu in substrate exceeds 100 mg/kg, other less stable form of Cu such as exchangeable (F1) and acid soluble (F2) starts to spike drastically.

The fate of Zn and Pb which are two common heavy metals in sewage sludge were also studied. Diener et al. (2015) reported that for a substrate with 2000 mg/kg of Zn, which resembles the concentration in sewage sludge, most Zn content was concentrated in the frass after the BSFL treatment, i.e., 3313 mg/kg. Subsequently, the Zn contents were also found in the larval exuviae at 1883 mg/kg and the larvae body at 866 mg/kg. Similarly, after feeding substrate containing Pb at 125 mg/kg, the highest Pb concentration was recorded in larval exuviae at 312.9 mg/kg, followed by frass at 267.9 mg/kg and finally larvae body at 141.7 mg/kg. These findings validated that the heavy metal content was always the highest in frass rather than larval body. Furthermore, it shows that BSFL can dispose heavy metals during moulting as exuviae, similar to terrestrial insects (Diener et al., 2015).

4.6. Pathogen reduction

The pathogen inhibition ability of BSFL had been studied by Awasthi et al. (2020b) using sewage sludge and several kinds of animal manures (chicken, cow and pig). Depending on the type of animal manures, the reduction of pathogen contents when treated with BSFL had increased by 20–60% as compared with the control without BSFL. The highest reduction of pathogen content was observed for chicken manure, where the reduction percentage increased from ~20% to ~80%. Pig manure showed the least improvement from ~30% to ~55%. Nonetheless, when the sewage sludge was tested, the reduction of pathogen decreased from ~42% to ~15%. This could be due to the increase of pathogen in the order of *Clostridiales*, *Bacillales* and *Lactobacillales* sp. However, the more commonly known pathogen including *Enterococcus faecalis* EnGen0369, *E. coli*, *Bacillus cereus* and *Staphylococcus aureus* M0406 species were all shown to reduce after BSFL treatment.

A similar observation by Liu et al. (2008) reported that BSFL could significantly reduce the *E. coli* content in dairy manure. The study was solely focusing on the reduction of *E. coli* bacteria as the dairy manure was first sterilized before inoculating 7 log cfu/g (colony forming unit/g) of *E. coli* into each sample. Depending on the amount of manure fed, 87–96.7% reduction of *E. coli* could be observed. The highest pathogen reduction of 96.7% was recorded for the sample fed with the highest manure amount while at the same time producing larvae with the highest mean weight. It could be inferred that by increasing the substrate feeding amount, it could increase the growth of BSFL and subsequently, improving its ability in *E. coli* removal. Furthermore, the effect of temperature on BSFL removing *E. coli* at between 23 ^{o}C and 35 ^{o}C was evaluated (Liu et al., 2008). Generally, the pathogen removal ability increased with increasing temperature. Indeed, at 35 ^{o}C, no *E. coli* was observed. However, the control (without BSFL) at 35 ^{o}C showed nearly a 2-fold decrease in *E. coli* content as compared with control at lower temperature. This signified that when the temperature was manipulated, the BSFL was no longer be the only variable that could contribute to the pathogen reduction, but temperature played a significant role in destroying the pathogen as well.

4.7. Co-substrates

Judging by the growth of BSFL, all previous works (Lalander et al., 2019; Cai et al., 2018; Liu et al., 2020b) had shown that the sewage sludge was not a favourable substrate for BSFL. The suitability of sewage sludge as BSFL substrate, however, can be improved with the introduction of co-substrates. By gradually mixing sewage sludge and chicken manure at the ratio of 1:4, the biomass conversion rate increased from 1% to 8% (Cai et al., 2018). As a result, the BSFL weight gained had increased from 0.5 g to approximately 11 g. Next, in the work of Raksasat et al. (2021), the palm kernel expeller was tested and proved viable as a co-substrate due to its high protein and lipid contents. The optimum mixing ratio of sewage sludge and palm kernel expeller was reported to be at 2:3, in which the heaviest larva (46.9 mg) was detected in comparison to 14.38 mg when only sewage sludge was fed. However, further increasing the proportion of palm kernel expeller would contrarily harm the larval growth as excessive protein content would induce the BSFL digestion system to carry out proteinogenic nitrogen detoxification process (Tschirner and Simon, 2015). Hence, while other nutritious organic wastes could be introduced as co-substrates, the optimization of mixing ratio is crucial for the development of BSFL.

Non-conventional organic matter such as biochar has also been tested as co-substrate. Beesigamukama et al. (2020) included 5–20 wt% of biochar into the substrate of brewery spent grain. The biochar used in the experiment was produced from the pyrolysis of rice husks. When the content of biochar was increased from 0 to 20 wt%, the weight of frass increased from 0.49 to 1.13 kg. This was accompanied by an increasing composition of nitrogen content being retained in the frass where the percentage increased from 37.1% to 56.2%. Accordingly, the increment in frass yield was mainly attributed to the lower waste degradation rate because of adding in biochar that was hard to be digested. The increased nitrogen content retained in frass could be due to the adsorption ability of NH_4^+ onto biochar (Agyarko-Mintah et al., 2017; Awasthi et al., 2016). Moreover, the concentration of nitrogen accumulated in the larvae biomass also increased from 4.8 g/100 g biomass to 11.6 g/100 g biomass (Beesigamukama et al., 2020). Not only that, the final larvae weight and waste-to-biomass conversion rate also increased with increasing of biochar amount added.

4.8. Fermentation

The quality of sewage sludge as a substrate could also improve through in-situ yeast fermentation. As shown by Wong et al. (2020), who had inoculated coconut endosperm waste with a commercial yeast species, *Saccharomyces cerevisiae*, observed an improvement in larval biomass conversion rate, growth rate and development time. When the yeast concentration was increased from 0 to 2.5 wt%, the waste-to-biomass conversion rate increased from 10% to 11.5%. Meanwhile, the growth rate increased by 30.76%, while the time needed to form prepupae was reduced from 15.5 to 13.5 days. This could be attributed to the fact that the yeast helped to breakdown the carbohydrates into monosaccharides. Another successful example in improving the substrate quality through in-situ fermentation was proven by using bacteria of the species *Bacillus* spp and *Paenibacillus polymyxa* (Rehman et al., 2019). Dairy manure, despite its high carbohydrate content, could not be served as a good substrate for BSFL due to the presence of high lignin (15.51 wt%), cellulose (26.85 wt%) and hemicellulose (14.71 wt%) contents, making the dairy manure hard for BSFL to digest. However, through the fermentation of dairy manure with the use of bacteria, the hard-to-digest carbohydrates could be degraded into simple carbohydrates thereby enhancing the nutritional value of dairy manure. Among them, the *Bacillus* strain (MRO_2) showed the highest lignin, cellulose, and hemicellulose reductions, and in turn demonstrating the greatest improvement in terms of larval weight and waste-to-biomass conversion rate. As compared with the control, the waste-to-biomass conversion rate was increased from 6.84% to 10.84%, with dry larvae mass increased from 16.35 to 25.94 g, while the larvae survival rate increased to 99.07% (Rehman et al., 2019). Analysis of primary sludge showed that 71.4 wt% of its total solids was made up of fiber-carbohydrate, which could be further breakdown into 32.2 wt% of cellulose, 2.5 wt% of hemicellulose and 13.6 wt% of lignin (Higgins et al., 1982). Hence, fermentation could be a promising method to degrade the complex carbohydrates in sewage sludge and improve its enrichment as a feeding substrate for BSFL.

5. Summary of biological treatment

These three methods of biological treatments of sewage sludge demand less energy consumption. However, the treatment duration required is much longer, ranging from 10 to 60 days, as opposed to the thermal treatment methods that only take a day or less. Nonetheless, the capital cost anticipated to build a vermicomposting and BSFL treatment systems is much lower, making them more feasible for application in developing countries. Without the involvement of high temperature, the nitrogen retention in the treated sludge residue is high, ranging from 68 to 79 wt%. Moreover, all of the biological treatments are capable of reducing pathogen to achieve at least the criteria for US EPA Class B biosolid without any pre-treatment entailed. Meanwhile, for the thermophilic anaerobic digestion, it can easily attain the standard of US EPA Class A biosolid. The mass reduction of sewage sludge happens during biological treatments, mainly through the removal of VS and water content, leaving the non-biodegradable particles to form the residue.

For carbon recovery purpose, the anaerobic digestion produces biogas and vermicomposting increases earthworm population, while BSFL treatment increases BSFL weight. Although the BSFL body lipid can be used to produce biodiesel, the yield is still very low. Furthermore, the anaerobic digestion manages to reduce the bioavailability of heavy metals, and thus stabilizing them. Vermicomposting has shown contrasting findings in terms of heavy metal stabilization, thus needing more works to conclude its performance. Meanwhile, BSFL has shown high survivability growing in heavy metals contaminated substrates. Changes in heavy metal concentrations also take place in the medium as BSFL ingest and store some of the heavy metals, as well as in the larval exuviae. Yet, no comprehensive study has investigated the bioavailability of heavy metals in frass after the BSFL treatment. A summary of the three biological treatments, including the treatment conditions, types of organic wastes, products and performances are presented in Table 4.

6. Challenges and ways forward

Anaerobic digestion of sewage sludge is a mature technology that has been deployed in many developed countries. The main limitation of the wide implementations of this technology is its high capital cost. The integration with MEC that produces more biogas also requires more investigations into identifying a cost-effective electrode for scaling up this promising technology. Although vermicomposting and BSFL valorization both show the potential in treating sewage sludge, there are still several challenges and limitations that need to be overcome as listed below.

a) Sewage sludge is not a suitable feeding substrate due to its low biodegradability and the presence of EPS, which locks all the nutrients within the gel-like structure.
b) The capabilities of vermicomposting and BSFL treatment in destroying pathogen only manage to reach the standard of US EPA Class B biosolid.
c) The capability of heavy metal stabilization is still inconclusive for vermicomposting and BSFL treatment.
d) Existing studies mainly operated in batch experimental mode. The earthworms or BSFL had to be separated manually from the residues once the treatment durations had ended. This is not sustainable for a scale-up process.
e) The earthworms and BSFL are both sensitive toward heat (Singh et al., 2019). The rearing condition should be moist for the optimum growth. Direct placement under a hot sun without shading will cause excessive evaporation of water from substrates. In contrast, the covered equipment placed under a hot sun will trap heat and raise the temperature inside the rearing system, retarding the growth of earthworms or BSFL. Hence, unlike anaerobic digestion or other thermal treatment methods of sewage sludge, vermicomposting and BSFL valorization require suitable growing environments to spur the reduction of substrates.

Common ways forward:

Table 4
Summary of biological treatment methods and their respective performances.

Treatment Method and Condition	Type of Organic Waste	Moisture Content	Generated product	Decomposed residual material	Volume/Weight Reduction	References
Anaerobic Digestion			Biogas	Digestate		
Duration: 4 weeks Temperature: 17 °C	Swine manure	NR	NR	Close to 100% N recovery in digestate. (67.1 wt% in supernatant, 31.5 wt% in settled digestate)	VS reduction: 77.3% TS reduction: 71.4%	(Massé et al., 2007)
Duration: NR Temperature: 30–57 °C	Secondary sludge	After thickening/belt filtration	Biogas composition: CH_4: 53–70% CO_2: 30–50% LHV: 23MJ/Nm^3	NR	VS reduction: 56–65.5%	(Appels et al., 2008)
Vermicomposting			Biomass (worms)	Vermicast		
Duration: 45 days Earthworm: *E. fetida*	Mixture of straw pallet and cattle manure	80–85 wt%	Biomass weight increment: ~80 wt%.	76.3–79.4% of N recovered in vermicast. Carbon content retained in vermicast: 54.5–56.5 wt%	NR	(Nigussie et al., 2016)
Duration: 45 days Earthworm: *E. fetida / E. eugeniae / P. Excavatus* or combination	Sewage sludge	50 wt%	Biomass weight increment: 9.09–28.57 wt%	TN concentration increased after vermicomposting.	TOC reduction: 10–25% in DM	(Khwairakpam and Bhargava, 2009)
BSFL Treatment (*Hermetia illucens)*			Biomass (BSFL), Biodiesel	Frass		
Duration: 32 – 51 days	Primary sludge, secondary sludge	82.5–91.7 wt%	Biomass conversion rate: 2.2–2.3% in DM Protein conversion rate: 7.8–15% in DM	NR	Weight reduction: 49.2–63.2 wt% in DM	(Lalander et al., 2019)
Duration: 24.3 days	Dairy manure	78.4 wt%	Biomass conversion rate: 6.3% in DM	68% of total nitrogen content retained in the residual.	Weight reduction: 25.8 wt% in DM 63.2 wt% of wet weight TOC reduction: 36% in DM	(Rehman et al., 2017)
Duration: 10 days	Animal manure (Cattle, pig, chicken)	65–70 wt%	Biomass Yield: 12.8–32.8 wt% Lipid Yield: 29.1–30.1 wt% Biodiesel/lipid yield: 93–96%	NR	NR	(Li et al., 2011)

NR: Not reported; DM: Dry mass

a) The effect of sewage sludge pre-treatment prior to vermicomposting and BSFL treatment should be examined. Breakdown of EPS releases more soluble nutrients into the substrate, and subsequently boost the growths of earthworms and BSFL (Dohányos et al., 2004; Ruffino et al., 2015). Furthermore, certain pre-treatments such as thermal treatment and chemical treatment are also expected to assist in pathogen reduction.
b) Continual research and optimization of blended substrates using the locally available co-substrates are the low hanging fruits that can boost the growth of both earthworms and BSFL.

Specific challenges and ways forward for BSFL valorization:

a) Current yield of biodiesel is still very low due to the low conversion rate of organic wastes into BSFL biomass. This is a limitation before BSFL treatment can be popularized for the purposes of carbon recovery and biodiesel production. The biodegradability of substrates has to be improved for attaining a better growth of BSFL. Thus, pre-treatment such as thermal treatment, enzymatic treatment, fermentation, hydrolysis, etc. should be investigated.
b) BSFL feeding on sewage sludge have a very low body weight as opposed to those feeding substrates which are more biodegradable such as fruit waste and kitchen waste. A light prepupa will later convert into a small adult fly that has less body fat. This could affect the lifespan of the BSF in terms of whether it has a sufficient stored energy to complete one whole lifecycle by mating and ovipositing thereafter. Without sufficient healthy BSF eggs, the entire sludge treating process is not sustainable and has to depend on the procurement of new BSF eggs. Thus, the sustainability of treating sewage sludge via BSFL employment needs more inclusive investigations.
c) Unconventional co-substrates such as biochar, fly ash and phosphatic rock have shown synergistic effects on stabilizing heavy metals and improving the overall performances of anaerobic digestion, thermal treatment and vermicomposting (Liew et al., 2021; Babel et al., 2009; Zhang and Wang, 2020; Suthar, 2010). Similar additives can also be vindicated in BSFL valorization.

7. Conclusions

This paper reviews extensively the performances of three biological treatment methods of sewage sludge, comprising anaerobic digestion, vermicomposting and BSFL valorization. Anaerobic digestion of sewage sludge is a mature technology that has seen numerous industrial applications. Vermicomposting of sewage sludge has also been conducted in full-scale but together with co-substrate. Meanwhile, BSFL valorization of sewage sludge is only tested in lab-scale. All these three treatment methods have high nitrogen recovery capabilities. The sewage sludge mass reductions are comparable with one another, but depending on the treatment durations in which could range between 30% and 63%. Anaerobic digestion is superior in carbon recovery, as it can generate biogas. BSFL treatment could potentially generate biodiesel if the limitation of low larval lipid yield is resolved. The residue from all three treatments can easily achieve the pathogen standard of US EPA Class B biosolid. Yet, it can be improved to Class A biosolid via thermal pretreatment of sewage sludge at 65 °C for an hour. Effective heavy metals stabilization had been reported for anaerobic digestion, but no conclusive finding has substantiated the effectiveness of either vermicomposting or BSFL treatment. Thus far, a comparison with anaerobic digestion shows that both vermicomposting and BSFL treatment are the potential low-cost sewage sludge treatment methods to be deployed in developing countries. Nonetheless, until the listed challenges have been addressed, more innovative solutions have to be assessed and validated prior to the executions.

CRediT authorship contribution statement

Chin Seng Liew: Conceptualization, Writing – original draft, Writing – review & editing. **Normawati M. Yunus:** Writing – review & editing. **Boredi Silas Chidi:** Writing – review & editing. **Man Kee Lam:** Writing – review & editing. **Pei Sean Goh:** Writing – review & editing. **Mardawani Mohamad:** Writing – review & editing. **Jin Chung Sin:** Writing – review & editing. **Sze Mun Lam:** Writing – review & editing. **Jun Wei Lim:** Conceptualization, Supervision, Writing – review & editing, Validation, Funding acquisition. **Su Shiung Lam:** Conceptualization, Supervision, Writing – review & editing, Validation, Funding acquisition.

Declaration of Competing Interest

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

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