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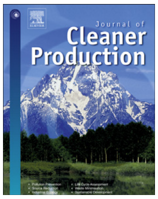

Valorization of food waste into biofertiliser and its field application

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ABSTRACT

Worldwide significant amounts of food waste are generated daily causing serious environmental issues, occupying land and requiring expenditure of resources for its treatment. A smart method for handling this food waste problem is the development of novel processes targeting the conversion of this waste into value added products. Although valorization of food waste to biofuels, biochemicals and biopolymers have been widely investigated, the utilization of food waste streams into biofertiliser has not been intensively reviewed. Conversion of food waste, especially agriculture residues into biofertiliser would reduce its environmental impact, improve nutrition levels of the soil, decrease requirements for synthetic chemical fertiliser and have a direct benefit on food production. This paper reviews recent progress in the field regarding the production of biofertiliser from food waste, using anaerobic digestion, aerobic composting, chemical hydrolysis, *in situ* degradation and direct burning methods. This review also highlights the latest field applications of biofertiliser derived from various food waste streams. It confirms that the technology for the conversion of food waste to biofertilisers is viable, but the production efficiency could be improved with better process control strategies, strict quality controls, development of a smart product distribution system and adoption of advanced technologies. Field tests have indicated that biofertilisers which are obtained in proper managed AD plants are safe and could partially replace the use of chemical fertilisers in field application.

1. Introduction

Food waste can be defined as the outlets of the food production industry, which are not currently used for defined end-products, not recycled or used in an alternative manner. These products have a lower economic value than the cost of collection or reuse in a traditional food production stream. There has been a growing concern over the generation and suitable treatment of food waste. In Europe and North-America, around 95–115 kg food products are wasted per capita per year (Gustavsson et al., 2011), the majority of this waste ends up as municipal solid waste (MSW). According to a FAO report, around 1.3 × 10^9 t of food is lost or wasted globally per annum (Gustavsson et al., 2011), which equated to approximately 30% of the weight of global crude oil output in 2011. Besides wasted food products, waste streams generated in industrial food processing systems and agriculture residues from plant cultivation also contribute significantly to food waste. Food processing waste can be defined as food material destined for consumption which is either lost or discarded during the production, distribution and consumption of the food. It has been estimated that up to 50% of food is lost during food production (Hall et al., 2009). Particularly waste generated by the animal, poultry and fishing industries is extremely heterogeneous and potentially contains pathogens, making use of these waste materials challenging. Agricultural residues are waste streams produced by crops cultivation activities in the food production chains. In China, around 580 Mt of straw is generated annually (Wang et al., 2010). Only 2–5% of this straw is utilized with the majority being burnt (Shi et al., 1996). The smoke generated from the burning of straw has become one of the key contributors to air pollution in China. Food waste has been shown to cause serious environmental issues, such as generating greenhouse gases and occupying land resources.

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With the increasing awareness of the problems associated with food waste, research on the conversion of food waste into biofuel, biochemical and biopolymers has received growing attention (Lin et al., 2013). In comparison with biofuel and biochemical production, the importance of generating biofertiliser from food waste has been under estimated. Fertiliser has a high market prospect with an estimated value by 2020 of over $\$150 \times 10^9$ per annum (Research Market, 2017). Replacing synthetic chemical fertiliser with biofertiliser derived from food waste would reduce the requirement for synthetic fertilisers, reducing the environmental impact of food waste and directly benefiting food production.

Fig. 1 illustrates the main processes that have been developed for the conversion of food waste streams into biofertiliser. Wasted food (including OFMSW, Organic Fraction of Municipal Solid Waste) and food processing waste contain high concentrations of carbohydrate, protein and/or fat with high moisture contents (Kiran et al., 2014). Presence of these compounds has been highlighted for their suitability for treatment via processes such as anaerobic digestion (AD), aerobic composting and chemical hydrolysis (Arshadi et al., 2016; Francavilla et al., 2016). Although biogas is the main product of AD, the co-production of digestate as biofertiliser is an important strategy to bring in additional income. By contrast, agricultural residues are generated during crops cultivation activities. They are rich in lignocellulose material, and have a relatively low moisture content (e.g. 10–15% for air dried wheat straw Pensupa et al., 2013). Although agriculture residues have a low economic value, they are an important renewable carbon and mineral resource for the soil. Agricultural residues can be degraded either *in situ* or collected, taken off farm and converted into biofertiliser before being added back to soil. Returning agriculture residues as a biofertiliser in a correct manner has been shown to improve the organic content of the soil, modify the soil particle structure, reduce water evaporation, improve niche microorganism activities and decrease fertiliser loss (Jordan et al., 2010).

This review summarizes recent research into the valorization of food waste, including wasted food, food processing waste and agriculture residues, into biofertiliser. It also highlights field trials for the use of food waste derived biofertiliser for the improvement of food production as well as a better understanding of the mechanism of addition of biofertiliser on plant cultivation.

2. Methods

The aim of this review is to provide a detailed overview of biofertiliser production from various food waste streams. A summary of biofertilisers produced from wasted food, food processing waste and agriculture residues is presented. This summary includes the production process, nutritional, quality control and the impact of field application of biofertiliser(s). The literature includes papers, scientific reports and presentations that have been obtained from scientific journals and online resources. Due to the high numbers of published papers, only papers published in the past 10 years were considered. Older articles were only cited to support discussion or provide extra examples.

3. Biofertiliser generations via anaerobic digestion

Anaerobic digestion (AD) is a natural organic matter degradation process that occurs in environments such as swamps, bottom of lakes and intestines of animals. It has been successfully used to treat sewage sludge for over a century. AD has been expanded for the treatment of organic waste, municipal solid waste and food processing waste. In 1990, the capacity of AD plants in Europe was 120,000 t, which had increased to nearly 9 Mt by 2015 (European Bioplastics, 2017). In the UK, the number of the non-water treatment based industrial scale AD plants has increased from 74 (2012) to 108 (2013), with a further 169 projects planned in 2013 (Hindle, 2013). As of March 2016, 104 out of 254 operational AD plants were using food waste as the main feedstock (Warp, 2017). In China, 38.5×10^6 household-scale anaerobic digesters were built by 2010

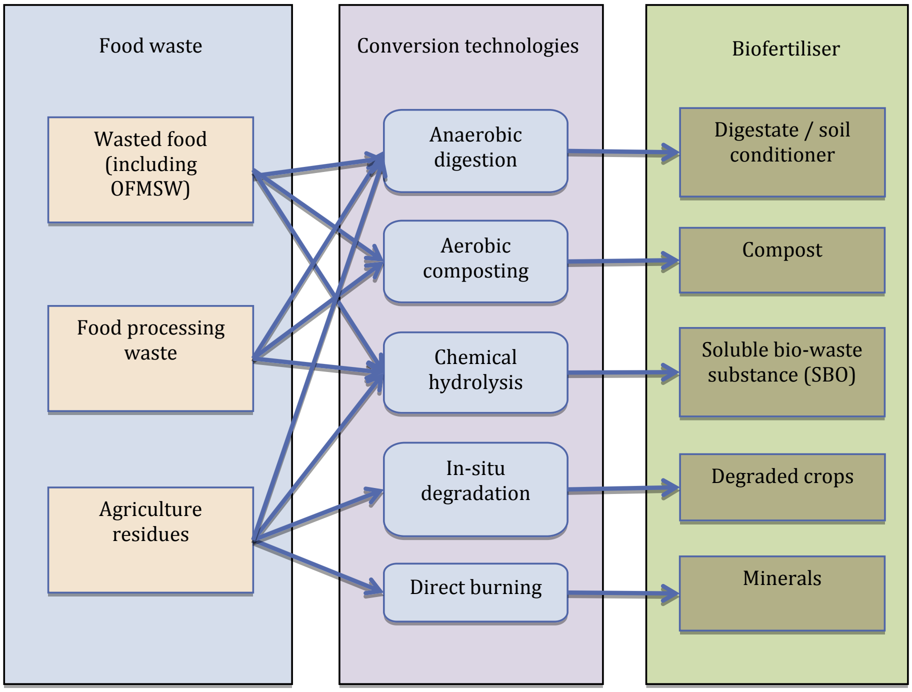

Fig. 1. Schematic diagram of the main technologies currently used for the conversion of food waste into biofertiliser. In this paper, ***Food waste*** includes wasted food, food processing waste and agriculture residues. ***Wasted food*** represents food wasted from household, restaurant and supermarket, including ***OFMSW*** (Organic Fraction of Municipal Solid Waste); ***Food processing waste***: food waste stream generated during food production process; ***Agriculture residues***: biomass waste generated related to agriculture activity, e.g. wheat straw. ***Biofertiliser***: a product generated from organic biomass that has a similar nutritional value as synthetic chemical fertiliser.

with an estimated annual biogas output to be 13.1 × 10^9 m^3 (Chen et al., 2012).

The primary objective of AD plants is the treatment of waste streams and the generation of biogas as a type of energy. The solid residue of AD plants could be further processed to biofertiliser, compost or soil conditioner as an additional income stream (Fuchs et al., 2010). In some sites, the liquid fraction (liquor) following AD can also be used as a biofertiliser (Tampio et al., 2016a). Fig. 2 shows a schematic diagram of the process in which heat, power and biofertiliser are simultaneously produced from food waste. Following AD, the digestate requires dewatering, ammonia control and sanitization before it can be applied as a solid biofertiliser, compost or soil conditioner.

Various food waste streams including fruit and vegetable waste, potato and starch processing waste, sugar processing waste, diary waste effluent, animal processing waste, crop residue and OFMSW have already been used as feedstock for AD at a commercial scale, as highlighted in Table 1. Table 2 lists the analyses of the nutritional value of biofertiliser derived from AD using various food waste streams.

The typical total nitrogen (N), phosphorous (P) and potassium (K) content in the digestate (before drying) are in the ranges of 1.5–6.2, 0.2–2.6 and 1.2–11.5 g/kg (Frischmann, 2012; Moller and Muller, 2012, Table 2). These values vary significantly due to the type of food waste used and whether a high nitrogen waste stream was co-digested. The nutrient contents (mainly N, P and K) presented in the digestate all originate from the feedstock. Due the high-water content of food waste and unbalanced nutrient composition, food waste is commonly co-digested with farm slurry or manures or green waste to adjust the carbon to nitrogen ratio (C/N) to a suitable range, which improves efficiency of AD (Yin et al., 2016). This practice is particularly important for biofertiliser application, as the high nitrogen content of the slurry and manures would promote the nutritional value of the biofertiliser of the food waste digestate. However, the nitrogen element should not be in the form of ammonia, as high ammonia nitrogen content may be detrimental for biofertiliser application due to its potential environmental impact.

A study published in 2010 by Wales Centre of Excellence for Anaerobic Digestion analyzed the chemical composition of food waste collected from 18 different locations in Wales (Esteves and Devlin, 2010). The average total solid content, carbohydrate, lipid and protein contents were 24.2 ± 0.4%, 93.3 ± 18.4 g/kg, 48.8 ± 10.8 g/kg and 77.2 ± 27.6 g/kg during the summer season; and were 27.7 ± 0.3%, 156.0 ± 20.1 g/kg, 59.3 ± 10.1 g/kg and 44.3 ± 14.0 g/kg during the winter season. The nutrient analysis results indicated the total nitrogen (Kjeldahl), and phosphorus contents were 6.89 g/kg and 0.83 g/kg for the summer season and 7.32 g/kg and 0.9 g/kg for the winter season. Significant variation was observed between each local authority, e.g. the nitrogen content ranged from 2.79 to 11.12 g/kg in the summer. In a similar study, where food waste was collected in San Francisco, California, US, the total solid content, nitrogen and phosphorus contents were 30.9 ± 0.1%, 31.6 ± 2.2 g/kg and 5.2 ± 0.8 g/kg (Zhang et al., 2007). Rigby and Smith (2013) estimated that the total nitrogen and total phosphorus contents in digested fibre from food waste AD digestate were 9.6 and 0.34% (dry matter). The total nitrogen content was higher than the average nitrogen contents reported by Zhang et al. (2007). This may due to the nature of food waste composition variation and a high nitrogen feedstock was included in this study

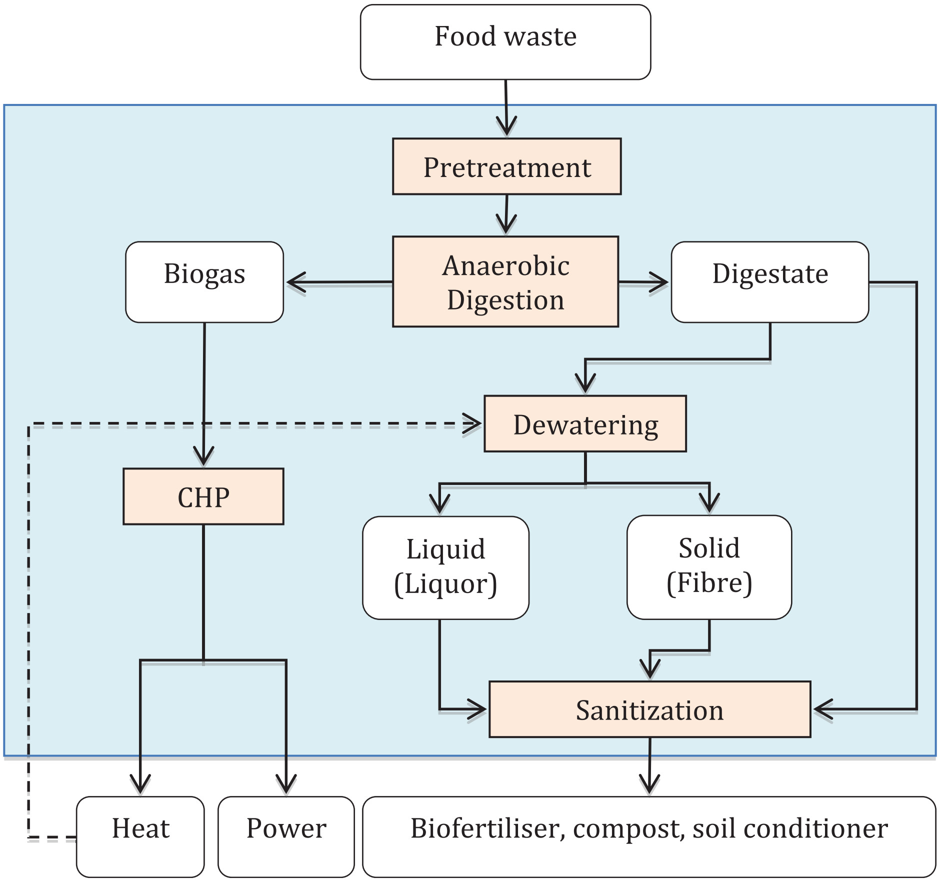

Fig. 2. Schematic diagram of an AD process that co-produces heat, power and biofertiliser.

Table 1
Some AD plants using food waste as sole feedstock or co-feedstock in the UK (Adapted from http://www.wrap.org.uk 2016 and company websites).

Plant name	Feedstock	Year of operation	Capacity (t/annum)	Energy output (KW)	Biofertiliser production
AC Shropshire Ltd	Pig Slurry and Food waste	2013	86,000	2000	Liquid biofertiliser is produced
Basingstoke (Tamar Energy)	Agricultural slurries and food waste. Input: Food waste (C&I 30ktpa max and/or municipal 30ktpa max) and agricultural slurries	2014	40,000	1500	Compost is used by Kenilworth Castle, the Royal Festival Hall, Glyndebourne Shoot, landscapers, gardeners. Biofertiliser is being used by Worth Farms.
Bio Dynamic (UK) Ltd	Municipal food waste and agricultural waste	2014	150,000	4000	Biofertiliser production is introduced in the website
Bore Hill Farm Biodigester	Commercial & Industrial food waste, Category 3 Animal By-Products, local 'Direct to AD' collection scheme.	2012	20,000	1060	Biofertiliser achieved PAS110 accrediation
Cannington Cold Stores Ltd	Yogurt waste, fruit juice, silage, manufactured spreads/dressings (household separated food waste from 2011)	2009	100,000	1300	Compost production is introduced in the website
Green Tye (Guy & Wright Ltd)	Tomato and fruit & vegetable waste (wholesale rejects)	2009	10,000	500	Spent liquid used as biofertiliser for wheat crops
Greenville Energy	Grass silage cattle slurry dairy waste and waste fruit and vegetables	2012	25,000	500	Biofertiliser is produced in the website
Holbeach (Tamar Energy)	Waste potatoes and other organic material including maize	2013	36,000	1500	Same company as Basingstoke (Tamar Energy)
McCain Foods	Waste water rich in potato starch	2010	950,000	1063	N/A
ReFood (PDM Group Ltd) - Doncaster	Commercial food waste	2011	160,000	5000	Liquid biofertiliser is used by local farm
Rose Hill Farm	Food waste, energy crops and animal manure	2014	35,000	1000	N/A
Scottish and Southern Energy (SSE)	Food waste & Organic material (industries such as agriculture, food production, food retail and alcohol production)	2011	75,000	2200	Biofertiliser production is introduced in the website
Barkip Biogas	Vegetable waste (small proportion of cattle slurry too)	2010	35,000	1100	Biofertiliser achieved PAS110 accrediation
Thornton Waste Technology Park (Global Renewables)	Municipal solid waste	2010	105,000	1900	N/A
Viridor Waste Management Ltd - Newton Heath	Food waste - municipal	2011	100,000	2000	Soil conditioner and compost products are produced.

Table 2
Nutritional value of fertiliser/compost obtained from AD processes using food waste (dry matter basis).

Feedstock	AD process	Total-N	NH_4-N	Total-P/Total-K	Application	Reference
Energy crop and pig slurry	Co-digestion	4.97 kg/m^3	2.64 kg/m^3	NA	Field test in Germany	Koster et al., 2014
Food and farm wastes	Co-digestion	2.32–4.64%	2800-52500 mg/kg	3.81–28 and 1.94–37.6 g/kg	Field test in UK	Rigby and Smith, 2013.
MSW	Batch, mesophilic	1.5%	NA	0.314% (P)	Soil enhancer	Walker et al., 2012
MSW	Batch, mesophilic and thermophilic	NA	NA	NA	Lab-pilot scale	Tampio et al., 2016b.
Olive waste and citrus pulp	Batch, mesophilic	6.0%	149 mg/L	840 and 631 mg/L	Lab scale germination study	Panuccio et al., 2016.
Ryegrass/sugar beet	Batch, mesophilic	6.2%	1.5 g/L	0.32 and 3.6 g/L	Lab scale pot tests	Gunnarsson et al., 2010
Straw	Co-digestion, mesophilic	3.1–14%	1.58–6.1%	0.4–2.6 and 1.2–11.5 kg/Mg	Field test in Germany	Moller and Stinner, 2009.
Triticale/cow manure	80 d AD, 180 d composting	2.9%	8.43 g/kg	0.119 g/kg (P)	NA	Pivato et al., 2016
Winter wheat/potatoes	Co-digestion, mesophilic	0.25% (FM)	0.16% (FM)	0.62 (P)	Field test in Germany	Stinner et al., 2008.
Food waste and human excreta	60 days at mesophilic temperature	2.4% (dm)	3.49%			Owamah et al., 2014

(FM: fresh matter; NA: Not available; P: Phosphorus; K: Potassium).

(Rigby and Smith, 2013).

In order to control the quality of the biofertiliser, contamination of unwanted materials including physical impurities (e.g. plastics, glass), chemical impurities (e.g. heavy metals) and biological impurities (e.g. animal pathogens, plant pathogens) should be prevented or removed. Information on the pre-digest feedstock selection, analysis and recording is required, such as the origin of the food waste, the location where the food waste stream was generated or collected and the main contents present in the waste stream. This is of particular important for AD plants when using OFMSW, as the composition of OFMSW varies significantly (Esteves and Devlin, 2010). Pretreatment, such as physical, chemical, biological and physical-chemical hydrolysis is often applied to certain types of food waste streams in the AD process (Zhang et al., 2014). Pretreatment reduces the particle size, screens out physical impurities (Al Seadi and Lukehurst, 2012), destroys potential pathogenic microorganisms (Evans et al., 2007) and speeds up the consequent AD process (Ma et al., 2017). Zhang et al. (2014) reviewed various pretreatment methods used in AD of food waste with the aim to improve biogas production. As biofertiliser is a co-product in AD process, an improved efficiency of AD would benefit biofertiliser production as well.

Heavy metal content is one of the major concerns of biofertiliser safety. The heavy metals that are commonly detected in food waste, especially MSW are cadmium (Cd), chromium (Cr), copper (Cu), lead (Pb), mercury (Hg), nickel (Ni), and zinc (Zn) (Abdullah et al., 2016). Utilization of a biofertiliser with a high metal content may lead to the contamination of arable land; with the potential for metals to accumulate in plant roots, remain in the soil, and pollute ground water. Food processing waste and proper source separated OFMSW normally contains lower concentrations of heavy metals (Govasmark et al., 2011). Several studies characterising biofertilisers derived from food waste have been carried out in the UK (Rigby and Smith, 2013), indicating that heavy metal contents were below the relevant standards. However, presence of heavy metals in biofertiliser must be managed in order to meet the increasingly strict environmental protection regulations.

Food waste feedstock may contain organic contaminants, such as PAHs (Polycyclic Aromatic Hydrocarbons), PCBs (polychlorinated biphenyls), DEPH (Di (2-ethylhexyl) phthalate), LAS (Linear Alkyl benzene Sulphonates), PCDD/F (dibenzo-p-dioxins and -furans), DL-PCB (dioxin-like polychlorinated biphenyls), pesticides and pathogen strains (Brandli et al., 2007a, 2007b; Al Seadi and Lukehurst, 2012; Benisek et al., 2015). Similar to heavy metals, these contaminants need to be destroyed or removed before a biofertiliser could be applied to arable land. Although the use of AD process leads to a reduction of the presence of pathogenic strains (Van Overbeek and Runia, 2011), research has revealed that the reduction in organic contaminants or pharmaceutical residues has been generally less effective via AD (Stasinakis, 2012; Davidsson et al., 2014). Therefore, applying pre-digestion source separation methods to prevent these organic contaminants entering the AD system is crucial.

4. Biofertiliser production via aerobic composting process

Composting is a typical aerobic digestion process, which converts organic matter into compost, a humus-rich, earth-like product. It has been long associated with the treatment of green waste from farm and garden. With the increasing limitations in landfill capacity, food waste which previously was earmarked for landfill, is used for aerobic composting (Chen et al., 2017). It has been estimated that food waste contributes to around 1/3 of the compost produced in the EU, with the remaining compost obtained from farm slurry/manure, sewage sludge and energy crops (Cesero et al., 2015). The advantages of composting have been well documented, which include: generation of a fertiliser, soil conditioner like product; reduction of waste volume; reduction in presence of pathogens, control germination of weeds in agricultural fields; and elimination of undesirable odorous compounds (Farrell and Jones, 2009; Li et al., 2013). In composting of food waste, microorganisms, including bacteria, fungi, mould and actinomycetes use the organic components in food waste degrading them into short chain chemicals, e.g. humic acid. Compost can be carried out in vessel, aerobic windrow or aerobic pile (Lim et al., 2016; Pandey et al., 2016). During composting process, the temperature often raises to a high level as microbes release heat (55 °C for 5–7 d or 75 °C for 2–3 d). This increase in temperature is responsible for the deactivation of pathogens and weed seeds (Miyatake and Iwabuchi, 2005).

In comparison with green waste, food processing waste normally contains high water content (e.g. 80% for citrus peel residue) (Lin et al., 2013) and an imbalance in nutrient (Kiran et al., 2014). Therefore, food processing wastes are commonly composted with green waste or bulking agents, such as sawdust, rice husk, wood chip and wheat straw to adjust to a suitable C/N ratio and to reduce the moisture content (Adhikari et al., 2009; Chang and Chen, 2010). For food processing waste, such as olive mill waste, which contains a low nitrogen content, co-composting with farm slurry/manure improves the nutritional value of the compost (Fernandez-Hernandez et al., 2014).

Table 3 lists the composition analysis of compost which is derived from food waste via aerobic composting process. The typical dry matter, total nitrogen (N), ammonia nitrogen (NH_4-N)

Table 3
Nutritional value of compost obtained from aerobic composting processes using food waste (dry matter basis).

Feedstock	Compost process	Dry material	Total N	NH_4-N	Total P	Electrical conductivity	C/N ratio	Reference
Corn stalk and pig manure	37 d, 30–75 °C	NA	2.0–2.8%	0.5–3.0 g/kg	NA	NA	10.8–16.2	Guo et al., 2012
Fruit, vegetable waste and yard wastes	15 weeks, 25–41 °C	40%	2.0%	NA	NA	4.9–9 dS/m	11.1–19.8	Sangamithirai et al., 2015
Olive mill waste and sheep/horse manure	30 weeks, 20–70 °C	86-89%	1.47–1.73%	NA	0.3–0.4%	NA	15.6–19.2	Fernandez-Hernandez et al., 2014
Olive oil husk and manure	116 d, up to 65 °C	68-82%	1.4–2.5%	NA	0.67–0.71%	1.45–7.3 dS/m	14.3–27.9	Montemurro et al., 2009.
Palm oil mill waste	35 d	NA	NA	NA	NA	1.5–4.0 dS/m	18–22	Mohammad et al., 2015
Restaurant food waste and rice bran	30 d, 30–75 °C	35-45%	NA	NA	NA	NA	15–21	Wang et al., 2017
Spent coffe grounds, spent tea leaves with yard wastes	15 weeks, 25–44 °C	NA	0.3–3%	NA	NA	6.7–7.2 dS/m	9.1–16.9	Sangamithirai et al., 2015
Sugar mill waste, green waste and farm manure	90 d ambient temperature in Pakistan	NA	2.5%	NA	NA	NA	18.32	Sohail et al., 2014
Waste coffee pulp, coffee husk	NA	NA	2.99%	NA	NA	NA	7.25	Preethu et al., 2007
Wasted food	30-33 d	46.3–51.6%	1.73–1.84%	1.3 g/kg	NA	NA	20.2–23.6	Sundberg et al., 2013
Wasted food and rice husk	32 to 130 d, 25–71 °C	~51%	1.6–2.6%	<0.1 g/kg	NA	NA	14.9–29	Chikae et al., 2006
Wasted food and saw dust	35 d, 35- 55 °C	NA	NA	1.6–6.0 g/kg	NA	NA	24.6–30.1	Wang et al., 2016
Wasted food and saw dust	9-15 d, 30–55 °C	NA	NA	NA	NA	NA	27.8–42.3	Chang and Hsu 2008
Wasted food and saw dust	60 d, 30–65 °C	NA	1.6%	NA	0.6%	NA	20	Lin 2008
Wasted food, saw dust, rice husk and rice bran	6-15 d, 30–60 °C	44-51%	0.87–1.59%	NA	NA	NA	32.7–51.5	Chang and Chen 2010

(NA: Not available).

and phosphorus (P) content in the compost are in the ranges of 44–52%, 0.9–3.0%, 0.5–6.0 g/kg and 0.3–0.7% (Table 3). Properly dried compost could have a moisture content as low as 11–14%, leading to a dry matter of nearly 90% (Fernandez-Hernandez et al., 2014). The C/N content is normally in the range of 15–30, depending on mainly the feedstock used and the length of the compost. Increase in pH and reduction of electrical conductivity were normally observed during food waste composting (Zhang and Sun, 2016), which are corresponding to the removal of volatile organic acid and the removal of salts.

5. Biofertiliser production via chemical hydrolysis

Chemical hydrolysis of food waste, especially food waste derived digestate and compost is a new alternative approach for the generation of a product that can be used a biofertiliser. In this process, organic waste is treated via alkaline or acid hydrolysis at moderate temperature of 60–100 °C (Rosso et al., 2015; Arshadi et al., 2016), resulting in a soluble bio-waste substance (SBO). Then the SBO is dried to form a solid product with a moisture content of around 10% (Sortino et al., 2013). The utilisation of microwave to replace conventional heating process reduced the reaction time by a magnitude of 1 or 2 orders (Rosso et al., 2015).

The composition analysis revealed the SBO contains mainly soluble lignin-like polymers and soluble saccharide polymers. Sortino et al., report a SBO obtained from alkaline hydrolysis of a compost contains 5.1% total nitrogen (N), 0.37% phosphorus (P) and 1.2% potassium (K) (w/w, db). The total nitrogen content of SBO was higher than that of compost (Table 3) and the phosphorus content was similar to that of compost, indicating it could be a high quality biofertiliser. The addition of SBO at a low dose of just 140 kg/ha significantly increased growth and productivity of red pepper (Sortino et al., 2013). The application of SBO as a biofertiliser also promoted plant growth and reduced plant disease for beans (Baglieri et al., 2014) and radish (Monterumici et al., 2015).

6. Direct returning agriculture residues back to soil as biofertiliser

Agriculture residues, such as wheat straw, rice straw and sugar cane bagasse, are typical waste streams generated in food production supply chains. Crop residues have been intensively investigated for the production of bioethanol (Mafe et al., 2014). Incorporating straw into the soil as a biofertiliser is a typical agriculture practice in many regions. A recent farm survey in the UK indicated that around 36% of cereal straw is returned back to the field (Glithero et al., 2013). In USA, the ratio of the straw returned back to the soil remains at about 68% (Yong et al., 2001), as the United States Department of Agriculture believes that the degraded straw plays an important role in soil fertility. In China, the annual straw production is approximately 580 Mt, which accounts for 20–30% of the total global straw production (Wang et al., 2010). The Chinese government aims to increase the percentage of straw utilization to over 85% by 2020 using a variety of approaches, including the incorporation of straw into the soil (Chinese Government, 2017).

Fig. 3. A photo shows the activity of returning straw back to soil by a tractor. The photo was taken in Meiqiao, BengBu, Anhui Province, 15/06/2016. The trial field area was 0.8 ha in total.

The direct return of straw back to soil process is simple and straightforward process, with direct return meaning ploughing of the crop residue back into soil after crop harvest and pulverization of straw (Fig. 3). The aim of *in situ* straw degradation is to release the nutrients during further decomposition of the straw by soil microorganisms (Gong et al., 2008). To accelerate straw degradation, external supplementation of soil microorganisms (such as *Azospirillum* sp, *Bacillus* sp), nutrients (such as sugar) or a combination of both is often applied (Borah et al., 2016). Soil microbes transform the organic constituents present in straw(s) into short chain organic compounds, which are released into the soil and used by plants. Nitrogen, phosphorus, potassium and other nutrients are transported back to the soil directly, or stored in soil microbes, these stores act as efficient long-term nutrient sources (Nie et al., 2007).

In European and North American countries, *in situ* degradation of straw mainly relies on the activities of endogenous soil microorganisms (Hong et al., 2016; Sun et al., 2016). Nevertheless, the addition of microorganisms is essential for *in situ* degradation in countries, which have huge populations but have reduced amounts of suitable farmland. A high percentage of arable land practices crop rotation, plus the issue of fertiliser overuse leads to an over production of straws which exceeds the ability of endogenous soil microbes to degrade all the waste efficiently (Zhao et al., 2016). Therefore, it is important to enhance the degrading capability of microorganisms by increasing the absolute number of microorganisms and by improving the degradation power of these microorganisms.

Soil microorganisms have relative strong degrading power when exposed to cellulose and hemicellulose. Under aerobic and mesophilic conditions, fungi play the primary role for the degradation of cellulosic and hemicellulosic. *Trichoderma, Penicillium, Aspergillus* and *Fusarium* are the common fungal species in soil that are responsible for cellulose and hemicellulose degradation (Eida et al., 2011; Karpe et al., 2015). Bacterial species, such as *Cytophaga, Sporocytophaga* and *Polyangium* also degrade cellulose and hemicellulose (Hyun et al., 2009; Li et al., 2011; Wang et al., 2012). In comparison with fungi, bacteria show less species diversity, but dominate the absolute quantity in soil. Cellulose and hemicellulose degrading bacteria can reach 10,000 strains per g dry soil. Soil microorganisms are relatively poor when it comes to decomposing lignin, and bacteria are generally less competent than fungi in lignin degradation. Furthermore, bacteria can only degrade lignin under aerobic conditions (Moller et al., 1999). Odier et al. (1981) examined the potential of *Xanthomonas, Pseudomonas* and *Acinetobacter* for the degradation of lignin but found 20–40% lignin was degraded after a 7-d cultivation. In order to further improve straw degradation, synergistic reactions in a multi-enzyme system, with a microorganism consortium are preferred. The appropriate combination of bacteria and fungi usually results in an efficient degradation (Zhao et al., 2000). A case study revealed that the synergistic degradation of rice straw between cellulosic and lignin-degrading microbe was more effective than that carried out by any individual strain (Zhao et al., 2000). Table 4 lists several reports regarding the impact of *in-situ* degradation on soil nutritional properties.

In situ degradation of straw primarily depends on the species, quantity and activity of soil microorganisms, thus a suitable environment for microbial growth and reaction is crucial. The key impact factors are discussed below.

(a) The C/N ratio is a key parameter which influences degradation rate and the decaying level.
(b) Soil water content is another crucial factor, which influences degradation efficiency. Straw degradation demands water, especially during the cellulose and hemicellulose hydrolysis stage. Hydrolysis commonly occurs at the preliminary stage of the biomass degradation process; hence addition of water to a range of 20–23% speeds up degradation (Zuo and Jia, 2004). The water requirement is reduced at later stages (e.g. after 30 d), but a soil moisture content of 16–20% is still required for microbial growth (Jiang et al., 2001).
(c) The depth of straws incorporation can affect the degradation efficiency as well due to microbial distribution and soil air permeability. The straw degrading rate on the ground surface is much slower than that for straws buried in soil. As soil microorganisms are mainly present at a depth of 0–10 cm in soil, straw at the surface has limited access to microorganisms (Ma et al., 1999). An optimum depth for *in situ* straw degradation is 10–25 cm deep from the surface. Further down there is a reduction in air permeability, thus reducing the decay rate. Soil pH, soil temperature, straw particle size, operation date and straw loading rate all directly or indirectly affect the degrading ability of microorganisms (Henriksen and Breland, 2002).

7. Biofertiliser derived from direct burning of crops residue

Direct burning is the most ancient way for directing some nutrient values of straw to soil. Direct burning of straw in the field transforms almost all organic matter into gaseous oxides and exhausts into the atmosphere. A small number of mineral elements, such as potassium exists in ash, which is then used as fertiliser. Although burning is convenient and fast, the benefits in terms of nutrient enrichment of the soil is limited. On the contrary, this treatment method causes soil erosion, air pollution and soil organic matter loss. Recent studies suggests that straw burning leads to a 65–80% loss in soil moisture and a 0.2–0.3% decrease in the organic matter content of soil each time (Rossi et al., 2016; Ventrella et al., 2016). It has been estimated that it would require 5–10 y to compensate for the organic matter loss if the organic matter was replenished by the natural straw degradation process only. The high temperature in the burning process destroys niche microbiology ecosystems by killing most of microorganisms, leading to increased opportunity for soil diseases. This procedure is unsustainable and is restricted in many developed countries, e.g. UK (UK government, 2017).

8. Impact of biofertiliser on crop/vegetable cultivation

8.1. Biofertiliser generated from AD and aerobic composting

Although the quality of biofertiliser largely depends on the feedstock used, there is no significant difference in terms of nitrogen, ammonia nitrogen and phosphorus contents between the biofertiliser generated following AD and aerobic composting (Tables 2 and 3). Rigby and Smith (2011) compared the physicochemical properties of food waste digestate from AD with nine commercially available garden fertilisers in the UK market. Results

Table 4
Summary of literature involving in-situ degradation of agriculture residue for bio-fertilizer production.

Feedstock	Degradation method	Location	Time	Results	Reference
Winter wheat and maize	National degradation, no addition of microorganism, nor nutrients	Quzhou, China,	1985 to 2001	Soil organic matter increased from 7.0 g/kg to 11.9 g/kg	Niu et al., 2011
Wheat Straws	Mix strain inoculation *Bacillus subtilis, Bacillus licheniformis, Hansenula* sp.,*Schizosaccha- romyces sp.* and *Thermoactinomyces sp.*	Jiangsu, China	June 2, 2014, to September 9, 2014	Microorganism inoculants accelerated the decomposition speed of the wheat straw incorporated into the soil	Liu et al., 2016
Maize straw	Straw returned at rates of 0, 2250, 4500, and 9000 kg ha^{-1} with 360 kg N/ha/y and 240 kg P_2O_5/ha/y	North-central China	October 1981 to June 2012	High rates of straw return changed microbial community structure and increased the activity of most hydrolytic enzymes	Zhao et al., 2016
Maize straw	Moisture rate at 22.5%, 20.0%, 17.5% and 15.0% respectively, and with 1.3% NH_4HCO_3	North West Agriculture & Forestry University, China	60 days	Soil moisture affected straw degradation at early stage. After straw incorporation, water also produced through straw degradation process, which supplement water for soil and benefit water retention of soil	Zuo and Jia, 2004
Rice straw	NPKS (N, P, K fertilizer application + rice straw returen), NPK (N, P, K fertiliser applied only), CK (unfertilised control)	Huangjin Village, Hunan Province China	October 1981 to March 2005	The soil easily extractable glomalin (EEG), total glomalin (TG) concentrations, soil organic C (SOC) and total N (TN) were all higher in the NPKS plot than in the NPK and CK plot. Rice straw return also enhanced the contents of microbial biomass (MBC) and microbial biomass N (MBN) in the NPKS plot	Nie et al., 2007
Wheat, corn, proso millet, sorghum, hay millet, sunflower, and sudex	Direct in-situ degradation. Half was fertilised with P (9.5 kg/ha) and the other half received no P fertiliser	Sterling (40°22′N, 103°8′W), Stratton (39°11′N, 102°16′W), Walsh (37°14′N, 102°10′W) in eastern Colorado	13 years (start from 1985)	The best decomposition resulted in Sterling, summit, with wheat-fallow rotation; Crop yields between the P and no-P fertilized halves of each experimental unit were not significantly different	Ma et al., 1999
Rice straw	(1) Zero-fertilizer (control); (2) Inorganic NPK fertiliser (NPK) (3) NPK fertiliser along with medium and manure (30% and 60% N from manure respectively) (4) NPK fertiliser along with rice straw	Xinhua, Ningxiang, and Taojiang, Hunan, China	1986 to 2003	The application of inorganic fertilizer along with straw significantly increased soil organic C, N_{tot}, soil C_{mic} and N_{mic} contents for all three sites, when compare to the control.	Hao et al., 2008

revealed that on average digestates had similar nitrogen value as the commercial garden fertilisers, but were lower in phosphorus and potassium content. The heavy metal content of the food waste derived biofertiliser met the UK standards defined in the Quality Protocol PAS110 (Rigby and Smith, 2011).

The utilization of biofertiliser in field tests concluded that use of AD digestate and compost has various benefits, including providing organic material, adjusting C/N ratio, enhancing pH, improving water holding capacity, alleviating salinity and increasing aggregate stability in soils (Sangamithirai et al., 2015; Wang et al., 2017). Several case studies have highlighted that application of biofertiliser alone may not provide all the nutrients required. Mkhabela and Warman (2005) compared three composts derived from OFMSW with chemical fertilisers on two plants (potato and sweet corn) over a two-year period (1996, 1997) in Canada. The results revealed that the compost used in this study had an equivalent phosphorus value as those found in inorganic fertilisers, but a lower nitrogen value. A later report from the same group also indicated a compost of OFMSW provided insufficient nitrogen, but enough minerals (Hargreaves et al., 2009). Hargreaves et al., 2008 reviewed the application of MSW derived compost in agriculture as a biofertiliser. In comparison with non-source separated MSW, compost obtained from source separated MSW (OFMSW) was considered to be safe for agriculture application. There was no accumulation of metals or an increase in salt concentration in the soil. Horrocks et al., 2016 carried out a field trail, which used compost of OFMSW on cereal and forage crops. Over 3–4 y period, only 13–23% of available nitrogen (mainly mineral nitrogen) present in the compost was taken by the crops. This was relatively low when compared with chemical fertilizer, in which the nitrogen uptake efficiency is around 25–50% (Hirel et al., 2011). However, compost was applied in relatively high amounts, e.g. 20–30 tonne/ha (Sortino et al., 2013). This may cause environmental impact due to accumulation of heavy metals over repeated applications. By comparison, the SBO obtained from chemical hydrolysis of compost contained higher organic N, no ammonia N, and therefore can be used at lower doses, e.g. 140 kg/ha (Sortino et al., 2013). The accumulation of toxic compounds would be minimized.

8.2. *Biofertiliser generated from in situ degradation of crop residue*

The organic content of straw, such as cellulose, hemicellulose, lignin and protein was transformed into organic matter by soil microbes with humic acid produced as the primary product (Song et al., 2017). Humic acid intercalates with metal ions such as calcium or magnesium in the soil forming a stable particle cluster to prevent soil erosion, to enhance the soil permeability and to improve water use efficiency (Malik and Azam, 1985). The increase of soil organic matter content by *in situ* degradation of straw has been proven by numerous long-term experiments (Lehtinen et al., 2014; Wei et al., 2015). Wei et al. (1990) found that after returning all the produced straws back to the field, the capacity of the ploughed layer increased by 0.19–0.20 g/cm^3, non-capillary porosity increased by 0.5–3%, and the number of particle cluster with a diameter greater than 2 mm raised by 202.9%. Therefore, the air permeability, heat preservation and water conservation of soil were improved (Wei et al., 1990). In Lingchuan, Shanxi province, China, around 1/3 of the maize straw is used as a biofertiliser and is returned back to field. As a result of this activity over 10 y, soil permeability has increased by 30%, soil erosion has decreased by 60–70%, and average grain output has increased by 15% (Shen and Chen, 2009).

The method of directly returning agriculture residues into soil has been shown to increase total microbial count and enzymatic activities (Marschner et al., 2003). Zeng et al. (1988) discovered that the *in situ* degradation achieved an increase of 142.9% for bacteria number and an increase of 115% in fungi number in the 0–20 cm ploughed layer after the degradation of straw. Bandick and Dick (1999) showed that the activities of urease, phosphatase and neutral phosphatase increased by 36.8%, 43.8% and 14.6% respectively in the soil as a result of *in situ* straw degradation. Furthermore, the activity of cellulase, sucrose hydrolase, catalase, and relevant lignin degrading enzymes were all upregulated (Bandick and Dick, 1999). A healthy micro-ecology has been shown to correlate with an enhancement of the soil's resistance to pest and degradability of external pollutants such as pesticide residues and petroleum (Dong et al., 2014; Wu et al., 2014).

Besides nutrients directly liberated from straws, soil fertility is improved through microbial reaction related to *in situ* degradation (Zhu et al., 2010). In most cases of *in situ* degradation, addition of microorganism is carried out (Liu et al., 2016; Borah et al., 2016). A certain amount of nitrogen should be supplemented to satisfy the N requirement by microorganisms for the decomposition process (C/N of 25–30). Unlike carbon sources which are consumed during microbial respiration, almost all of nitrogen supplemented from fore-mentioned processes are transformed into biological nitrogen and are stored in soil. In the case either external nitrogen fixing microorganisms are added or natural nitrogen fixing microorganisms are stimulated, nitrogen fixation activity is accelerated and thus increases absolute contents of nitrogen in soil (Recous et al., 1999). At the same time, unstable inorganic nitrogen fertilisers are converted into stable biological nitrogen as a consequence of increased microbial activity. Nitrogen is released into soil following decay and decomposition of these microorganisms creating a slow-release of nitrogen (Mary et al., 1996). This observation has been confirmed by Kessel et al. (2000), in which he correlated the direct returning of straw back to soil with an improvement in absorption efficiency of growing crops by increasing organic matter in soil and a reduction in nitrogen loss by modifying soil physiochemical properties.

9. Challenges and future perspectives

The utilization of food waste in AD has already been established at a commercial scale. The digestate in AD plant using food waste as sole or main feedstock has been used as biofertiliser in the agricultural sector. Currently, the technology for converting certain food processing waste and proper source separately MSW is well developed (Rigby and Smith, 2011). However, the heterogeneous nature of food waste is still a challenge for operating an AD efficiently and for controlling subsequent biofertiliser quality. The excessive usage of nitrogen fertiliser in farmland could lead to various pollution, such as nitrate leaching to drink water, ammonia volatilization and NO_x emissions to atmosphere (Zavattaro et al., 2016). In 1991, EU approved the Nitrates Directive, to regulate the annual load of nitrogen fertiliser to agriculture land, especially in Nitrate Vulnerable Zones (Zavattaro et al., 2016). In the UK, the maximum total nitrogen loading rate must be below 250 kg per ha within any 12 month window for an individual field in the Nitrate Vulnerable Zones. This restriction also applies to biofertiliser, which is derived from AD plants. With the soaring installed AD capacity in recent years, biofertiliser supply could exceed the local demand in the near future. As the storage of biofertiliser is challenging, in order to spread biofertiliser into a farm in a further distance from the AD plant, a cost effective drying technology is desired to remove the water content in the digestate, which will enable biofertiliser to be transported at a reasonable cost. The chemical hydrolysis of digested food waste could be a possible solution, which generates SBO containing higher total nitrogen content and requires low loading rate to the field (Sortino et al., 2013). Another promising technique for biofertiliser upgrading is ammonia striping, which extracts ammonia out of wet digestate and then concentrates to a solid fertiliser. Although this process is well developed in chemical industry, the economic feasibility of its application in AD plant should be evaluated. Very recently, a new anaerobic digestion process has been proposed by adding SBO in the fermentation system (Francavilla et al., 2016). It produces a digestate with low ammonia content in comparison with the conventional anaerobic digestion process.

With the increasing understanding of the process, aerobic composting of food waste for biofertiliser production has already been commercialized. This is supported by the latest bibliometrics study on food waste (Chen et al., 2017), in which the papers published with the key word "compost" dropped from the 2nd highest during 1997–2002 to 6th during 2009–2014. Well-managed composting process is an appropriate option for a sustainable food waste management. A main challenge in food waste composting is the release of CH_4, N_xO, NH_3 and other odourous gases to the atmosphere (Saer et al., 2013; Salemdeeb et al., 2016). It contributes to greenhouse gas emission, and the lose of potential energy that could be captured from food waste. Therefore, AD is an attractive technology than composting for treatment of food waste in general. Another challenge is the long composting process, which normally takes 30–90 d. Composting in vessels under control condition could be faster (Pandey et al., 2016), but may not be cost effective. As food waste contains a higher water content, the amount of compost leachate would be high and would require additional treatment to avoid high ammonia emission. Improving compost quality is always important. A newly developed vermicomposting process has attracted wide interests by adding earthworm to the digester. Lim et al. (2016) compared vermicomposting process with traditional composting process. Vermicomposting could derive a biofertiliser with improved quality in term product texture and lower heavy metal content (Lim et al., 2016).

AD is an economical feasible technology to treat food wastes as a sole or co-digestion feedstock in commercial scale AD plants (Table 1). The solid residue of AD is rich in nutrients and can be considered as organic fertiliser in proper managed AD plants (Mata-Alvarez et al., 2014). The main incomes of a food waste based AD facility come from the following three streams: (1) The saving of waste disposal charge (for a factory that generates food waste) or the payment received for treatment the food waste collected; (2) The saving of expense on heat and power, which is obtained from the burning of biogas (methane) or the sale of excess electricity to national grid; (3) the sale of biofertiliser. The market value of biofertiliser depends upon the consistency of the quality, the nutrient value of the biofertiliser (mainly the nitrogen content), the location of the plant, the season of a year, and the public acceptance of biofertiliser. Using UK as an example, according to a market survey of digestate carried out in March–June 2012 (King et al., 2013), biofertiliser from AD has the greatest potential to be used in: land restoration, e.g. soil improvement; organic component for soil manufacture and field grown horticulture. However, several major concerns also have been raised, such as the odour control, pathogen control, high water content, high salt content and quality variation. Since then, AD technology developed rapidly in the UK and worldwide. By 2016, among the fifteen AD plants introduced in Table 1, twelve of them produced biofertiliser as a product as introduced in their websites, and seven of them demonstrated commercial land application of their biofertiliser (Table 1). Nonetheless, the selling prices of the biofertiliser are not available. The potential saving of partial replacement of chemical fertiliser by biofertiliser was estimated to be £84–118/ha (based on the usage of 30 m^3/ha liquid biofertiliser) (Taylor and Tompkins, 2015; Regrow, 2017). Tamar Energy claimed that their biofertiliser provided an estimated saving of £60,000/annual on Worth Farms on potato and vegetable crops (Tamar Energy, 2017). Ma et al. (2017) compared co-digestion of food waste with activated sludge for biogas production only with co-production of biogas and biofertiliser in Singapore. With the assumption that the biofertiliser could sell for a price of 1.5 SGD/kg (~£ 0.83/kg), the overall annual revenue of co-production of biogas and biofertiliser was estimated to be 180% higher than the sole biogas production. However, the capital investment of these two approaches has not been discussed.

In comparison with AD, the aerobic composting system requires low capital investigation, but loses the potential profit of energy generation. Lim et al. (2016) reviewed the economic feasibility and sustainability of aerobic composting technology for organic solid waste management. It concluded that aerobic composting process was generally viable although some reports showed negative economic perspective (Galgani et al., 2014; Fan et al., 2016). Chen (2016) analyzed six food waste composting plants in Taiwan. Three of them make profits while the three government-affiliated units have negative profit. The retail prices of biofertiliser are £0.16–0.3/kg; those are generally cheaper than the local mineral fertiliser (£0.15–0.9/kg). The low mark value, together with the compost quality variation, feedstock supply stability may affect the economic feasibility of aerobic composting system.

Returning straw back to soil could be a low cost option to generate an organic fertiliser from high lignocellulosic food waste. *In situ* straw degradation using only indigenous microorganisms takes considerable time, normally 3–6 months. The long decaying period limits the amount of agriculture residue that could be loaded into the field. External addition of soil microorganisms is inevitable for increasing the loading rates of straw in order to enhance soil fertility or for reducing the decomposition period to enable field for the next rotation of crops. However, there is a potential biological risk in the release of cultured microorganism into natural environments. Strict regulations should be implemented to control operation procedure. Addition of microorganism in the medium also poses a risk of promoting an unwanted strain that is unculturable in natural environment. Lab experiments should be carried out to demonstrate the safety before a field test can be carried out.

10. Conclusion

Food waste management is an emerging challenge worldwide. Conversion of food waste stream into biofertiliser could be a promising alternative approach for the valorization of food waste. It reduces environment burden of waste disposal, brings in additional income to food processing industry, directs benefit agricultural regions and reduces the use of chemical fertilisers. Generation of biofertiliser as an additional income to complement biogas production has already been implemented in many AD plants, and field tests of these biofertilisers from properly managed AD plants have been demonstrated to be safe (Table 1). With the increasing understanding of the underlying principle of biomass degradation in AD, aerobic compost and in soil, biofertiliser production from food waste could play an increasingly important role in the near future.

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References

Abdullah, J.J., Greetham, D., Pensua, N., Tucker, G.A., Du, C., 2016. Optimizing cellulase production from municipal solid waste (MSW) using solid state fermentation (SSF). J. Fundam. Renew. Energy Appl. 6 (3).

Adhikari, B., Barrington, S., Martinez, J., King, S., 2009. Effectiveness of three bulking agents for food waste composting. Waste Manag. 29 (1), 197–203.

Al Seadi, T., Lukehurst, C., 2012. Quality Management of Digestate from Biogas Plants Used as Fertiliser. IEA Bioenergy, Task.

Arshadi, M., Attard, M., Lukasik, R.M., Brncic, M., Da Costa Lopes, A.M., Finell, M., Geladi, P., Gerschenson, L., Gogus, F., Herrero, M., Hunt, A., Ibáñez, E., Kamm, B., Mateos-Aparicio, I., Matias, A., Mavroudis, N., Montoneri, E., Morais, A., Nilsson, C., Papaioannoua, E.H., Richel, A., Rupérez, P., Skrbic, B., Bodroza Solarov, M., Švarc-Gajić, J., Waldron, K., Yuste-Córdoba, F.J., 2016. Pre-treatment and extraction techniques for recovery of added value compounds from wastes throughout the agri-food chain. Green Chem. 18 (23), 6160–6204.

Baglieri, A., Cadili, V., Mozzetti Monterumici, C., Gennari, M., Tabasso, S., Montoneri, E., Nardi, S., Negre, M., 2014. Fertilization of bean plants with tomato plants hydrolysates. Effect on biomass production, chlorophyll content and N assimilation. Sci. Hortic. 176 (11), 194–199.

Bandick, A., Dick, R., 1999. Field management effects on soil enzyme activities. Soil Biol. Biochem. 31, 1471–1479.

Benisek, M., Kukucka, P., Mariani, G., Suurkuusk, G., Gawlik, B.M., Locoro, G., Giesy, J.P., Blaha, L., 2015. Dioxins and dioxin-like compounds in composts and digestates from European countries as determined by the in vitro bioassay and chemical analysis. Chemosphere 122, 168–175.

Brandli, R.C., Bucheli, T.D., Kupper, T., Furrer, R., Stahel, W.A., Stadelmann, F.X., Tarradellas, J., 2007a. Organic pollutants in compost and digestate. Part 1. Polychlorinated biphenyls, polycyclic aromatic hydrocarbons and molecular markers. J. Environ. Monit. 9, 456–464.

Brandli, R.C., Kupper, T., Bucheli, T.D., Zennegg, M., Huber, S., Ortelli, D., Muller, J., Schaffner, C., Iozza, S., Schmid, P., Berger, U., Edder, P., Oehme, M., Stadelmann, F.X., Tarradellas, J., 2007b. Organic pollutants in compost and digestate. Part 2. Polychlorinated dibenzo-p-dioxins, and -furans, dioxin-like polychlorinated biphenyls, brominated flame retardants, perfluorinated alkyl substances, pesticides, and other compounds. J. Environ. Monit. 9, 465–472.

Cesero, A., Belgiorno, V., Guida, M., 2015. Compost from organic solid waste: quality assessment and European regulations for its sustainable use. Resour. Conserv. Recycl. 94, 72–79.

Chang, J., Chen, Y.J., 2010. Effects of bulking agents on food waste composting. Bioresour. Technol. 101 (15), 5917–5924.

Chang, J.I., Hsu, T.E., 2008. Effect of compositions on food waste composting. Bioresour. Technol. 99, 8068–8074.

Chen, L., Zhao, L., Ren, C., Wang, F., 2012. The progress and prospects of rural biogas production in China. Energy Policy 51, 58–63.

Chen, H., Jiang, W., Yang, Y., Yang, Y., Man, X., 2017. State of the art on food waste research: a bibliometrics study from 1997 to 2014. J. Clean. Prod. 140 (2), 840–846.

Chen, Y.T., 2016. A cost analysis of food waste composting in taiwan. Sustainability 8 (11), 1210.

Chikae, M., Ikeda, R., Kerman, K., Morita, Y., Tamiya, E., 2006. Estimation of maturity of compost from food wastes and agro-residues by multiple regression analysis. Bioresour. Technol. 97, 1979–1985.

Chinese Government (2017). http://www.saic.gov.cn/ywdt/gsyw/zyxwbysj/gwywj/201701/t20170106_174029.html (last access, 22/03/2017).

Davidsson, A., Kjerstadius, H., Haghighatafshar, S., Fick, J., Olsson, M., Wachtmeister, H., Eriksson, E., la Cour Jansen, J., 2014. Effect of anaerobic digestion at 35, 55 and 60 °C on pharmaceuticals and organic contaminants. Water Sci. Technol. 69 (6), 1282–1288.

Dong, Y., Lang, Z., Kong, X., Lu, D., Liu, Z., 2014. Kinetic and multidimensional profiling of accelerated degradation of oil sludge by biostimulation. Environ. Sci. Processes Impacts 17 (4), 763–774.

Eida, M.F., Nagaoka, T., Wasaki, J., Kouno, K., 2011. Evaluation of cellulolytic and hemicellulolytic abilities of fungi isolated from coffee residue and sawdust composts. Microbes Environ. 26 (3), 220–227.

Esteves, S., Devlin, D., 2010. Food Waste Chemical Analysis. http://www.wrapcymru.org.uk/sites/files/wrap/Technical_report_food_waste_characterisation_Wales_2009x2.9086.pdf (last access: 22/03/2017).

European Bioplastics (2017). http://docs.european-bioplastics.org/publications/bp/EUBP_BP_Anaerobic_digestion.pdf (last access, 10/03/2017).

Evans, T.D., Boor, M., MacBrayne, D., 2007. Biofertiliser plant design – food waste to biofertiliser and biogas. In: Proc. 12th CIWEM European Biosolids & Biowastes Conference, 13–15 November 2007 AquaEnviro (Manchester).

Fan, Y.V., Lee, C.T., Klemeš, J.J., Bong, C.P.C., Ho, W.S., 2016. Economic assessment system towards sustainable composting quality in the developing countries. Clean Technol. Environ. Policy 1–13. https://doi.org/10.1007/s10098-016-1209-9.

Farrell, M., Jones, D.L., 2009. Critical evaluation of municipal solid waste composting and potential compost markets. Bioresour. Technol. 100, 4301–4310.

Fernandez-Hernandez, A., Roig, A., Serramia, N., Civantos, C.G., Sanchez-Monedero, M.A., 2014. Application of compost of two-phase olive mill waste on olive grove: effects on soil, olive fruit and olive oil quality. Waste Manag. 34, 1139–1147.

Francavilla, M., Beneduce, L., Gatta, G., Montoneri, E., Monteleone, M., Mainero, D., 2016. Biochemical and chemical technology for a virtuous bio-waste cycle to produce biogas without ammonia and speciality bio-based chemicals with reduced entrepreneurial risk. J. Chem. Technol. Biotechnol. 91 (10), 2679–2687. https://doi.org/10.1002/jctb.4875.

Frischmann, 2012. Enhancement and Treatment of Digestates from Anaerobic Digestion. http://www.wrap.org.uk/sites/files/wrap/Digestates (last visit: 10/03/2017).

Fuchs, W., Wager, F., Kirchmayr, R., Braun, R., Dorsg, B., 2010. Digestate treatment: comparison and assessment of existing technologies. In: Third International Symposium on Energy from Biomass and Waste. CISA, Venice, Italy.

Galgani, P., van der Voet, E., Korevaar, G., 2014. Composting, anaerobic digestion and biochar production in Ghana, environmental-economic assessment in the context of voluntary carbon markets. Waste Manag. 34, 2454–2465.

Glithero, N.J., Wilson, P., Ramsden, S.J., 2013. Straw use and availability for second generation biofuels in England. Biomass Bioenergy 55, 311–321.

Gong, C., Wang, J., Chen, X., Zhang, Z., 2008. Comprehensive utilization of crop straw and returning Technology. Sci. Technol. Innovation Her 9, 251.

Govasmark, E., Stäb, J., Holen, B., Hoornstra, D., Nesbakk, T., Salkinoja-Salonen, M., 2011. Chemical and microbiological hazards associated with recycling of anaerobic digested residue intended for agricultural use. Waste Manag. 31 (12), 2577–2583.

Gunnarsson, A., Bengtsson, F., Caspersen, S., 2010. Use efficiency of nitrogen from biodigested plant material by ryegrass. J. Plant Nutr. Soil Sci. 173, 113–119.

Guo, R., Li, G.X., Jiang, T., Schuchardt, F., Chen, T.B., Zhao, Y.Q., Shen, Y.J., 2012. Effect of aeration rate, C/N ratio and moisture content on the stability and maturity of compost. Bioresour. Technol. 112, 171–178.

Gustavsson, J., Cederberg, C., Sonesson, U., van Otterdijk, R., Meybeck, A., 2011. Global Food Losses and Food Waste (FAO, Rural Infrastructure and Agro-industries Division, 2011).

Hall, K.D., Guo, J., Dore, M., Chow, C.C., 2009. The progressive increase of food waste in America and its environmental impact. PLoS One 4 (11), e7940.

Hao, X., Liu, S., Wu, J., Hu, R., Tong, C., Su, Y., 2008. Effect of long-term application of inorganic fertilizer and organic amendments on soil organic matter and microbial biomass in three subtropical paddy soils. Nutrient Cycl. Agroecosyst. 81, 17–24.

Hargreaves, J.C., Adl, M.S., Warman, P.R., 2008. A review of the use of composted municipal solid waste in agriculture. Agric. Ecosyst. Environ. 123 (1), 1–14.

Hargreaves, J.C., Adl, M.S., Warman, P.R., 2009. The effects of municipal solid waste compost and compost tea on mineral element uptake and fruit quality of strawberries compost. Sci. Util. 17 (2), 85–94.

Henriksen, T.M., Breland, T.A., 2002. Carbon mineralization, fungal and bacterial growth, and enzyme activities as affected by contact between crop residues and soil. Biol. Fertil. Soils 35, 41–48.

Hindle, M., 2013. Anaerobic digestion in the United Kingdom. BioCycle 54 (5), 41–43.

Hirel, B., Tetu, T., Lea, P.J., Dubois, F., 2011. Improving nitrogen use efficiency in crops for sustainable agriculture. Sustainability 3, 1452–1485.

Hong, J., Ren, L., Hong, J., Xu, C., 2016. Environmental impact assessment of corn straw utilization in China. J. Clean. Prod. 112, 1700–1708.

Horrocks, A., Curtin, D., Tregurtha, C., Meenken, E., 2016. Municipal compost as a nutrient source for organic crop production in New Zealand. Agronomy 6 (2), 35.

Hyun, H.S., Chung, J.W., Lee, H.B., Youn, J.K., Lee, C.Y., Kim, D.H., Cho, K.Y., 2009. Isolation of cellulose-degrading myxobacteria *Sorangium cellulosum*. Korean J. Microbiol. (in Korean) 45, 48–53.

Jiang, C., Yang, J., Xie, D., Qu, M., 2001. Decay of organic materials in newly-weathered purple rock fragments and its adjustment. J. Southwest Agric. Univ. 23 (5), 463–467.

Jordan, A., Zavala, L.M., Gil, J., 2010. Effects of mulching on soil physical properties and runoff under semi-arid conditions in southern Spain. Catena 81, 77–85.

Karpe, A.V., Beale, D.J., Morrison, P.D., Harding, I.H., Palombo, E.A., 2015. Untargeted metabolic profiling of *Vitis vinifera* during fungal degradation. FEMS Microbiol. Lett. 362 (10).

Kessel, C.V., Eagle, A.J., Bird, J.A., Horwath, W.R., Linquist, B.A., Brouder, S.M., Hill, J.E., 2000. Rice yield and nitrogen utilization efficiency under alternative straw management practices. Agron. J. 92 (6), 1096–1103.

Kiran, E.U., Trzcinski, A.P., Ng, W.J., Liu, Y., 2014. Bioconversion of food waste to energy: a review. Fuel 134, 389–399.

King, C., Bardos, P., Nortcliff, S., 2013. Market Expectations and Requirements for Digestate. http://www.wrap.org.uk/sites/files/wrap/Market%20expectations%20and%20requirements%20for%20digestate%20report.pdf. last visit: 23/03/2018.

Koster, J.R., Dittet, K., Muhling, K.-H., Kage, H., Pacholkski, A., 2014. Cold season ammonia emissions from land spreading with anaerobic digestates from biogas production. Atmos. Environ. 8435–8438.

Lehtinen, T., Schlatter, N., Baumgarten, A., Bechini, L., Kruger, J., Grignani, C., Zavattaro, L., Costamagna, C., Spiegel, H., 2014. Effect of crop residue incorporation on soil organic carbon and greenhouse gas emissions in European agricultural soils. Soil Use Manag. 30, 524–538.

Li, P.P., Wang, X.J., Yuan, X.F., Wang, X.F., Cao, Y.Z., Cui, Z.J., 2011. Screening of a composite microbial system and its characteristics of wheat straw degradation. Agric. Sci. China 10, 1586–1594.

Li, Z., Lu, H., Ren, L., He, L., 2013. Experimental and modeling approaches for food waste composting: a review. Chemosphere 93, 1247–1257.

Lim, S.L., Lee, Lh, Wu, T.Y., 2016. Sustainability of using composting and vermicomposting technologies for organic solid waste biotransformation: recent overview, greenhouse gases emissions and economic analysis. J. Clean. Prod. 111, 262–278.

Lin, C.S.K., Pfaltzgraff, L.A., Herrero-Davila, L., Mubofu, E.B., Abderrahim, S., Clark, J.H., Luque, R., 2013. Food waste as a valuable resource for the production of chemicals, materials and fuels. Current situation and global perspective. Energy. Environ. Sci. 6 (2), 426–464.

Lin, C., 2008. A negative-pressure aeration system for composting food wastes. Bioresource Technol. 99, 7651–7656.

Liu, Y., Zhao, S., Zhu, Q., Ding, W., 2016. The effect of microorganism inoculants on the color changing of the wheat straws after incorporated into soil.. In: 2016 ASABE Annual International Meeting. American Society of Agricultural and Biological Engineers.

Ma, L., Peterson, G.A., Ahuja, L.R., Sherrod, L., Shaffer, M.J., Rojas, K.W., 1999. Decomposition of surface crop residues in long-term studies of dry land agroecosystems. Agron. J. 91, 401–409.

Ma, Y., Yin, Y., Liu, Y., 2017. New insights into co-digestion of activated sludge and food waste: biogas versus biofertilizer. Bioresource Technol. 241, 448–453.

Mafe, O.A.T., Davies, S.M., Hancock, J., Du, C., 2014. Development of an estimation model for the evaluation of the energy requirement of dilute acid pretreatments of biomass. Biomass Bioenergy 72, 28–38.

Malik, K., Azam, F., 1985. Effect of humic acid on wheat *(Triticum aestivum L.)* seedling growth. Environ. Experim. Botany 25, 245–252.

Marschner, P., Kandeler, E., Marschner, B., 2003. Structure and function of the soil microbial community in a long-term fertiliser experiment. Soil Biol. Biochem. 35, 453–461.

Mary, B., Recous, S., Darwis, D., Robin, D., 1996. Interactions between decomposition of plant residues and nitrogen cycling in soil. Plant and Soil 181 (1), 71–82.

Mata-Alvarez, J., Dosta, J., Romero-Güiza, M.S., Fonoll, X., Peces, M., Astals, S., 2014. A critical review on anaerobic co-digestion achievements between 2010 and 2013. Renew. Sustain. Energy Rev. 36, 412–427. https://doi.org/10.1016/j.rser.2014.04.039.

Miyatake, F., Iwabuchi, K., 2005. Effect of high compost temperature on enzymatic activity and species diversity of culturable bacteria in cattle manure compost. Bioresource Technol. 96, 1821–1825.

Mkhabela, M., Warman, P., 2005. The influence of municipal solid waste compost on yield, soil phosphorus availability and uptake by two vegetable crops grown in a Pugwash sandy loam soil in Nova Scotia. Agric. Ecosyst. Environ. 106 (1), 57–67.

Mohammad, N., Alam, M.Z., Kabashi, N.A., 2015. Optimization of effective composting process of oil palm industrial waste by lignocellulolytic fungi. J. Mater. Cycles Waste Manag. 17, 91–98.

Moller, K., Muller, T., 2012. Effects of anaerobic digestion on digestate nutrient availability and crop growth: a review. Eng. Life Sci. 12 (3), 242–257.

Moller, K., Stinner, W., 2009. Effects of different manuring systems with and without biogas digestion on soil mineral nitrogen content and on gaseous nitrogen losses (ammonia, nitrous oxides). Eur. J. Agron. 30 (1), 1–16.

Moller, J., Miller, M., Kjoller, A., 1999. Fungal-bacterial interaction on beech leaves: influence on decomposition and dissolved organic carbon quality. Soil Biol. Biochem. 31 (3), 367–374.

Montemurro, F., Diacono, M., Vitti, C., Debiase, G., 2009. Biodegradation of olive husk mixed with other agricultural wastes. Bioresource Technol. 100 (12), 2969–2974.

Monterumici, C.M., Rosso, D., Montoneri, E., Ginepro, M., Baglieri, A., Novotny, E.H., Kwapinski, W., Negre, M., 2015. Processed vs. non-processed biowastes for agriculture: effects of post-harvest tomato plants and biochar on radish growth, chlorophyll content and protein production. Int. J. Mol. Sci. 16 (4), 8826–8843.

Nie, J., Zhou, J.M., Wang, H.Y., Chen, X., Du, C., 2007. Effect of long-term rice straw return on soil glomalin, carbon and nitrogen. Pedosphere 17, 295–302.

Borah, Nilay, Barua, R., Nath, D., Hazarika, K., Phukon, A., Goswami, K., Barua, D.C., 2016. Low energy rice stubble management through *in situ* decomposition. Procedia Environ. Sci. 35, 771–780.

Niu, L., Hao, J., Zhang, B., Niu, X., 2011. Influences of long-term fertiliser and tillage management on soil fertility of the North China Plain. Pedosphere 21 (6), 813–820.

Odier, E., Janin, G., Monties, B., 1981. Poplar lignin decomposition by gram-negative aerobic bacteria. Appl. Environ. Microbiol. 41, 337–341.

Owamah, H.I., Dahunsi, S.O., Oranusi, U.S., Alfa, M.I., 2014. Fertiliser and sanitary quality of digestate biofertiliser from the co-digestion of food waste and human excreta. Waste Manag. 34, 747–752.

Pandey, P.K., Vaddella, V., Cao, W., Biswas, S., Chiu, C., Hunter, S., 2016. In-vessel composting system for converting food and green wastes into pathogen free soil amendment for sustainable agriculture. J. Clean. Prod. 139, 407–415.

Panuccio, M.R., Attina, E., Basile, C., Mallamaci, C., Muscola, A., 2016. Use of recalcitrant agriculture wastes to produce biogas and feasible biofertilizer. Waste and Biomass Valorization 7 (2), 267–280.

Pensupa, N., Jin, M., Kokolski, M., Archer, D.B., Du, C., 2013. A solid state fungal fermentation-based strategy for the hydrolysis of wheat straw. Bioresource Technol. 149, 261–267.

Pivato, A., Vanin, S., Raga, R., Lavagnolo, M.C., Barausse, A., Rieple, A., Laurent, A., Cossu, R., 2016. Use of digestate from a decentralized on-farm biogas plant as fertiliser in soils: an ecotoxicological study for future indicators in risk and life cycle assessment. Waste Manag. 49, 378–389.

Preethu, D.C., Prakash, B.N.U.H., Srinivasamurthy, C.A., Vasanthi, B.G., 2007. Maturity indices as an index to evaluate the quality of compost of coffee waste blended with other organic wastes. In: Proceedings of the International Conference on Sustainable: Solid Waste Management, Sept 5-7, Chennai, India, pp. 270–275.

Recous, S., Aita, C., Mary, B., 1999. In situ changes in gross N transformations in bare soil after addition of straw. Soil Biology & Biochemistry 31, 119–133.

Regrow (2017). https://regrow-biofertiliser.co.uk/recalculator/nutrient-supply-examples (last access: 21/Sep/2017).

Research Market, (2017). https://www.researchandmarkets.com/research/fcrscs/fertilizers (last access: 11/Sep/2017)z

Rigby, H., Smith, S.R., 2011. New Markets for Digestate from Anaerobic Digestion. http://www.wrap.org.uk/sites/files/wrap/New_Markets_for_AD_WRAP_format_Final_v2.c6779ccd.11341.pdf (last visit: 15/09/2017).

Rigby, H., Smith, S.R., 2013. Nitrogen availability and indirect measurements of greenhouse gas emissions from aerobic and anaerobic biowaste digestates applied to agricultural soils. Waste Manag. 33 (12), 2641–2652.

Rossi, C., Pereira, M., García, A., Berbara, R., Gazolla, P., Perind, A., 2016. Effects on the composition and structural properties of the humified organic material of soil in sugarcane strawburning: a chronosequence study in the Brazilian Cerrado of Goiás State. Agric. Ecosyst. Environ. 216, 34–43.

Rosso, D., Fan, J., Montoneri, E., Negre, M., Clark, J., Mainero, D., 2015. Conventional and microwave assisted hydrolysis of urban biowastes to added value lignin-like products. Green Chem. 17 (6), 3424–3435.

Saer, A., Lansing, S., Davitt, N.H., Graves, R.E., 2013. Life cycle assessment of a food waste composting system: environmental impact hotspots. J. Clean. Prod. 52, 234–244.

Salemdeeb, E.K.H.J., zu Ermgassen, M.H., Balmford, Kim A., Al-Tabbaa, A., 2016. Environmental and health impacts of using food waste as animal feed: a comparative analysis of food waste management options. J. Clean. Prod. 140, 871–880.

Sangamithirai, K.M., Jayapriya, J., Hema, J., Manoj, R., 2015. Evaluation of in-vessel co-composting of yard waste and development of kinetic models for co-composting. Int. J. Recycl. Org. Waste Agric. 4, 157–165.

Shen, Y.Y., Chen, H., 2009. The progress of study on soil improvement research with straw stalk. Chin. Agric. Sci. Bull. 25 (19), 291–294.

Shi, Y., Shen, Q., Lou, W., Liu, S., Cui, Z., 1996. Isolation and screening of cellulose-decomposing mixing strains. J. Nanjing Agric. Univ. 19 (3), 59–62.

Sohail, A.Q., Ambrin, R., Mehrunisa, M., Muhammad, A.S., 2014. Nutrient composition of rock phosphate enriched compost from various organic wastes. J. Sci. Res. 2, 47–51.

Song, G., Novotny, E., Mao, J., Michael, H., 2017. Characterization of transformations of maize residues into soil organic material. Sci. Total Environ. 579, 1843–1854.

Sortino, O., Dipasquale, M., Montoneri, E., Tomasso, L., Avetta, P., Bianco Prevot, A., 2013. 90% yield increase of red pepper with unexpectedly low doses of compost soluble substances. Agron. Sustain. Dev. 33, 433–441. https://doi.org/10.1007/s13593-012-0117-6.

Stasinakis, S.A., 2012. Review on the fate of emerging contaminants during sludge anaerobic digestion. Bioresour. Technol. 121, 432–440.

Stinner, W., Moller, K., Leithold, G., 2008. Effect of biogas digestion of clover/grass-leys, cover crops and crop residues on nitrogen cycle and crop yield in organic stockless farming system. Eur. J. Agron. 29 (2–3), 125–134.

Sun, J., Peng, H., Chen, J., Wang, X., Wei, M., Li, W., Mellouki, A., 2016. An estimation of CO2 emission via agricultural crop residue open field burning in China from 1996 to 2013. J. Clean. Prod. 112, 2625–2631.

Sundberg, C., Yu, D., Franke-Whittle, I., Kauppi, S., Smårs, S., Insam, H., Romantschuck, M., Jönsson, H., 2013. Effects of pH and microbial composition on odour in food waste composting. Waste Manag. 33, 204–211.

Tamar Energy (2017). http://www.tamar-energy.com/compost-biofertiliser/biofertiliser (last access: 21/Sep/2017).

Tampio, E., Marttinen, S., Rintala, J., 2016a. Liquid fertiliser products from anaerobic digestion of food waste: mass, nutrient and energy balance of four digestate liquid treatment systems. J. Clean. Prod. 125, 22–32.

Tampio, E., Salo, T., Rintala, J., 2016b. Agronomic characteristics of five different urban waste digestates. J. Environ. Manag. 15 (169), 293–302.

Taylor, M., Tompkins, D., 2015. Understanding and Marketing Digestate. http://www.wrap.org.uk/sites/files/wrap/Understanding%20and%20marketing%20digestate.pdf (last access: 21/Sep/2017).

UK government (2017). http://www.legislation.gov.uk/uksi/1991/1590/contents/made (last access: 15/09/2017).

Van Overbeek, L., Runia, R., 2011. Phytosanitary Risks of Reuse of Waste Streams and Treated Wastes for Agricultural Purposes. University of Wageningen. Plant Research International, Report 382. http://edepot.wur.nl/167480 (last visit: 10/03/17).

Ventrella, D., Stellacci, A., Castrignanò, A., Charfeddine, M., Castellini, M., 2016. Effects of crop residue management on winter durum wheat productivity in a long term experiment in Southern Italy. Eur. J. Agron. 77, 188–198.

Walker, L., Cord-Ruwish, R., Sciberras, S., 2012. Performance of a commercial-scale DiCOM demonstration facility treating mixed municipal solid waste in comparison with laboratory-scale data. Bioresour. Technol. 126, 404–411.

Wang, Y., Bi, Y., Gao, C., 2010. The assessment and utilization of straw resources in China. Agric. Sci. China 9, 1807–1815.

Wang, X., Peng, Z., Sun, X., Liu, D., Chen, S., Li, F., Xia, H., Lu, T., 2012. The FPase properties and morphology changes of a cellulolytic bacterium, *Sporocytophaga* sp. JL-01, on decomposing filter paper cellulose. J. Gener. Appl. Microbiol. 58, 429–436.

Wang, X., Selvam, A., Wong, J., 2016. Influence of lime on struvite formation and nitrogen conservation during food waste composting. Bioresour. Technol. 217, 227–232.

Wang, X., Pan, X., Zhang, Z., Lin, X., Zhang, Y., Chen, S., 2017. Effects of the feeding ratio of food waste on fed-batch aerobic composting and its microbial community. Bioresour. Technol. 224, 397–404.

Warp, 2017. http://www.wrap.org.uk/content/operational-ad-sites (last access, 10/03/2017).

Wei, T., Cheng, L., Zhu, L., 1990. The research progress of improvement in soil after Returning. Chinese J. Agric. Mechani. Res. 2, 48–52.
Wei, T., Zhang, P., Wang, K., Ding, R., Yang, B., Nie, J., Jia, Z., Han, Q., 2015. Effects of wheat straw incorporation on the availability of soil nutrients and enzyme activities in semiarid areas. PLoS One 10 (4), e0120994.
Wu, B., Lan, T., Lu, D., Liu, Z., 2014. Ecological and enzymatic responses to petroleum contamination. Environ. Sci. Proces. Impacts 16 (6), 1501–1509.
Yin, Y., Liu, Y., Meng, S., Kiran, E.U., Liu, Y., 2016. Enzymatic pretreatment of activated sludge, food waste and their mixture for enhanced bioenergy recovery and waste volume reduction via anaerobic digestion. Appl. Energy 179, 1131–1137.
Yong, H., Zhen, R., Yong, I., 2001. The effect of stubble return on agro-ecological system and crop growth. Chin. J. Soil Sci. 32, 209–213.
Zavattaro, L., Assandri, D., Grignani, C., 2016. Achieving legislation requirements with different nitrogen fertilization strategies: results from a long term experiment. Eur. J. Agron. 77, 199–208. https://doi.org/10.1016/j.eja.2016.02.004.
Zeng, G., Fu, S., Jin, P., 1988. The effect of organic materials on soil fertility improvement. Heilongjiang Agric. Sci. 3, 35–39.
Zhang, L., Sun, X., 2016. Influence of bulking agents on physical, chemical, and microbiological properties during the two-stage composting of green waste. Waste Manag. 48, 115–126.
Zhang, C., Su, H., Baeyens, J., Tan, H., 2014. Reviewing the anaerobic digestion of food waste for biogas production. Renew. Sustain. Energy Rev. 38, 383–392.
Zhang, R., El-Mashad, H.M., Hartman, K., Wang, F., Guangqing, L., Choate, C., Gamble, P., 2007. Characterization of food waste as feedstock for anaerobic digestion. Bioresour. Technol. 98 (4), 929–935.
Zhao, X., Lin, Q., Sun, Y., Wang, Y., Zhang, Y., Zhang, M., 2000. Decomposition of different cellulose materials by some cellulose-decomposing microbes. J. Microbiol. 20 (3), 12–14.
Zhao, S., Li, K., Zhou, W., Qiu, S., Huang, S., He, P., 2016. Changes in soil microbial community, enzyme activities and organic material fractions under long-term straw return in north-central China. Agric. Ecosyst. Environ. 216, 82–88.
Zhu, H., Wu, J., Huang, D., Zhu, Q., Liu, S., Su, Y., Wei, W., Syers, J.K., Li, Y., 2010. Improving fertility and productivity of a highly-weathered upland soil in subtropical China by incorporating rice straw. Plant and Soil 331 (1), 427–437.
Zuo, Y., Jia, Z., 2004. Effect of soil moisture content on straw decomposing and its dynamic changes. J. Northwest Sci. Technol. Univ. Agric. For. 32 (5), 61–63.