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Review

Dry anaerobic digestion of organic waste: A review of operational parameters and their impact on process performance

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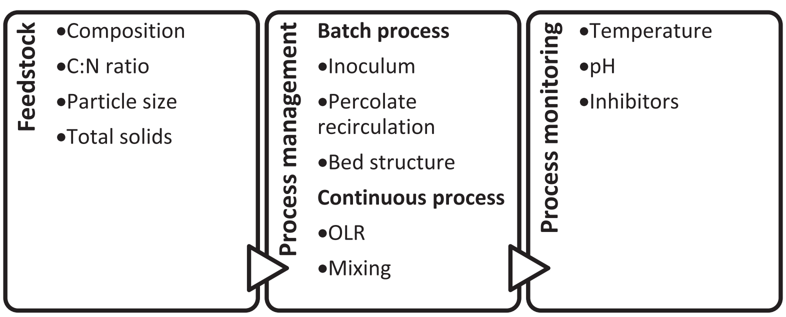
GRAPHICAL ABSTRACT



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ABSTRACT

Dry digestion is a suitable technology for treating organic wastes with varying composition such as the organic fraction of municipal solids waste. Yet, there is a need for further research to overcome some of the disadvantages associated with the high total solids content of the process. Optimisation of inoculum to substrate ratio, feedstock composition and size, liquid recirculation, bed compaction and use of bulking agents are some of the parameters that need further investigation in batch dry anaerobic digestion, to limit localised inhibition effects and avoid process instability. In addition, further attention on the relation between feedstock composition, organic loading rate and mixing regimes is required for continuous dry anaerobic digestion systems. This paper highlights all the areas where knowledge is scarce and value can be added to increase dry anaerobic digestion performance and expansion.

1. Introduction

According to the European Commission, 58 million tonnes of municipal solid waste were disposed to landfill in Europe in 2017 (Eurostat, 2018), of which 46% is considered to be organic (Kaza and Bhada-Tata, 2018). These numbers have decreased in the European Union over the last few years as a result of individuals' behavioural changes in source segregating and recycling of organic and non-organic residues, with a reduction of 20.6% in the volume of landfilled waste between 2013 and 2017 (European Commission, 2019b). The need for sustainable waste management strategies has been reinforced by national and international legislative targets to reduce landfill disposal in

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Europe (European Commission, 2019a). Enforcement of waste reduction legislation has encouraged a shift from linear to circular waste management systems, enabling waste diversion from landfill and prioritising resource recovery during waste processing. On this note, anaerobic digestion (AD) has proved to be particularly effective for the treatment of organic waste streams, supporting renewable energy generation while avoiding risks of uncontrolled greenhouse gases emissions (GHGE) resulting from landfilling (Papageorgiou et al., 2009). As an example, in 2017 the AD biogas production in Europe amounted to 1.94 billion cubic meters (European Biogas Association, 2019), resulting from the treatment of around 5% of the total biodegradable waste generated across Europe (De Baere and Mattheeuws, 2014).

Dry AD, also referred to as high-solids or solid-state digestion, is one of the possible modes of operation of the AD process, the other being wet digestion. Dry AD is typically used to treat organic materials with high solids content, between 20% and 40%, making it particularly attractive for treatment of the organic fraction of municipal solid waste (OFMSW) and agricultural wastes (AW) (Guendouz et al., 2010), with methane yields ranging between 0.2 and 0.6 m^3/kg of volatile solids (VS) depending on the feedstock and configuration (Karthikeyan and Visvanathan, 2013). Dry AD popularity has increased in the recent years, but the lack of adequate knowledge compared to wet AD and the perceived operational complexity to achieve stable production make it still unpopular for some companies and regulators. In Europe in the last decade, dry AD has showed a 50% increase in capacity in the period from 2010 to 2015 (Charlottenburg and Rosenheim, 2015), although it only accounts for treatment of 35% of all the waste treated by AD (Charlottenburg and Rosenheim, 2015). Also other countries like China have shown an increase in popularity, being an ideal solution for the 0.9 billion tons of straw generated every year (Fu et al., 2018b), but being still behind wet AD in new installations.

Contrary to wet anaerobic digesters (ADs), continuous dry AD processes will lack of internal mixing and the new substrate and digestate are mixed outside the reactor before the feeding. Dry AD processes can be found as batch and continuous systems, with batch mode being generally preferred as it is easier to operate compared to continuous systems and resembles the in-vessel composting process, familiar to waste managers and farmers (Karthikeyan and Visvanathan, 2013). Batch reactors usually work at the higher end of the range of solids, between 30 and 40% TS, with methane yields between 0.2 to 0.5 m^3/kg VS. The fresh material is mixed with digestate (inoculum) from the previous batch and reactors are usually front-loaded using a front-end loader, similarly to the composting processes. To compensate for the lack of mixing, the percolate collected from the reactor can be recirculated to the top of the mixture, improving homogenisation of the system and gas mobilisation. In addition, the recirculation helps reducing the amount of initial solid inoculum needed as the percolate is rich in anaerobic microbial biomass. Percolate can also be mixed with the solid waste at the beginning of the reaction, reducing or even completely eliminating the need of using solid digestate as starting inoculum (Wilson et al., 2016). The most popular batch system currently in operation in Europe is the garage-type reactor with percolate recirculation. In this type of digester, the percolate drains through the material and is collected in a drainage system at the floor of the ADs, then stored separately in a tank and sent back to the top of the digester to be sprayed onto the digestion material as needed. Systems that allow for batch (DRANCO, VALORGA, KOMPOGAS) or continuous (DRANCO, VALORGA, KOMPOGAS) operation are currently available on the market (Table 1), operating with different TS contents (20–60%), organic loading rate (OLR) (up to 15 kg VS/m^3/d) and methane yields depending on feedstock and temperature of operation.

There are other clear advantages for dry AD, which include a reduced need of water addition, a higher OLR potential or a higher flexibility to peaks of demand for batch systems, as the ADs can allow higher capacity if needed. In addition, there is a reduction of abrasion in the reactor from sand and grit as a consequence of the lack of mobile parts (Karthikeyan and Visvanathan, 2013). All these factors can simplify day by day handling and operation over wet digestion as they allow the use of smaller reactors, simplify handling of digested residues and minimise wastewater effluents and nutrient losses during pretreatment. To illustrate, data gathered from literature (Table 2) showed that dry ADs operate at higher organic loading rates compared to wet ADs for materials like the OFMSW or AW, resulting in methane production per digester volume between 2 (OFMSW) to 10 (chicken manure) times higher with the same footprint as in wet systems.

Notwithstanding the main advantages, dry ADs have also some disadvantages, which include long degradation times and a potential accumulation of toxic and inhibitory compounds (e.g. volatile fatty acids, ammonia and heavy metals) due to the high TS content (Ajay et al., 2011). This can often lead to lower methane production per kilogram of VS and the need for higher inoculation ratios (Ajay et al., 2011; Chen et al., 2008).

Anaerobic process steps taking place in the ADs and the microbial communities inside the reactor are similar for wet and dry AD, with both processes usually treated similarly. However, dry AD has specific technical problems linked to the high TS content in the reactor, which require operational steps to be addressed differently than in wet AD in order to achieve optimised stable processes. The focus of this review is therefore to critically evaluate the scientific literature on dry AD, particularly on the effect that different feedstocks, process control and operation conditions have on process performance and operation. Furthermore, the paper aims to emphasize any gaps in knowledge that need to be addressed to enable a greater implementation of high solids digestion.

2. Parameters affecting dry anaerobic digestion performance

2.1. Feedstock characteristics

2.1.1. Composition

Dry anaerobic processes can be used to treat a variety of organic solid wastes, like agriculture waste (AW), OFMSW, garden waste (GW) and industrial waste (IW), as single substrate or in co-digestion. All of them have different composition and characteristics that will affect the process conditions and biogas production. OFMSW is widely available, with over 800 million tonnes per annum generated worldwide (Kaza et al., 2018) and a raising trend with the generation of MSW expected to reach a billion tonnes per year by 2025 (World Bank, 2010). In spite of its easy availability, OFMSW presents a high heterogeneity with great variations in composition depending on annual season and waste collection area. The waste collection method is also a factor affecting the composition, as source sorted OFMSW (SS-OFMSW) characteristics are different to the mechanically sorted OFMSW (MS-OFMSW), which will contain more contaminants and inert materials like glass and plastics. Biogas production for OFMSW ranges between 60 to 200 m^3/ton of treated waste, with a methane production of 0.13 to 0.4 m^3/kg VS (Li et al., 2011), where MS-OFMSW usually achieves lower productions than SS-OFMSW, due to the contamination problems (Table 3). Extensive examples of methane potentials obtainable for the different feedstocks can be found in literature, where the high methane potentials (Table 3) show the suitability of the different sourced MSW and AW for dry-AD digestion.

The range of values of methane potential for the different materials evidences the importance of a good characterisation of the feedstock composition in order to optimise process performance (solids degradation and methane production). Knowing substrate composition can also support control of process inhibition and provide the basis for selecting co-digestion substrates to enhance process efficiency and stability. As an example, Callaghan et al. (2002) investigated co-digestion of cattle slurry with fruit waste, vegetable wastes and chicken manure, reporting a stable operation when cattle manure was added

Table 1
Comparison of different commercially available dry AD systems.

Digester type	Feeding regime	T (°C)	Material fed	TS (%)	SRT (days)	OLR (kg VS/m^3/d)	VS removal (%)	Methane yield (m^3/kg $VS_{removed}$)	Reference
DRANCO	Continuous	50–55	SS-OFMSW	20–40	20	10–15	40–70	0.21–0.30	(Elsharkawy et al., 2019; Fagbohungbe et al., 2015; Karthikeyan and Visvanathan, 2013)
KOMPOGAS	Continuous	55	OFMSW	30	29	4.3	60–70	0.39–0.58	
VALORGA	Continuous	37–55	OFMSW	36–60	20–33	10–15	60–65	0.21–30	
BEKON	Batch	40–55	OW	40	28–35	NA	65–70	0.17–0.37	(Fu et al., 2018a)
BIOFerm	Batch	37	OFMSW	25	28	NA	50–55	0.21–0.35	(Fu et al., 2018a)
SEBAC	Batch	55	OFMSW	30	25–40	4.4–7.1	65–85	0.22–0.53	(Fdéz.-Güelfo et al., 2010)

Key: SS-OFMSW: source sorted organic fraction of municipal solid waste, OW: Organic waste.

Table 2
Comparison of biogas yields for feedstocks treated by dry and wet digestion.

Waste type	Type of digestion	Methane yield (m^3/kg VS_{Fed})	OLR (kg VS/m^3/d)	Methane yield (m^3/m^3 Digester)	Reference
Sweet potato vine	Dry	0.25	4.6	1.2	(Zhang et al., 2018)
	Wet	0.32	0.9	0.3	
OFMSW	Dry	0.14	90.0	12.2	(Di Maria et al., 2017)
	Wet	0.20	30.0	6.0	
Corn Stover	Dry	0.13	106.1	14.1	(Brown et al., 2012)
	Wet	0.12	14.5	1.8	
Switchgrass	Dry	0.12	106.1	12.3	(Brown et al., 2012)
	Wet	0.11	14.5	1.6	
Wheat straw	Dry	0.12	106.1	12.7	(Brown et al., 2012)
	Wet	0.14	14.5	2.0	
Chicken manure	Dry	0.18	5.3	1.0	(Bi et al., 2019)
	Wet	0.35	1.8	0.1	

until 50% of the feed. However, the addition of fruit and vegetable waste over 30% of the feed increased instability, and did not increase VS destruction or methane production, but produced a drop in pH from 7.7 to 7.2, indicating the risk of acidification if feedstock control is poor. Also André et al. (2019) reported an increase of methane yields when cattle manure (CM) and roadside grass cuttings (RGC) were digested together in a 60 L batch pilot scale reactor with percolate recirculation. Co-digestion increased methane yields from 0.20 and 0.17 m^3/kg VS for RGC and CM when mono-digestion was used, to 0.23 m^3/kg VS when 40% of RGC and 60% of CM where mixed. Therefore, a good understanding of the feedstock to be digested is necessary, not only because variability in composition will lead to different process yields at similar operational conditions (Brown et al., 2012), but also because it will show co-digestion opportunities and point out possible inhibitory substances.

2.1.2. C/N ratio

Feedstock total organic carbon (TOC), total nitrogen (TN) and their ratio are critical process parameters in the AD process. The addition of co-substrates to balance one or the other component is common practice to achieve stable digestion (Karthikeyan and Visvanathan, 2013) and optimal C:N ratios of 20 to 30 (Bouallagui et al., 2009). The nitrogen found in the AD reactor is mainly derived from proteins, and necessary for microbial growth, although a low C:N ratio in the system (high amount of nitrogen) can produce an ammonia accumulation in the digester, resulting in toxic levels for the process (Jokela and Rintala, 2003), affecting biogas and methane yields and eventually causing process failure (Chen et al., 2008). Addition of materials like paper waste or AW is common to increase the feedstock's carbon content (Li et al., 2011) and avoid inhibitory problems. As an example, Wang et al. (2012) used wheat straw to increase the C:N ratio in wet AD of dairy and chicken manure. The authors observed high values of 223 mg/l free ammonia (FA) at a C:N ratio of 15. These FA values decreased to 9.1, 7.5 and 2.2 mg/l, after straw addition, when C:N ratios were increased to 25, 30 and 35 respectively, achieving a stable digestion. Zeshan et al. (2012) reported a 30% drop in ammonia in the digestate when the C:N ratio was increased from 27 to 32 for a mixture of FW, green waste and paper waste in a pilot scale thermophilic dry anaerobic digester. On the same note, Zhang et al. (2012) used cardboard packaging to improve stability of food waste (FW) continuous digestion, allowing a higher OLR of 4 kg VS/m^3/day when co-digestion was used, compared to 2 kg VS/m^3/day when only FW was used, reporting lower concentrations of FA and no VFA accumulation, consequence of an increase in C/N ratio from 11 to 29.

2.1.3. Particle size

Particle size reduction is a common pre-treatment of solids for biological processes, as it releases intracellular organic matter and improves kinetics by providing a greater particle surface area (Muller, 2003). Despite the common implementation of size reducing pretreatments at full scale sites, and its importance for biogas yields, studies focussed on understanding the benefits of particle size reduction in

Table 3
Comparison of different materials and their methane yields when treated in dry AD.

Material fed	Thermophilic/Mesophilic	TS (%)	Methane yield (m^3/kg VS)	Reference
AW	M	18–35	0.013–0.331	(Brown et al., 2012; Hashimoto, 1989; Liu et al., 2019; Rouches et al., 2019)
GW	M	18–27	0.049–0.48	(Brown et al., 2012; Liu et al., 2009)
	T	24–27	0.217–0.357	(Liu et al., 2009)
MS-OFMSW	T	20	0.137–0.230	(Bolzonella et al., 2003; Lopes et al., 2004)
SS-MSW	M	n.a	0.128–0.319	(De Laclos et al., 1997; Kusch et al., 2008; Liu et al., 2009)
	T	n.a	0.400–0.631	(Liu et al., 2009; Rintala and Ahring, 1994)
MSW (Unsorted)	T	n.a	0.160–0.190	(Chugh et al., 1999)

Table 4
Impact of particle size on biogas yields for dry and wet AD.

Type of AD	Feedstock	Particle sizes (mm)	Methane yield (m^3/kg VS)	Effect of size reduction on methane yield	Reference
Dry AD	Wheat straw	0.1 0.7 1.4	0.052 0.075 0.096	Decrease	(Motte et al., 2013)
	Rice Straw	10 25	0.240 0.232	No statistical difference	(Zhang and Zhang, 1999)
	Corn Stover	1 12.7	0.217 0.191	Increase	(Wang et al., 2019)
	Banana peel	1 5 10	0.281 0.294 0.266	No statistical difference	(Tumutegyereize et al., 2011)
Wet AD	Paper fraction of MSW	10 1	0.117 0.118	No statistical difference	(Pommier et al., 2010)
	Wheat straw	20 2	0.179 0.244	Increase	(Gallegos et al., 2017)
	Sisal fibre waste	2 100	0.220 0.178	Increase	(Mshandete et al., 2006)
	Food Waste	0.4 0.8	0.673 0.791	Decrease	(Izumi et al., 2010)

dry AD are relatively scarce and contradictive at times. Motte et al. (2013) reported a decrease of 22% and 46% in methane yield when wheat straw particle size was gradually reduced from 1.4 to 0.7 and 0.1 mm (Table 4), while Zhang and Zhang (1999) showed no significant effect on the methane yield after grinding rice straw from 25 to 10 mm (Table 4), although reported faster kinetics for the smaller size.

A greater number of studies are available for wet digestion, although contradictory findings reported also for wet processes do not allow to draw a clear conclusion on the optimum particle size (Table 4). Smaller particle size of the pre-treated feedstock are generally associated with increased biogas production and accelerated kinetics, with previous references reporting increases up to 23% for a size drop from 100 to 2 mm when treating sisal fibre (Mshandete et al., 2006). Izumi et al. (2010) showed the opposite effect for size reduction, with a 0.4 mm reduction in particle size resulting in a 15% lower methane formation, which was attributed to the release of inhibitory compounds. Pommier et al. (2010) on the other hand, reported no statistically significant differences when the paper fraction of the MSW was reduced from 10 to 1 mm.

The impact of particle size on biogas formation has also been investigated in landfills, where literature showed that size reduction led to a decrease on biogas yield due to a higher volatile fatty acids (VFA) formation, which often produces an inhibition of the methanogenic activity (Barlaz et al., 1990). The limited and inconsistent reports available in literature make difficult to establish a clear pattern, highlighting the need for more research on the impact of the particle size for the different materials used as feedstock for dry AD.

2.1.4. Total solids

Dry AD processes work at higher TS contents than regular wet AD, which allows for treatment of higher amounts of waste per volume of digester. However, common problems associated with dry digestion, such as the lower methane production per kilogram of VS compared to wet AD (Table 2) and the accumulation of inhibitors, are directly linked to the high concentration of solids present in the system. The reduced water content is generally regarded as a reason for the gas and liquid diffusion problems and the accumulation of inhibitors. These, in turn, reduce substrate availability to the microbial biomass and affect their metabolism (Ge et al., 2016; Visvanathan, 2010). A number of studies

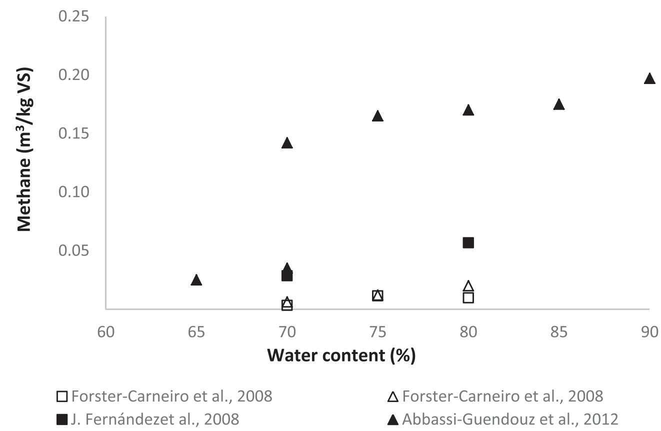

Fig. 1. Impact of batch ADs water content on methane production in dry anaerobic digestion treating FW (Forster-Carneiro et al., 2008), OFMSW (Fernández et al., 2008), and cardboard (Abbassi-Guendouz et al., 2012).

can be found in literature showing an increase in methane and biogas yields when water content increases (Fig. 1), all of them reporting that an increase of the water content increases methane yields, as this increase produces a better homogenisation in the ADs, reducing diffusion problems, increasing interaction between microorganisms and nutrients and diluting potential inhibitors. Also supporting the advantages of a reduced TS content, Le Hyaric et al. (2012) reported a linear increase of the specific methanogenic activity with the increase of water content, mainly due to a greater homogeneity within the system, which favours interaction between microorganisms and nutrients and dilutes potential inhibitors (Forster-Carneiro et al., 2008; Kusch et al., 2008).

2.2. Process control

Different process parameters are critical for batch and continuous operation and need to be analysed separately. For this reason, inoculum, percolate recirculation and bed characteristics are covered for batch processes, as I:S ratio is a defining parameter for process start-up, while percolate recirculation and bed structure are critical to improve homogeneity. For continuous processes OLR and mixing are covered, as are regarded to be the main design parameters in continuous ADs.

2.2.1. Batch process

2.2.1.1. Inoculum. Inoculum loading at the beginning of the batch digestion process is a way of accelerating the start-up period, and an efficient method to provide the necessary microbial population to the new substrate (Chen et al., 2008; Chugh et al., 1999; Di Maria et al., 2013). The most common procedure to inoculate the fresh material is the use of material digested from a previous batch, or digestate. Digestate characteristics change with time and operating conditions. Other materials used as inoculum include sewage sludge from a wastewater treatment works or digested manure from farm wet AD reactors (Karthikeyan and Visvanathan, 2013). Inoculum addition helps controlling retention time and biogas yield but reduces the available space to treat more new material. Reducing the amount of start-up inoculum increases waste treatment capacity but can lead to longer retention times and lower biogas and methane yields due to inhibitory problems.

Although the inoculum to substrate ratio (I:S) is one the key parameters for batch dry AD, there is not an optimum accepted ratio, as this varies with the type of system, operating conditions and substrate characteristics. Different authors and technology suppliers have recommended values, like the company Bekon (Harsewinkel, Germany), using 50% of the digestate as inoculum to start a new batch system in their garage-type reactors (Karthikeyan and Visvanathan, 2013). Other authors like Di Maria et al. (2012) suggested ratios between 1:1.5 to

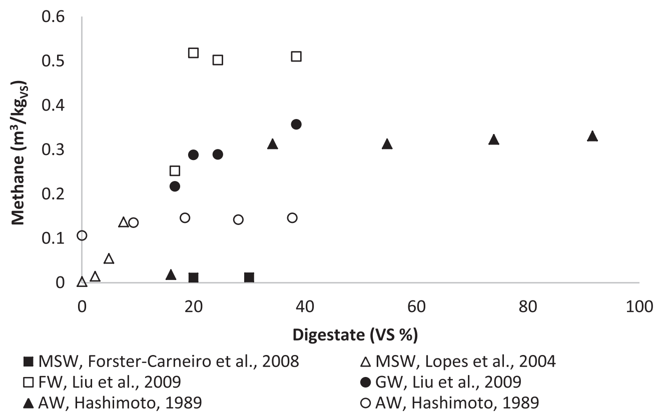

Fig. 2. Impact of digestate percentage on methane production in dry anaerobic digestion.

1:2.5 (61 to 72% of total weight) for a full scale dry AD plant with percolate recirculation treating food waste, while Kusch et al. (2008) reported values between 20% and 50% in weight for digestion of horse dung in leachate bed reactors, and Hashimoto (1989) indicated an optimum of 34% for small lab-scale batch reactors. Although there is not a consensus, it has been frequently reported that an increase of the digestate content can increase methane production until a maximum where the production becomes independent of digestate load (Fig. 2). Different examples are reported in literature (Fig. 2) were an increase in the amount of digestate used at the beginning of the batch process resulted in an increase of methane yield for different materials like MSW (Forster-Carneiro et al., 2008; Lopes et al., 2004), FW (Liu et al., 2009), GW (Liu et al., 2009) and AW (Hashimoto, 1989). This increase was explained by the greater presence of microorganisms responsible for the AD process, especially methanogenic archaea, as the amount of inoculum was increased in the fermenters.

2.2.1.2. Percolate recirculation. Inoculum addition at the beginning of the process is a common solution to avoid inhibition problems and speed up the process, but this is directly linked to a reduction of the available volume in the reactor to treat new substrate. One of the most common solutions proposed in literature to reduce the use of solid inoculum is the recirculation of percolate. Wilson et al. (2016) achieved a decrease on solid inoculum from 40 to 10%, without methane loss, when mature leachate from other ADs was used as inoculum. Whereas Kusch et al. (2008) reported that the use of solid inoculum could be completely avoided if enough percolate was used as the initial inoculum and then recirculated. In addition to a greater volume for feedstock treatment, the recirculation of percolate has been reported to provide additional benefits including: (1) an increase of moisture; (2) an improved contact between the methanogenic archaea and nutrients and (3) a greater reactor homogeneity, all of which allow for shorter times to reach the ultimate methane yield and increased final productions. Chan et al. (2002) reported close to 4 times more methane production and a shorter time of 9 days to achieve the maximal gas production rate when digesting MSW, sewage sludge and marine dredging with recirculation of percolate, compared to 11 days when recirculation was not used. Benbelkacem et al. (2010) recorded a 60% increase in total methane production when percolate was recirculated instead to added at the beginning of the batch process and drained.

Additionally, percolate recirculation in dry AD provides a washing effect of the inhibitors present in the reactor, such as VFA and ammonia. Pezzolla et al. (2017) studied the digestion of pig slurry with increasing frequencies of percolate recirculation compared to a no recirculation scenario, although failed to quantify the volume of percolate used in each recirculation. When recirculation was not used the VFA concentration peaked at the beginning of the digestion, increasing the risk of acidification in the ADs. VFA levels were gradually reduced as the recirculation frequency was increased, being 19 times lower for the highest frequency. Concentration of TAN present in the final digestate followed the same trend and was gradually reduced with the increase of percolate recirculation, decreasing from 5.4 to 0.4 g/kg, while increasing the total methane yield 2.5 times at the highest recirculation.

Different strategies for percolate recirculation have been reported in literature (Table 5). Studies using continuous or intermittent recirculation schemes are available, with all of them agreeing on that the use of percolate increases methane production compared to no recirculation. An increase of recirculation frequency, regardless of continuous or intermittent, is also reported to result in higher methane yields, as the homogeneity and the contact between methanogenic archaea and nutrients is increased. Although percolate recirculation is widely implemented in industrial sites with positive effects on process performance, some authors have reported negative effects when is recirculated in an excessive amount (Chen et al., 2008; Sponza and Ağdağ, 2004), showing that further understanding is necessary on the percolate composition and how it evolves with digestion, as can accumulate inhibitors like TAN, especially when dealing with nitrogen rich compounds like FW or OFMSW.

This shows the necessity of optimising percolate recirculation to maximise methane production, as well as of better describing recirculation strategy and percolate dosed amounts in scientific literature, as previous references generally lack description of the strategy used for percolate recirculation and quantification of amounts dosed.

2.2.1.3. Bed structure. Dry batch AD is affected by factors like micro and macro porosity, the degree of compaction or the permeability of the material to be digested (Andre et al., 2018), although only few studies have being done in this field. Shewani et al. (2017) studied the impact of micro and macro-porosity of cow manure digested in a leaching bed using computational fluid dynamics (CFD). They reported that bed compaction together with a macro-saturation of the pores caused by the successive recirculation of percolate, reduced homogeneity inside the bed and increased the liquid hold up, which could lead to operational

Table 5
Different percolate recirculation strategies and their impact on methane yield in dry anaerobic digestion.

Material	Conditions	Recirculation strategy	Percolate (L/kg Substrate*)	Methane yield (m^3/kg VS)	Reference
Manure	M	Once per hour	0.340	0.100–0.114	(Degueurce et al., 2016)
MSW	M	Once every 24 h	0.100	0.160–0.190	(Chugh et al., 1999)
FW	M	Continuous exchange	0.024–0.049	0.214–0.229	(Dearman and Bentham, 2007)
MSW	M	Continuous	0.310–0.875	0.084–0.210	(Sponza and Ağdağ, 2004)
OFMSW	M	3 times per day	0.049	0.211	(Di Maria et al., 2013)
Pig slurry + straw	M	1,2,4 times per day	0.133	0.150–0.241	(Pezzolla et al., 2017)
OFMSW + straw	T	1,3,10,15 times per day	0.250–0.750	0.172–0.225	(Rico et al., 2015)
Maize	M	Continuous & twice per day	0.100	0.389–0.347	(Kusch et al., 2012)
Pig urine + rice straw	T	Immersion and recirculation 1 h per 3 days	–	0.083–0.138	(Meng et al., 2019)

Key: *Total volume added during the whole batch digestion.

problems like bed fouling. Buffière et al. (1998) described the multiple effects that biofilms can have on the solid matter blocking the pores as the thickness of the biofilm formed in the media increases and produces a drop in acidogenic bacteria activity. The growth of this group of microorganisms is diffusion limited and hence particularly affected by the bed fouling. After measuring dry bulk density, total porosity and permeability of the bed material at different stages of the digestion André et al. (2015) also observed a compaction effect of the solid matter on the dry batch AD during the recirculation of percolate. After 19 days of operation, the compaction in the system prevented the percolate to flow through the media, forcing flow around the sides of the solid waste. After this point methane production was not observed to be impacted by the recirculation scheme and total yield at the end of the digestion was unaffected when percolate return stopped.

The use of bulking agents mixed with digestate and substrate is known to be standard procedure at full scale plants to alleviate compaction of the bed, but only a few references can be found in literature (Demirer and Chen, 2008; Kim et al., 2003; Xu et al., 2011). Materials like woodchip or saw dust are used to increase of permeability through the bed, reducing the compaction of the bed along the digestion period, increasing homogenisation and liquid–solid contact. Han et al. (2015) digested FW in a leachate bed reactor with and without bulking agent, reporting a 50% increase on percolate production and a 53% on COD concentration in the percolate when corn cobs where used as bulking agent, although VFA concentration was very similar.

Available literature shows the impact that bed structure can have in hydrodynamics, operation and performance of dry AD, highlighting the need to establish correlations between bed structure, compaction and methane production in order to optimise the process.

2.2.2. Continuous process

2.2.2.1. Organic loading rate. Biogas and methane production in continuous systems are determined by the organic loading rate (OLR) that can be modified while maintaining process stability. OLR, generally defined as kilograms of VS loaded per volume of digester per day, is hence considered one of the main design parameters for continuous dry AD (Fagbohungbe et al., 2015). Optimum values are higher than in wet AD processes (Karthikeyan and Visvanathan, 2013), and wide ranges can be found in literature (Table 6) for different feedstocks. Higher values are reported when OFMSW is used (9 to 19 kg VS/m^3/day) compared to sewage sludge (8.5 kg VS/m^3/day). Methane yields and stability of the process are also highly linked to the TS in the digester and the composition of the feedstock.

Nizami and Murphy (2010) reported ranges between 12 and 15 kg VS/m^3/day for corn silage, while others proposed lower values (8.5 kg VS/m^3/day, (Duan et al., 2012) demonstrating that OLR values are highly dependent on the feedstock material.

Maximum OLR values are constrained by the accumulation of inhibitory compounds like VFA or ammonia (Fernández-Rodríguez et al., 2014). As an example, Zeshan et al. (2012) reported a drop in methane from 218 to 121 L CH_4/kg VS when OLR was quickly increased from 4.0 to 10.7 kg VS/m^3/day for the digestion of OFMSW, consequence of an accumulation of VFA in the system that produced a drop in the pH. Li et al. (2017) reported a decrease of methane production in co-digestion of chicken manure and poplar leaf when OLR was increased from 4.0 to 8.0 kg VS/m^3/day. The reduction on this occasion was due to the accumulation of ammonia in the system. These results highlight the need for understanding feedstock characteristics, possible inhibitory mechanisms and co-digestion opportunities. OLR needs to be carefully selected to simultaneously maximise waste treatment capacity per unit of asset volume and renewable energy production, without reaching values that can compromise stable operation due to inhibitor accumulations.

2.2.2.2. Mixing. Effective reactor mixing is critical to maintain optimal biogas outputs. Mixing keeps microorganisms and substrate in contact, ensures consistent conditions within the whole digester volume and accelerates process kinetics and methane yields. The most common mechanisms of mixing in dry AD systems are biogas recirculation and mechanical mixing (Fagbohungbe et al., 2015). The commercial system VALORGAS uses biogas recirculation at the bottom of the AD as a mixing technique, while KOMPOGAS systems use a slowly rotating internal axial mixing and DRANCO ADs use external mixing (Karthikeyan and Visvanathan, 2013), with the internal mixing being regarded as more energy intensive (Karim et al., 2005).

Mixing, or lack of, becomes more critical in dry AD due to the high TS content (Singh et al., 2019). Karim et al. (2005) reported the necessity of mixing when TS are more than 5%, showing increases in methane production of around 20% when any form of mixing was used in 4 L lab scale reactors using cow manure slurry with a 10% TS content.

Generally mixing is regarded to have a positive effect in the digestion, but some authors report negative effects. Intense mixing during start-up or high load periods can produce negative effects, as high shear forces break microbial flocs and syntrophic relationships between methanogens and bacteria (Singh et al., 2019), producing an acidification of the system due to VFA accumulation. Karim et al. (2003) reported a lower methane production at the start-up of the process when mechanical mixing was used in 4 L lab scale wet ADs. pH was recorded to drop under 6 with mechanical mixing while only a smaller drop with pH over 6 was reported when biogas recirculation was used, and no drop when the ADs where unmixed. On the same note, Kaparaju et al. (2008) reported increases in methane production of 12.5% when minimal mixing (10 min before feeding/extraction) was used compared to continuous mixing on the start-up of semicontiguous lab ADs digesting manure at 8% TS.

Wet AD literature evidences that individual reactors can be affected in different way by mixing, even leading to inhibition if the mixing is too intense, or if it is applied in a wrong regime, while the impact of mixing in dry AD is particularly poorly documented. Whilst reports on dry AD suggesting that the lack of mixing is responsible for inhibitory problems are common, other authors (Dong et al., 2010) have pointed to the ability of dry AD to work at higher VFA concentrations than wet processes. Pointing to the relatively limited contact between methanogens and VFA in the ADs and emphasising the need of further research.

Table 6
Optimum values of OLR and methane yields for different feedstocks in continuous dry anaerobic digestion studies.

Feedstock	TS (%)	OLR (kg VS/m^3/day)	Methane yield (m^3/kg $VS_{destroyed}$)	Reference
Sewage sludge	20	8.5	0.190	(Duan et al., 2012)
Corn silage	12	3.5–8.5	0.327–0.410	(Veluchamy et al., 2019)
Swine manure	24	4–8	0.050–450	(Hu et al., 2019)
OFMSW	20	0.65–10.65	0.121–0.327	(Zeshan et al., 2012)
OFMSW	30	11.8	0.097	(Fdéz.-Güelfo et al., 2010)
OFMSW	20	9.2	0.230	(Bolzonella et al., 2003)
SS-OFMSW	20	12.1	0.490	(Pavan et al., 2000)
MS-OFMSW	18	19.0	0.342	(Gallert and Winter, 1997)

2.3. Operating conditions

2.3.1. Temperature

Depending on their temperature of operation dry ADs are classified as mesophilic or thermophilic processes. In mesophilic digestion temperature ranges between 35 and 40 °C, while thermophilic digestion operates between 50 and 57 °C (Visvanathan, 2010).

Thermophilic digestion has several advantages over its mesophilic counterpart, like higher rate of destruction of organic solids, better liquid–solid separation on the dewatering process, and a superior growth rate of microorganisms (Kim et al., 2006). Compared to mesophilic digestion, thermophilic conditions provide higher gas production at shorter residence time, which allows for smaller digester volumes and higher treatment capacity (OLR). The enhanced performance at increased temperatures has been widely reported. Kim et al. (2006) showed a 50% higher methane production when digesting FW in wet AD at 12% TS at temperatures between 45 and 50 °C than at 40 °C. Others like Liu et al. (2009), found an increase of 50 and 100% of the methane production when temperature was increased from 35 to 50 °C while digesting GW and FW in wet AD. In dry AD Fernández-Rodríguez et al. (2013) observed an increase of 27% in total methane production on digestion of OFMSW at 55 °C compared to 35 °C, together with an increase in kinetics, reducing the operation time to achieve the same production from 40 to 20 days.

Despite the potential for higher renewable energy production, operation at thermophilic conditions has drawbacks like a higher energy requirement for heating and a need for a more meticulous process control, as the microorganisms are more sensitive to changes in environmental conditions (Visvanathan, 2010). As an example of this higher sensitivity, Kim and Speece (2002) compared the response to the continuous increase of OLR for both continuously and daily fed ADs at mesophilic and thermophilic conditions. Propionate was considered the inhibitory parameter and was monitored accordingly, reporting constant values of around 50 mg/l for all the mesophilic reactors when OLR was increased from 2 until 10 kg VS/m^3/day, when the propionate started to accumulate with values over 2,000 mg/l producing a drop in pH and the inhibition of the system. Thermophilic ADs had different behaviour, increasing propionate concentration continuously as the OLR increased, producing an earlier inhibition due to the drop of pH at 7.5 kg VS/m^3/day.

Compared to mesophilic, thermophilic processes are also highly sensitive to ammonia, as the percentage of FA found in the media increases with temperature (Hansen et al., 1998). Hansen et al. (1998) used swine manure with TAN concentration of 6 g/l in continuously stirred tank reactors at four different temperatures of 37, 45, 55 and 60 °C, obtaining decreasing methane yields of 0.19, 0.14, 0.07 and 0.02 m^3/kg VS as temperature was increased. The increasing temperature in the ADs raised FA concentration from 0.75 to 2.6 g/l, which became toxic for the methanogenic bacteria as confirmed by the decrease of the apparent specific growth from 1 to 0.67 when FA increased from 1.1 to 1.3 g/l.

The need for a more exhaustive process control to maintain stability makes implementation of thermophilic dry AD less popular, although greater efficiency than for mesophilic assets can be achieved if process is studied in detail.

2.3.2. pH

Literature discussing the optimum operating pH ranges in dry AD is scarce, and hence more research on the area can favour process understanding. This being true, existing reports in wet AD can be extrapolated to dry AD, as the biological steps responsible for the process are similar. Optimum pH values for digestion have been reported to be between 6.8 and 7.2 (Ward et al., 2008), with microorganisms responsible for the different digestion stages presenting different optimum pH values. For the hydrolysis and acidogenesis steps pH optimum values are regarded to be between 5.5 and 6.5, but not all authors agree on the same values. Yu and Fang (2002) studied the acidification of dairy waste in continuous mesophilic digesters and found an increase of the dairy pollutants degradation when pH was in the range of 4 to 5.5, with most of them being converted to VFA. It was also reported that a further increase to pH 6.5 increased degradation slightly, but resulted in a lower VFA production due to the increased methane production. Kim et al. (2003) used synthetic sludge in batch reactors at mesophilic and thermophilic conditions operating with pH 4.5, 5.5, and 6.5 and obtained a maximum hydrolysis and acidogenesis rate at pH 6.5 for both mesophilic and thermophilic conditions. Higher rate of hydrolysis was observed at thermophilic conditions, with a peak of VFA concentration at 4 days instead of the 11 days showed at mesophilic but being more sensible to pH variations.

Methanogen activity is considered to be higher, between 6.5 and 8.2 with a maximum at 7.0 (Mao et al., 2015). Latif et al. (2017) studied the influence of low pH on continuous anaerobic digestion of sludge, at pH values from 5 to 7, achieving maximum methane production at pH 7 and observing a drop in production as pH was decreased, with a 88% reduction at pH 5.5. Similar results were obtained by Jain and Mattiasson (1998), who found optimum methanogenic values at pH 7 when digesting pulp industrial waste water, with marginal productions of methane as the pH was reduced to 5.5, and no production at 4 and 4.5. Other examples of pH-related problems are at values greater than 8. Kadam and Boone (1996) reported the complete inhibition of the methanogens at TAN concentrations of over 5 g/l at pH 7.5, and only at 1.6 g/l when pH was 8.5, as FA formation is more favoured at this pH.

Once methane production is stabilized the pH should be maintained between 7.2 and 8.2 (Karthikeyan and Visvanathan, 2013), as process compounds like VFA are specially toxic below pH 7 (Ward et al., 2008) and at pH values higher than 8.2 the ammonium equilibrium is displaced towards the more toxic FA form (Hansen et al., 1998). In one of the available studies about dry AD, Di Maria et al. (2017) reported an initial pH of 6.5 in dry batch AD of OFMSW when hydrolysis and acidogenesis were predominant at the beginning of the digestion and VFA accumulated to over 12,000 mg/l. This was followed by an increase to a pH of 8 when methanogenic activity reduced the VFA concentration to 1000 mg/l.

The results in wet AD can be mostly extrapolated to dry AD due to the similarity of the microbial processes. However, these do not account for specific characteristics of dry AD, like the mass transfer problems and the deficient mixing linked to the high TS solids content in the ADs. This lack of heterogeneity in the system can lead to different conditions in different parts of the ADs which can even lead to localised inhibition, pointing out the necessity of further research in dry AD.

The buffer capacity of the digestate in the ADs is usually measured using alkalinity values. Alkalinity provides resistance to big and sudden changes in pH, through the equilibrium between carbon dioxide and bicarbonate in the media (Ward et al., 2008). The buffering capacity of the system is proportional to the concentration of bicarbonate and can be indicative of process performance. A drop of the alkalinity can show accumulation of VFA before the pH is affected (Veluchamy et al., 2019). Alkalinity of the system can be modified in different ways, adding bicarbonate or bases, reducing OLR, increasing HRT or modifying the I:S ratio (Ward et al., 2008). Some examples are found in literature in dry AD, like Kim and Oh (2011) reported an increase in alkalinity from 7000 to 8000 mg $CaCO_3$/l when the HRT in a continuous digester was increased from 30 to 40 days, producing an increase in pH from 7.1 to 7.6. Ağdağ and Sponza (2005) also studied the effects of alkalinity after 65 days of batch anaerobic digestion of organic solid wastes in a leachate bed reactor with recirculation. One reactor was operated without alkalinity addition, while the others had bicarbonate additions of 3 and 6 g/l/d. The results showed higher operating pH as the alkalinity addition increased, with pH values of 6.5, 7.2 and 7.3 for each reactor, with lower TAN and VFA at the highest alkalinity and a production of 28% more methane; showing the positive effect of the alkalinity in the system.

These results show the important relationship between alkalinity, pH and operational parameters when operating dry AD systems. An example of this is the control of the OLR fed in continuous dry digestion, which could be increased further when running dry ADs if pH is in the adequate range for operation, and alkalinity is high enough to buffer a peak of VFA production.

2.3.3. Inhibitors

Inhibitory problems are common to both dry and wet AD, with dry AD systems more prone to inhibitors accumulation. This is linked to the high OLR and TS content and the low or null mixing, which result in poor homogenisation (Abbassi-Guendouz et al., 2012) and facilitate accumulation of inhibitors like fatty acids and ammonia (Ajay et al., 2011; Chen et al., 2008; Fernández-Rodríguez et al., 2014). At the same time, dry AD has higher tolerance to inhibitors (Dong et al., 2010; Fagbohungbe et al., 2015; Nagao et al., 2012), and can operate at higher concentrations of VFA or ammonia, as the inhibitors are localized due the poor diffusion in the ADs and frequently do not affect the entire reactor volume.

2.3.3.1. Fatty acids. Short chain fatty acids, also known as VFA, are intermediate compounds produced in the hydrolysis step, consequence of the breaking down of more complex structures like long chain fatty acids. The main VFA present in the media during the AD process are acetic, butyric and propionic acids, which are commonly accumulated at the start up period in the ADs (Massaccesi et al., 2013). Inhibition of the AD process occurs when VFA are produced in the hydrolysis step at a faster rate than they are assimilated by acetogenesis or methanogenesis, which results in a pH drop and inhibition of the methanogenic archaea (Guendouz et al., 2010). Generally, the inhibitory effect of VFA starts at levels of more than 2000 mg/l for acetic acid or 8000 mg/l for total VFA (TVFA) (Karthikeyan and Visvanathan, 2013). Kusch et al. (2012) reported a drop in pH when VFA production peaked at the beginning of the run when digesting MSW in dry batch ADs with different percolate reticulation strategies, only observing an increase to a stable pH value of 7.5 when the VFA concentration in the different ADs dropped below 2000 mg/l (Fagbohungbe et al., 2015). The low or null mixing conditions at which dry ADs are operated and the high TS content, can frequently lead to poor solid liquid mass transfer and accumulation of VFA in some localized areas, not affecting the totality of the methanogenic archaea but producing localized inhibition (Dong et al., 2010). This lack of diffusion and contact often contributes to the instability of the process, and contributes to the longer reaction times required in dry AD, but some authors (Fagbohungbe et al., 2015) reported some benefits. The poor diffusion through the media can in practice mean that dry AD can be operated at higher VFA concentrations than in wet AD, as VFA are getting in contact with methanogens in a steady and slow flux, avoiding the pH shock and inhibition.

Different effects on the process are observed at both batch and continuous operations, as a VFA peak on batch AD is expected at the beginning of the process as hydrolysis and acetogenesis are taking place, while peaks in continuous systems are not desirable, and can lead to the total failure of the digestor. The operational approaches used to palliate the accumulation of VFA are also different. For batch system the most common solutions against VFA accumulation are: (i) an increase of the I:S ratio or (ii) percolate recirculation. On this note, Hashimoto (1989) increased the inoculum content from 10 to 90%, avoiding as a result VFA accumulation and increasing pH from 4.9 to 7.6 in batch reactors. On continuous systems the main parameter used is the reduction of the OLR, as a reduction on the feed can help methanogens to consume existing VFA in the digestor. Fernández-Rodríguez et al. (2014) reported the reduction of the OLR as an effective solution for continuous dry AD, achieving a VFA reduction from 227 to 58.7 mg/l when OLR was reduced from 20 to 12.5 kg/m^3/day.

2.3.3.2. Ammonia. Proteins are the main source of nitrogen in the process and, although nitrogen is essential to microbial growth, high concentrations of nitrogen can create inhibitory problems (Koster and Lettinga, 1988). The most common forms of inorganic nitrogen present are free ammonia (FA) and ammonium (NH_4^+), where the sum of both is known as total ammonia nitrogen (TAN). FA values between 300 and 800 mg/l are commonly reported as inhibitory (Duan et al., 2012; Gallert and Winter, 1997; Yabu et al., 2011) while ammonium is tolerated at higher values of 1500 to 3000 mg/l (Appels et al., 2011; Chen et al., 2008; Dong et al., 2010), with other studies (Koster and Lettinga, 1988; Nakakubo et al., 2008) reporting no inhibition at total ammonia concentrations higher than 4000 mg/l. The higher resistance to inhibition reported by some authors is explained by the adaptation of the microorganisms to higher ammonia concentrations. Nakakubo et al. (2008) reported no inhibition when digesting pig manure with solid fractions separated from pig manure at thermophilic conditions with TAN concentrations of 4.6 g/l, with a progressive methane reduction until 50% when concentration was increased to 11 g/l.

Methanogens are the most prone archaea to ammonia inhibition, and their inhibition may cause a pH drop due to VFA accumulation in the ADs. Mechanisms on how ammonia toxicity occurs were explained by Kayhanian (1999), who identified two potential inhibition mechanisms. One is the inhibition of the methane synthesizing enzyme directly by the ammonium ion, and the second is the diffusion of the hydrophobic FA molecule passively into the cell causing proton imbalance or potassium deficiency.

The distribution of FA and ammonium in the media is governed by their chemical equilibrium and influenced by pH and operating temperature (Fricke et al., 2007), where FA concentration, which has the greater inhibitory effect, is increased as pH and specially temperature are increased. The higher FA at higher temperatures makes thermophilic processes more prone to ammonia inhibition. Yirong et al. (2017) studied ammonia toxicity in mesophilic and thermophilic wet AD reactors using FW rich in nitrogen, reporting stable operation and methane production at mesophilic conditions with concentrations of TAN over 4 g/l. However, at thermophilic conditions the VFA started to accumulate after the TAN reached 3.5 g/l, equivalent to 0.85 g/l FA, maintaining the pH stable until the TAN reached 5 g/l, when the accumulation overcome the buffering capacity of the system and the reactor failed.

Different strategies have been used to overcome ammonia accumulation, like the reduction of OLR or the use of co-digestion with other materials, using carbon rich wastes like cardboard or paper to increase the C/N ratio (Zhang et al., 2012), especially when nitrogen rich feedstocks like OFMSW or FW were used.

Ammonia inhibition is one of the main problems linked to the digestion of organic wastes in dry AD, increasing the need of close monitoring. Knowledge from wet AD can be mostly extrapolated, but it is necessary to understand dry AD specifics, as higher ammonia concentrations are frequently tolerated

A greater understanding of inhibition mechanisms in dry AD and the impact of localised inhibition on the process overall performance would allow the use of feedstock and operational parameters to control inhibitors accumulation. This would then transform thermophilic dry AD into a viable option and allow mesophilic dry AD to operate at higher treatment capacities (OLR).

3. Conclusions

Dry AD systems are suitable processes to treat organic wastes like OFMSW, but the perceived complexity of operation currently limits implementation. Understanding the impact that operational parameters have on performance and how can allow stable operation (avoid inhibition) is hence critical. Main general gaps in literature are linked to capacity optimisation against I:S ratios and understanding of localised inhibition mechanisms. Bed compaction, percolate regime and bulking agents use are needed in batch operation, while in continuous regimes mixing and feedstock effect need deeper understanding. Only by combining knowledge on feedstock and process operation will become dry AD a stable and profitable system for green economy.

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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Appendix A. Supplementary data

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