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Performance of a Commercial-Scale DiCOM™ Demonstration Facility Treating Mixed Municipal Solid Waste in Comparison with Laboratory-Scale Data

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ABSTRACT: The current paper describes the performance of a commercial-scale (20,000tpa) demonstration facility of the DiCOM™ process, a biological treatment for the organic fraction of municipal solid waste (OFMSW). The 21-day process combines aerobic composting and high-solids (30% DM), thermophilic (55°C) anaerobic digestion (AD), within a single vessel. Mechanically sorted OFMSW, derived from mixed household MSW (324t), was exposed to sequential aerobic/anaerobic/aerobic treatment. The AD, initiated by adding anaerobic inoculum from a previous trial, was stable (without pH intervention) and the onset of methanogenesis, rapid (< 3h). Volatile fatty acids formed during AD (including propionate) were exhausted prior to reuse of the inoculum. As measured by an electron flux from solids to gaseous end-products, AD accounted for the greatest portion of solids degradation (86% = 160 m^3 CH_4/dry t OFMSW). However, unlike laboratory trials, limited degradation occurred during initial aerobic treatment. The discharged solids were classified as a composted soil conditioner.

Keywords: Solid waste; Thermophilic anaerobic digestion; Composting; Renewable Energy; Biogas

1. Introduction

Increases in world population density and associated waste production, coupled with dwindling available land, increased fuel/transport costs and the need for environmentally sustainable waste treatment, has highlighted the need for close-to-source waste treatment facilities having a small footprint. Both, aerobic composting (Castaldi et al., 2005; Gajalakshmi and Abbasi, 2008; Golueke and Diaz, 1996; Rosen et al., 1993; Veeken and Hamelers, 2002) and anaerobic digestion (Braber, 1995; Fricke et al., 2005; Mata–Alvarez et al., 2000; Ostrem et al., 2004) have been described as biological alternatives for the treatment of this waste. The combination of thermophilic anaerobic digestion with in-vessel composting in a single vessel is one approach that aims at profiting from the benefits of anaerobic digestion (energy recovery) and composting (low odour compost as end-product). One such process has been tested and optimised at laboratory-scale (Walker et al., 2006a, 2006b). This process, termed DiCOM^{TM} has now been constructed as a commercial-scale (20,000 tpa) process for the treatment of high solids (20-40 % DM), thermophilic (55 °C) anaerobic digestion (AD) combined with in-vessel composting of OFMSW within a single vessel.

The purpose of this paper is to: test whether the performance of gas production and composting efficiency described from laboratory experiments (Walker et al., 2006a, 2006b) and tested at pilot scale (8 m^3, unpublished) could be reproduced at full-scale; and investigate the effect of higher solids to liquid content on process performance, which is caused by the higher compaction occurring at the larger scale.

In particular the second point is of scientific and applied interest. Due to the vessel

height of 22 m, a gradient of compaction over the height of the vessel would be expected, resulting in up to an estimated 1100 kg/m^3 of solids (at 50% moisture) at the point of highest compression within the vessel. Materials consolidation has been reported to decrease the pore space within a solid matrix by reducing, or ultimately eliminating, air channels between the solid particles (McCartney and Chen, 2001; Richard et al., 2004). Consequently, during the anaerobic digestion phase of DiCOM^{TM} operation, greater consolidation will decrease the volume of water that can penetrate into these pores.

Preliminary data has suggested that the maximum anticipated consolidation in a fully loaded DiCOM^{TM} reactor could result in an approximate 40 % decrease in pore space. Thus, the solid to liquid ratio would increase from approximately 750 kg/m^3 with no compaction, as is the case at the top of the vessel (and as previously recorded in laboratory-scale experiments), to 2750 kg/m^3 at the point of greatest compaction. The increased compaction can be equated directly to 4 times less water available for the anaerobic digestion process within the regions of material exposed to maximum consolidation. With less water available, the release of fermentation products, such as volatile fatty acids, will more readily result in undesirable elevated concentrations and increased risk of process failure. This paper outlines the performance of one of a series of commissioning trials of the commercial-scale DiCOM^{TM} demonstration facility.

2. Materials and Methods

2.1. Plant operation

The DiCOM™ demonstration facility was constructed at the site of the Western Metropolitan Regional Council (WMRC) Waste Transfer Station (WTS) in Perth,

Western Australia (W.A.). The plant, which consisted of a proprietary mechanical sorting system (rotating trommel), conveying arrangements and a single DiCOM^{TM} bioconversion vessel (700t OFMSW capacity), was commissioned in February 2009.

During normal WMRC waste collection operation (6 to 10 July 2009), all mixed domestic MSW collected from the western suburbs of Perth, W.A., (324 t) was delivered to the WMRC WTS. Upon delivery, the waste was moved, via a skid loader (≈15 t/h), onto a conveyer belt, where oversized objects (microwave ovens, computers etc.) were removed by hand, prior to entering the DiCOM^{TM} mechanical sorting system (Fig. 1). Fresh water (≈ 28 L/min) was introduced to the trommel to reduce dust and improve organic separation efficiency. Oversized objects generated a reject stream (4 t) that was directed to landfill. All domestic MSW received was processed on a daily basis, with the captured organic fraction (OFMSW) transported, by the conveying system into the DiCOM^{TM} bioconversion vessel.

Ambient air was drawn into the reactor during loading to ensure the reactor headspace remained aerobic during the loading period. At the conclusion of daily waste processing the reactor was sealed and the contents exposed to an aeration regime designed to avoid channelling of air as follows: Pressurised air was introduced into the vessel to raise the internal pressure to a predetermined set-point at which time the air flow was stopped. The over-pressure was maintained for a “soak” period prior to being released (Fig. 2). This aeration cycle was repeated continuously overnight during loading and during post-anaerobic treatment. The odourous air generated was treated by way of acid scrubbing and activated carbon to remove odour. The aeration cycle, and its frequency, was managed automatically by distributed control system (DCS) feedback control.

At the conclusion of loading, the reactor was sealed and the headspace flushed with an inert gas, in this case nitrogen (N_2), to create an oxygen-free atmosphere. The OFMSW (C:N of 33:1; moisture content of 56 %; VS content of 80 %; 8 % protein; 4 % lipids; 32 % cellulose; 7 % hemicellulose; 12 % lignin) contained within the reactor was then fully submerged with an anaerobic liquor/inoculum (320 m^3) from a previous DiCOM™ trial. The liquor was recirculated via an external heat exchanger to maintain the temperature (55 ± 3 °C) and to improve process monitoring and control. The biogas generated was characterised (Advanced Optima Continuous Gas Analyzer AO2040) and quantified (ST98 FlexMASSter™ mass flow meter) before being flared. After 11 days of anaerobic digestion (day 16: Fig. 2), the anaerobic liquor was drained and stored anaerobically for use as inoculum in subsequent trials. The solids were dewatered anaerobically (to 55 % moisture) prior to the vessel headspace being transitioned from being methane (CH_4) rich (65 %) to aerobic using an inert gas (N_2) and air. At this time, cyclic air pressurisation was again initiated. During periods of aerobic treatment, the solids were mechanically dewatered (to 55 % moisture) to avoid the solids at the base of the bioconversion vessel becoming waterlogged and ensure aerobic microbial activity within the solid matrix. Water reclaimed during dewatering was collected and added to the stored anaerobic liquor to be used as part of the process water system. The composted end-product was removed after 21 days of DiCOM™ processing.

The plant was monitored and operated by a dedicated, purposed built, computer based DCS via a process field bus (PROFIBUS) interface. pH, redox, liquor and solids temperature, hydrostatic, air and biogas pressure, gas and liquid flow rates, liquid and solid fill levels, O_2, CH_4 and CO_2 concentrations, motor current draw and overall plant

electricity consumption were computer monitored, logged and used as control parameters.

2.2 Chemical analysis

Liquid samples collected for analysis (NH_4^+, volatile fatty acid (VFA)) were centrifuged immediately (14,000 rpm for 10 min) and the supernatant stored (-20 °C) prior to analysis. Volatile solids content of solid samples were determined according to standard methods (American Public Health Association, 1992: Method 2540E) on composite samples (20 g), collected hourly while loading solids to, and unloading from, the reactor. VFA were analysed as described previously (Walker et al., 2009). The grit/heavy (inert) fraction was separated by agitating OFMSW in water (solid:liquid = 1:10 w/v) and decanting the suspended slurry through a 1mm square mesh. MSW characteristics were analysed by ChemCentre, Bentley, W.A.. Process water analysis was performed by ALS Water Resource Group, Malaga, W.A.. Compost samples were analysed for compliance with the Australian Standard for Composts, Soil Conditioners and Mulches (AS4454–2003) by RichGro Pty Ltd, Jandakot, W.A..

2.3. Mass balance

The incoming MSW, end-product and reject streams were weighed (± 0.1 t) and liquor volumes measured (± 0.1 m^3). The moisture content of the solids was determined by heating composite grab samples (2 kg), collected hourly while loading solids to, and unloading from, the reactor, at 105 °C until constant mass was attained. The moisture content of the reject stream was determined by air-drying a sample (160 kg) under ambient conditions. The moisture content of the input MSW was estimated from the moisture content of reject and OFMSW steams and the mass of water added to the

trommel. The mass of biogas was determined from biogas composition and flow rate and also includes the mass of carbon contained within the CO_2 produced during aerobic treatments. The mass of water lost from the system through evaporation was not considered in the mass balance.

2.4. Laboratory-scale trial

Mechanically sorted OFMSW, obtained from mixed MSW, was processed and prepared as previously described (Walker et al., 2009). The OFMSW was treated in an insulated, cylindrical, 7 L high temperature PVC, computer controlled, laboratory-scale DiCOM™ reactor as described by Walker et al., (2006a). The reactor was operated as a sequencing batch reactor capable of providing a combination of in-vessel composting and anaerobic digestion. After the reactor was loaded with OFMSW and the reactor sealed, the trial consisted of 5 days of aerobic treatment, followed by 7 days of thermophilic (55 °C) anaerobic treatment and finally 7 days of aerobic maturation. Pressurised aeration that mimicked full-scale operation was operated during aerobic operation. A complete description of operational parameters can be found elsewhere (Walker et al., 2006a).

3. Results and Discussion

3.1. Mass and water balance of a Full-scale DiCOM™ trial

In order to verify the material flows through the commercial-scale embodiment of the DiCOM™ process, a mass balance was performed. For the described trial, approximately 200 t of the organic stream was introduced into the bioconversion vessel over a period of 4 days (Fig. 3). Three weeks of processing resulted in an approximate 40 % reduction in organic solids (volatile solids, VS) (Fig. 3), with the solids being

converted into gaseous end-products (CH_4 and CO_2) from aerobic and anaerobic treatment.

The water mass balance, accounting for 94 % of input water, showed that, of the total water introduced to the process (145 t), approximately half of the water exited the system in reject (24 %, 35 t) and compost (26 %, 38 t) streams. The remaining half of the introduced water (approximately 60 t) was found to accumulate within the process water recirculation system (Fig. 3). As a result, continued operation of the system (i.e. fresh water being added to the trommels), would require discharge of process water from the system. The discharge of process water is undesirable. It would not only result in the production of a wastewater stream requiring treatment but the loss of buffering capacity and methanogenic inoculum. However, the quantity of wastewater produced could be minimised if process water was used during mechanical separation rather than fresh water.

3.2. Overview of performance of an optimised laboratory-scale DiCOM™ trial

An optimised laboratory-scale DiCOM™ trail was conducted on mechanically sorted OFMSW derived from mixed MSW. The overall solids degradation rate was expressed as an electron flux, defined as the rate at which electrons are removed from the OFMSW (Walker et al., 2009). Considering that an O_2 molecule, can accept four electrons and that a CH_4 molecule represents eight electron equivalents, the oxidation of organic molecules into the final gaseous end products, CO_2 and CH_4, can be directly compared. In terms of overall degradation rates, the anaerobic phase accounted for the greatest portion of solids degradation, with initial aeration, anaerobic and aerobic maturation treatment phases accounting for 38, 51 and 11 % (14646, 19395 and 4183

mol electron equivalents/dry kg OFMSW) of the total electron flow (38224 mol electron equivalents/dry kg OFMSW), respectively (Fig. 4, dotted line). The laboratory data set chosen here is typical of that observed in more than 10 laboratory trials, some of which have been published (Walker et al., 2006a, 2006b, 2009), and represents the reproducible behaviour and general trends of the process.

CH_4 generation from the laboratory-scale DiCOM^{TM} trial (0.10 m^3/kg VS) was in the same order of magnitude of other laboratory-scale thermophilic anaerobic systems recently reported in the literature (Table 1). Martín-González et al., (2011) reported a methane production rate (MPR) of 0.36 m^3/kg VS from a continuous (organic loading rate (OLR) = 4.3 kg VS/m^3/d), wet anaerobic digestion of source collected OFMSW (SC-OFMSW). Fdez.-Güelfo et al., (2011) found a MPR of 0.10 m^3/kg VS from a semi-continuous (OLR = 11.8 kg VS/m^3/d), dry anaerobic digestion of synthetic OFMSW. The variation in CH_4 yields, between these three systems, can be attributed to differences in feed material composition (Martín-González et al., 2011) and SRT (Fdez.-Güelfo et al., 2011) and variations in operational parameters. For example, during DiCOM^{TM} operation a significant portion of the electron flow (38 %) is diverted to oxygen rather than to CH_4.

3.3. Expected differences between full-scale and laboratory-scale operation

The laboratory-scale DiCOM^{TM} reactor was not a to-scale reproduction of its full-scale counterpart nor could the laboratory data reflect industrial-scale parameters such as materials compression, airflow wall effects, matrix channelling (Mason and Milke, 2005), OFMSW particle size and system heat losses. Consequently, it was anticipated that laboratory data would not be directly (1:1) transferable to the full-scale system but

trends observed at laboratory-scale could be recognised at larger scale (Körner et al., 2003).

Significant changes between laboratory and commercial operation were anticipated, which include:

1. A decrease (approximately 70 %) in intensive aeration during the loading period could result in less aerobic solids degradation occurring during the loading phase of the treatment process, posing a greater demand on the anaerobic phase to stabilise the end-product. The laboratory reactor was fully loaded at the start of the treatment cycle and the OFMSW intensely aerated for a period of 5 days. However, the commercial-scale reactor was loaded over 4 consecutive days and intensive aeration only provided each night.
2. An approximate doubling in the maximum consolidation of wet OFMSW (578 to 1100 kg/m^3), which could result in reduced matrix porosity and consequently, a reduction in air penetration and difficulty in maintaining aerobic conditions during loading.
3. The higher compaction ratio in full-scale led to the use of less water, resulting in a higher dry solids content (≈ 30 %) than at laboratory-scale (18 %), which increases the likelihood of VFA accumulation and process acidification.

3.4. Comparison of solid degradation in laboratory and full-scale processes

Overall solids degradation rate in the full-scale process was measured as an electron flux (Fig. 4, solid line) and could be directly compared to the performance of the laboratory scale reactor. In contrast to laboratory operation the first aerobic phase did not result in substantial degradation of organics. This observed lack of aerobic activity,

when compared to laboratory performance, may be explained by the less intensive air supply provided to the solids at commercial-scale, which originated from:

- the reactor being gradually filled, such that the material loaded during the last day was exposed to only 14 hours of aerobic treatment;
- the lower air penetration through the more consolidated material; and
- the full-scale reactor only being intensively aerated during night hours (14h/24h).

As expected, when compared to laboratory data, the lower oxygen consumption at full-scale reserved more of the reducing power, contained within the solids, for methane production during the subsequent anaerobic treatment phase, resulting in about 74% more total methane produced (33.7 mol electron equivalents/dry kg OFMSW compared to 19.4 in laboratory-scale). The onset of methanogenesis at small and large-scale was in both cases rapid, with CH_4 being produced within hours (4 and 3 h, respectively) of the introduction of the methanogenic inoculum (Fig. 5). Even though at laboratory-scale peak CH_4 production was attained more rapidly (≈ 2 days), both reactors produced similar maximum rates (Fig. 5: 1.4 and 1.5 m^3 CH_4/t VS/h). Exhaustion of easily available methanogenic substrates was achieved more rapidly in the laboratory trial as a direct consequence of the enhanced aerobic degradation during the initial 5 days of treatment. Inline with laboratory data, the anaerobic phase of the commercial facility accounted for the greatest portion of solids degradation, with initial aeration, anaerobic and aerobic maturation treatment phases accounting for 1, 86 and 13% (282, 33724 and 5103 mol electron equivalents/dry kg OFMSW) of the total electron flow (39109 mol electron equivalents/dry kg OFMSW), respectively.

3.5 VFA metabolism

From laboratory results it was generally found that VFA accumulated temporarily up to 120 mM (7.5 g/L), which is known to be in the range that can potentially inhibit methanogenesis (Dogan et al., 2005; McMahon et al., 2004; Siegert and Banks, 2005). The lower water content in the full-scale trial can be seen to equate to a higher (1.3 times) proportion of organics being available per L of process water. Hence VFA accumulation was expected to be more substantial than in laboratory experiments, particularly when considering the potential VFA generation from organics that had not been aerobically degraded during the loading phase (37 % of the electron flow in the laboratory trial). However, the anaerobic digestion phase of the DiCOM^{TM} process was found to be stable (Fig. 6B). Even though VFA accumulated during the initial days of anaerobic digestion to about 350 mM (24.4 g/L) (Fig. 6B), the buffer capacity of the anaerobic liquid was adequate to avoid acidification, maintaining the pH between 6.8 and 7.8 without requiring process control intervention.

Rapid acetate removal (Fig. 6B: 7.5 mM/h = 0.45 g/L/h) coincided with the maximum CH_4 production rate (Fig. 5: 1.2 m^3/h/t dry OFMSW/h = 1.5 m^3/h/t VS = 1.5 L/L/d) and indicated that significant acetoclastic organisms (encompassing direct acetoclastic methanogenesis and syntrophic acetate conversion) were present within the reactor after 3 days of anaerobic treatment. It is known that the start-up of thermophilic anaerobic digestion is the most significant drawback of thermophilic anaerobic digestion, due to the limited availability of thermophilic inocula (Suwannoppadol et al., 2011). Without a source of thermophilic inoculum, the start-up period of thermophilic anaerobic digestion can be prolonged, with Ahring (1994) reporting up to one year for a reactor to reach steady state. Suwannoppadol et al., (2011) found that, for thermophilic anaerobic digestion of MSW, without added inoculum, no substantial CH_4 production occurred

within 3 days, and the system required approximately 5 days for acetate consumption to exceed production. In the current study, CH_4 was produced (0.1L/L/d) within 24 h with significant acetate consumption (> production) occurring within 3 days of the commencement of digestion (Fig. 5B). The rapid onset of acetoclastic activity, also noted during laboratory (Walker et al., 2006b) and pilot-scale (data not shown) trials, suggests that the reuse of liquor is an effective method to transfer a viable methanogenic culture.

3.6. Propionate metabolism

While acetate and butyrate were readily degraded within the reactor, propionate accumulated, which was not observed in the optimised laboratory trial (Fig. 6A). However this striking complete lack of propionate degradation has been observed in sub-optimally performing trials at laboratory (Walker et al., 2006b) and pilot-scale (data not shown). The phenomenon can be explained by acetate inhibition of propionate degradation (van Lier et al., 1993). Van Lier and colleagues found that propionate degradation, of a propionate fed methanogenic sludge, in a thermophilic up-flow anaerobic sludge bed (UASB), was severely inhibited by 50 mM (3 g/L) acetate.

Propionate degradation to its primary products, acetate, hydrogen and CO_2 is an energetically very poor reaction. The spontaneity (Gibb's Free Energy) is strongly affected by the concentration of the end products, in particular H_2 and acetate. Previous studies have documented the inhibitory effects of acetate (and hydrogen) accumulation on propionate degradation (Fukuzaki et al., 1990, Pind et al., 2003). Consequently, from the above considerations, it can be concluded that, during peak methane production, propionate accumulation within the commercial-scale DiCOM^{TM} reactor cannot be

avoided, as both H_2 and acetate levels render propionate degradation thermodynamically unfavourable.

For the process described here, non-degraded propionate at the end of an anaerobic batch is undesired, as the liquor drained from one batch is designed to serve as the inoculum for the anaerobic phase of the next batch. Continuous re-use of this liquor would lead to incremental propionate accumulation making the process unsustainable.

Laboratory trials indicated that propionate degradation could be readily accomplished subsequent to the anaerobic digestion of the solid material and following the depletion of acetate and the cessation of H_2 production from other metabolites, including butyrate. In-line with thermodynamic theory, propionate degradation required low hydrogen and acetate concentrations, the precise inhibiting concentrations being able to be calculated from the mole fraction of end products and substrates and the equilibrium constant of the propionate degradation reaction (Hoh and Cord-Ruwisch, 1996).

In this study it was attempted to apply the technique, proven at laboratory and pilot-scale, of degrading propionate externally to the digestion reactor. To implement this, at full-scale, a large reaction (storage) vessel was used in which the drained anaerobic liquor was stored under anaerobic conditions at 55 ºC for 14 days (Fig. 6B). To maximise the potential transfer of methanogenic organisms into this propionate-degrading vessel, the digested solids were mechanically dewatered under anaerobic conditions. Results show that incubation under thermophilic conditions provided propionate degradation (from day 23 Fig. 6B: 0.4 mM/h = 0.03 g/L/h) but, inline with the literature (Golkowska and Greger, 2010), the rate was slower than that of acetate

degradation. A significant time delay (day 16 to 22, Fig 6B) was evident, from the time the VFA laden anaerobic liquor was incubated in the storage vessel, before the propionate degradation commenced. This delay may be caused by residual acetate being present in the reactor, possibly resulting from the addition of regular aliquots of organic laden water that had been pressed from the aerated solids.

It can be concluded from the results that, as long as additional storage facilities that provide conditions to encourage thermophilic propionate degradation are in place, the risk of continued propionate build-up can be lowered. Also it appeared that propionate degrading bacteria were present in the liquor (Fig. 6: > Day 23) despite the apparent absence of propionate degradation (Fig. 6: < Day 20) raising the questions whether the propionate degrading syntropic bacteria may have developed by catalysing reactions other than propionate conversion. This would be an interesting topic of further study.

The levels of propionate-degrading populations have been reported to be low in stable digesters, with those having a history of very stable operation likely to be susceptible to failure during a sudden influx of organic material (McMahon et al., 2001). Conversely, systems that are subjected to periodic organic perturbations (batch fed systems) possess higher substrate utilization capacities and is especially true for propionate-degrading populations (Xing et al., 1997a, 1977b). As the batch operation of the DiCOM^{TM} process provides periodic organic perturbations to the microbial consortium contained within the anaerobic liquor it is anticipated that, over time, the propionate degrading capacity could be enhanced.

3.7. Commercial-scale methane production

CH_4 production resulted in 160 m^3/dry t OFMSW (0.22 m^3/kg VS), which is comparable to literature values for larger-scale thermophilic AD of OFMSW (Table 1). Using an electrical and heat conversion efficiency of 39.7 % and 74 %, respectively, the energy output from the CH_4 generated was 0.65 MWh and 2.6 GJ per dry t OFMSW. Using this theoretical electrical and heat energy production, and logged data for electrical and natural gas consumption for the current trial, an extrapolation of the energy requirements for the operation of a commercial-scale facility (55,000 tpa) can be made. Considering a commercial-scale facility consisting of 3 DiCOM™ bioconversion vessels, each processing 700 t of OFMSW (55 % moisture), the predicted electrical and heat requirements for operation (0.22 MWh and 0.30 GJ per dry t OFMSW) are less than the energy yield of the biogas. While this energy balance does not imply that the process being investigated is more efficient than other processes, it does suggest that the process can, in theory, be operated such that energy produced during process operation can meet its energy demand. This is one of the criteria that must be met for the energy neutral commercial operation of the process.

3.8. Commercial-scale end-product characterisation

As not all sorting technologies were installed for this trial (e.g. removal of glass, grit, etc.), the composted end-product contained physical and possibly chemical contaminants which were not quantified during analysis. A characterisation of the solid end-product against AS4454–2003 (a standard test for acceptable compost Standards Australia, 2003), classified the composted material as a composted soil conditioner as outlined in Table 2.

4. Conclusion

From this study it can be concluded that the DiCOM^{TM} process, which combines in-vessel thermophilic anaerobic digestion and composting processes, as previously tested at laboratory-scale, was reproducible at full-scale in a reactor treating 160 t of OFMSW. When compared to laboratory trials, the lower water content, caused by material consolidation, did not result in digester acidification, with the majority of VFA degraded within the anaerobic phase of the process (here 11 days). It was also found that establishing a customised reactor that degrades propionate external to the bioconversion vessel could prevent the anticipated problem of unsustainable propionate accumulation.

5. References

Ahring, B.K., 1994. Status on science and application of thermophilic anaerobic digestion. Water Sci. Technol. 30, 241–249.

American Public Health Association. 1992. *Standard Methods for the Examination of Water and Wastewater 18th Edition*. Greenburg, A.E., Clesceri, LS. and Eaton, A.D. (Eds.), American Public Health Association, Washington, D. C. USA.

Braber, K. (1995). Anaerobic digestion of municipal solid waste: a modern waste disposal option on the verge of breakthrough. Biomass Bioenerg. 9, 365-376.

Castaldi, P., Alberti, G., Merella, R., and Melis, P. 2005. Study of the organic matter evolution during municipal waste solid composting aimed at identifying suitable parameters for the evaluation of compost maturity. Waste Manag. 25, 209– 213.

Dogan, T., Ince, O., Oz, N.A., Ince, B.K., 2005. Inhibition of volatile fatty acid production in granular sludge from a UASB reactor. J. Environ. Sci. Health, Part A: Toxic/Hazard. Subst. Environ. Eng. 40, 633–644.

Fdez.-Güelfo, L.A., Álvarez-Gallego, C., Márquez, D.S., García, L.I.R., 2011. Dry-thermophilic anaerobic digestion of simulated organic fraction of Municipal Solid Waste: Process modeling, Bioresour. Technol. 102 (2), 606–611.

Fricke, K., Santen, H., Wallmann, R., 2005. Comparison of selected aerobic and anaerobic procedures for MSW treatment. Waste Manag. 25, 799–810.

Fukuzaki, S., Nishio, N. Shabayashi, M., Nagai, S., 1990. Inhibition of the fermentation of propionate to methane by hydrogen, acetate, and propionate. Appl. Environ. Microbiol. 56, 719–723.

Gajalakshmi, S., Abbasi, S.A. (2008). Solid waste management by composting: State of the art. Crit. Rev. Environ. Sci. Technol. 38, 311–400.

Golkowska, K., Greger, M., 2010. Thermophilic digestion of cellulose at high-organic loading rates. Eng. Life Sci. 10, 600–606.

Golueke, C.G., Diaz, L.F., 1996. Historical view of composting and its role in municipal waste management. In: M. De Bertoldi (Ed.). The Science of Composting: Part 2. Chapman and Hall, Glasgow, UK, pp3–14.

Hartmann, H., Ahring, B.K. 2006. Strategies for the anaerobic digestion of the organic fraction of municipal solid waste: an overview. Water Sci. Technol. 53, 7–22.

Hoh, C.Y., Cord-Ruwisch. R., 1996. A practical kinetic model that considers endproduct inhibition in anaerobic digestion processes by including the equilibrium constant. Biotechnol. Bioeng. 51, 597–604.

Körner, I., Braukmeier, J., Herrenklage, J., Leikam, K., Ritzkowski, M., Schlegelmilch, M. and Stegmann, R., 2003. Investigation and optimization of composting processes – test systems and practical examples. Waste Manag. 23, 17–26.

Martín-González, L., Castro, R., Pereira, M.A., Alves, M.M., Font, X., 2011. Thermophilic co-digestion of organic fraction of municipal solid wastes with FOG

wastes from a sewage treatment plant: Reactor performance and microbial community monitoring, Bioresour. Technol. 102 (7), 4734–4741.

Mason, I.G., Milke, M.W., 2005. Physical modelling of the composting environment: A review. Part 1: Reactor systems. Waste Manag. 25, 481–500.

Mata–Alvarez, J., Mace, S., Llabres, P., 2000. Anaerobic digestion of organic solid wastes. An overview of research achievements and perspectives. Bioresour. Technol. 74, 3–16.

McCartney, D., Chen, H., 2001. Using a biocell to measure effect of compressive settlement on free air space and microbial activity in windrow composting. Compost Sci. Util. 9, 285–302.

McMahon, K.D., Stroot, P.G., Mackie, R.I., Raskin, L., 2001. Anaerobic codigestion of municipal solid waste and biosolids under various mixing conditions—II: microbial population dynamics. Water Res. 35, 1817-1827.

McMahon, K.D., Zheng, D., Stams, A.J.M., Mackie, R.I., Raskin, L., 2004. Microbial population dynamics during start-up and overload conditions of anaerobic digesters treating municipal solid waste and sewage sludge. Biotechnol. Bioeng. 87, 823–834.

Ostrem, K.M., Millrath, K., Themelis, N.J., 2004. Combining anaerobic digestion and waste-to-energy. 12th North American Waste To Energy Conference (NAWTEC 12), May 17–19, Savannah, Georgia.

Pind, P.F., Angelidaki, I., Ahring, B.K., 2003. Dynamics of the anaerobic process: Effects of volatile fatty acids. Biotechnol. Bioeng. 82, 791–801.

Richard, T.L., Veeken, A., De Wilde, V., Hamelers, B., 2004. Air–filled porosity and permeability relationships during solid–state fermentation. Biotechnol. Prog. 20, 1372–1381.

1 Rosen, C., Halbach, T., Swanson, B., 1993. Horticultural uses of municipal solid waste
2 composts. Horttechnology 3, 167–173.
3 Siegert, I., Banks, C., 2005. The effect of volatile fatty acid additions on the anaerobic
4 digestion of cellulose and glucose in batch reactors. Process Biochem. 40, 3412–
5 3418.
6 Standards Australia, 2003. Australian Standard™ Composts, Soil conditioners and
7 Mulches. AS 4454–2003 (ISBN 0 7337 5040 0), Sydney, Australia.
8 Suwannoppadol, S., Ho, G., Cord-Ruwisch, R., 2011. Rapid start-up of thermophilic
9 anaerobic digestion with the turf fraction of MSW as inoculum. Bioresour. Technol.
10 102, 7762–7767.
11 van Lier, J.B., Grolle, K.C.F., Frijters, C.T.M.J., Stams, A.J.M., Lettinga, G., 1993.
12 Effects of acetate, propionate, and butyrate on the thermophilic anaerobic
13 degradation of propionate by methanogenic sludge and defined cultures. Appl.
14 Environ. Microbiol. 59, 1003–1011.
15 Veeken, A., Hamelers, B., 2002. Sources of Cd, Cu, Pb and Zn in biowaste. Sci. Total
16 Environ. 300, 87–98.
17 Walker, L., Charles, W., Cord–Ruwisch, R., 2006a. Performance of a laboratory–scale
18 DiCOM® reactor – a novel hybrid aerobic/anaerobic municipal solid waste treatment
19 process. In: Kraft, E., Bidlingmaier, W., de Bertoldi, M., Diaz, L.F. and Barth, J.
20 (Eds.), Proceedings of the Fifth International Conference of ORBIT Association on
21 Biological Waste Treatment, Sept.13–15, Weimar, Germany. Part 3, pp849–858.
22 Walker, L., Charles, W., Cord–Ruwisch, R., 2006b. The effect of direct transfer of
23 anaerobic inoculum on the performance of a laboratory–scale DiCOM® reactor.
24 Biomass and Waste to Energy Symposium, Nov. 29–Dec., 1 Venice, Italy.

Walker, L., Charles, W., Cord-Ruwisch, R., 2009. Comparison of static, in-vessel composting of MSW with thermophilic anaerobic digestion and combinations of the two processes. Bioresour. Technol. 100, 3799–3807.

Xing, J., Criddle, C., Hickey, R., 1997a. Long-term adaptive shifts in anaerobic community structure in response to a sustained cyclic substrate perturbation. Microb. Ecol. 33, 50–58.

Xing, J., Criddle, C., Hickey, R., 1997b. Effects of a long term periodic substrate perturbation on an anaerobic community. Water Res. 31, 2195–2204.

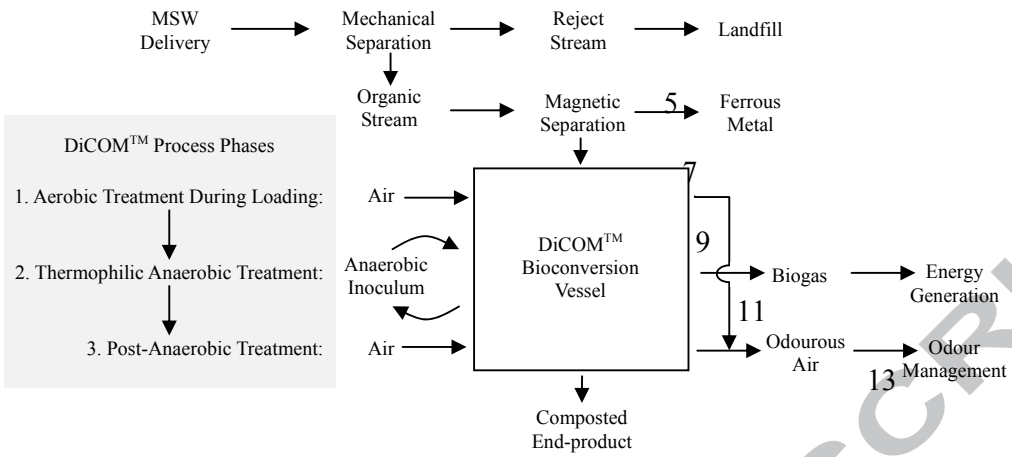

Fig. 1. Process flow diagram for a commercial-scale demonstration DiCOM™ plant.

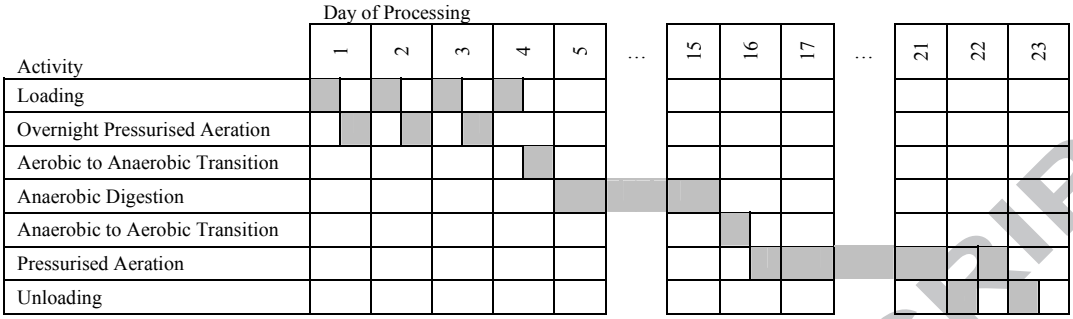

Fig. 2. Gantt chart indicating timing of activities during DiCOM^{TM} demonstration facility operation.

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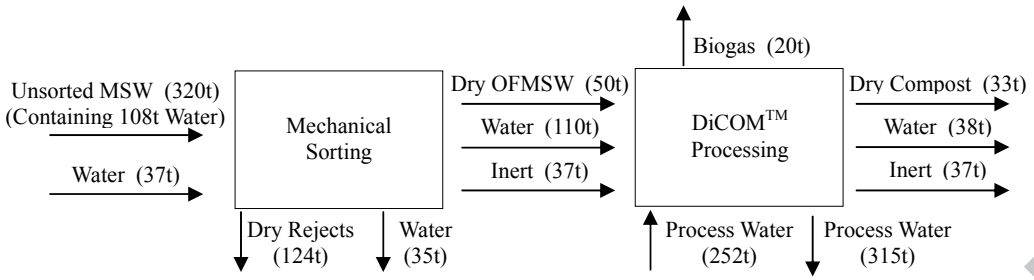

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4 **Fig. 3.**Mass balance for a commercial-scale DiCOM™ batch trial.

5

6

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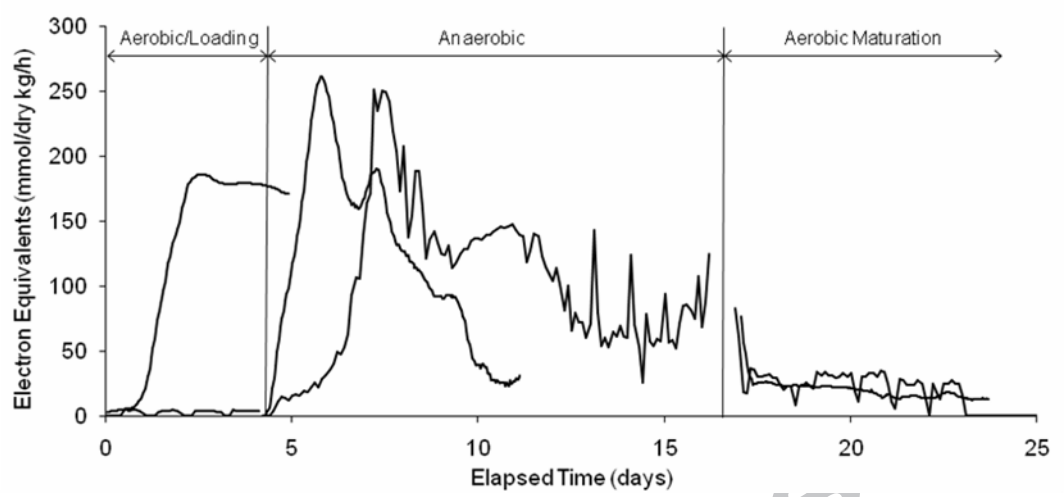

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3 **Fig. 4:**Comparison of electron flux for laboratory and commercial-scale DiCOM^{TM}

4 reactors. The timing for the laboratory-scale reactor data has been adjusted so that the

5 commencement of each phase coincides with that of the commercial-scale reactor.

6 Legend: (──) Commercial-scale reactor; (─ ─) Laboratory-scale reactor.

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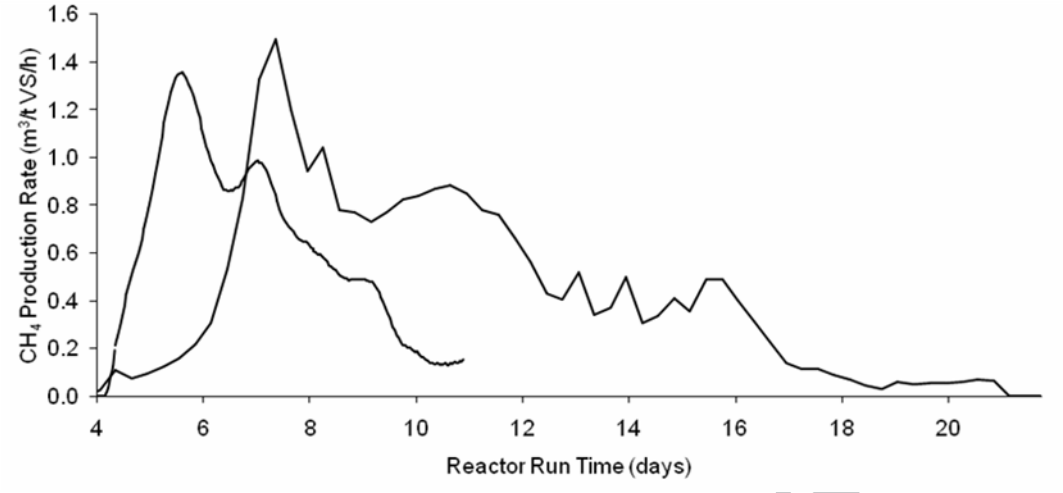

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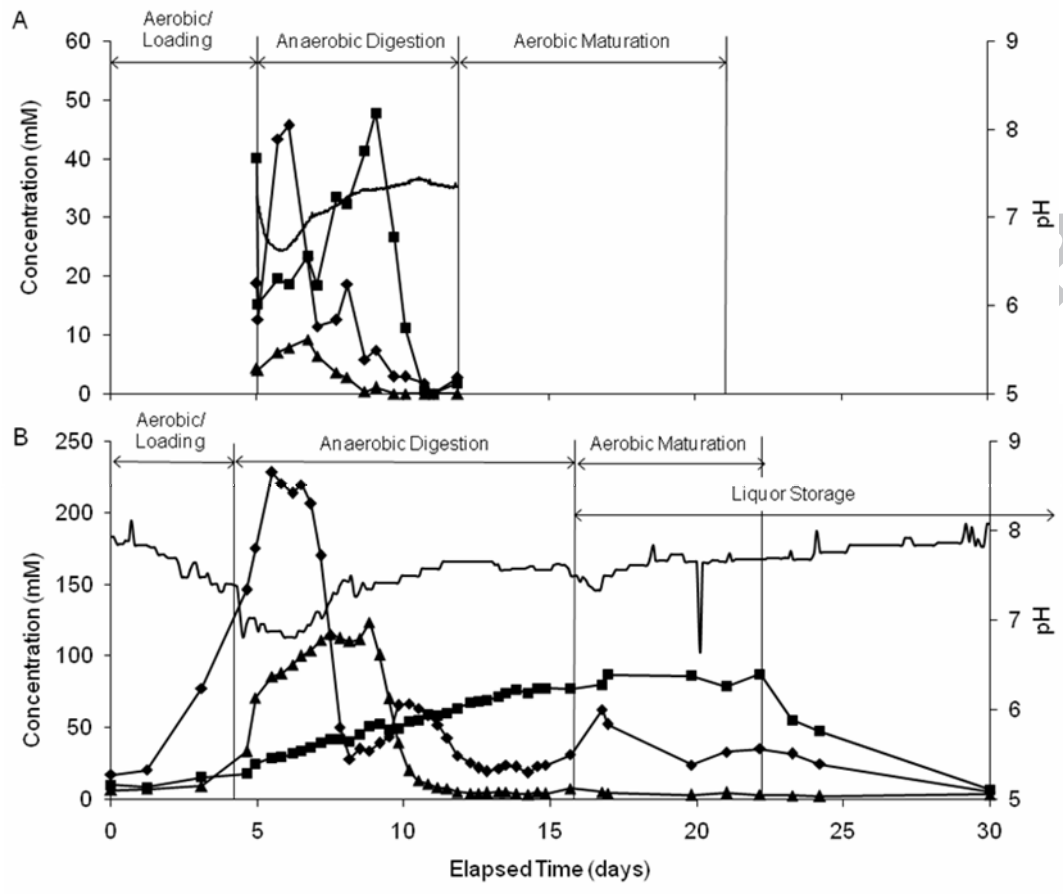
3 **Fig. 5.**Methane generation time course during the anaerobic phase of an optimised
4 laboratory (─ ─) and commercial-scale (──) DiCOM^{TM} batch trial. The timing for
5 the laboratory-scale data has been changed so that the commencement of the anaerobic
6 digestion phase coincides.

1



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3 **Fig. 6.**VFA and pH time course during (A) the anaerobic phase of an optimised

4 laboratory and (B) a commercial-scale DiCOM^{TM} batch trial.

5 **Legend:** (♦) Acetate; (■) Propionate; (▲) Butyrate; (──) pH.

6

Table 1. Process parameters and biogas yields of thermophilic (50–56°C) anaerobic digestion of OFMSW.

Process	Capacity (m^3)	Waste	TS During AD (%)	HRT/SRT (d)	Biogas Yield (m^3/kg VS)
DiCOM™ [a]	900	Mechanically-sorted OFMSW	17	12	0.44
BTA [b]	3.4	Source-sorted (SS) OFMSW	6–16	12	0.39
DRANCO [b]	56	Organic household waste - no paper	30–35	15–21	0.45
KOMPOGAS [b]	200	Fruit, yard & vegetable waste	15–40	13	0.39
SEBAC [b]	3 x 0.7	OFMSW (paper, yard, & food waste)	–	21	0.34
DiCOM™ [a]	0.007	Mechanically-sorted OFMSW	18	7	0.17
Wet Anaerobic Digestion [c]	0.005	SC-OFMSW	7	16	0.52
Dry Anaerobic Digestion [d]	0.005	Synthetic OFMSW	30	15	0.27

[a] Current study
[b] Hartmann and Ahring, (2006)
[c] Martín-González et al., (2011)
[d] Fdez.-Güelfo et al., (2011)

Table 2. Compost parameters as compared to the Australian Standard (AS 4454-2003).

Characteristic & Unit	Unit	Requirement From AS 4454-2003	Results
pH		5.0 to 7.5	7.45
Phosphorus, total	%	≤ 0.1 for P sensitive plants	0.314
Nitrogen, total	%	≥ 0.6	1.50
Organic matter content	%	≥ 25	42.5
Wettability	min.	< 7	< 1
Toxicity	mm	≥ 60	80
Particle size grading:			
Maximum size	mm	≤ 16	Soil conditioner
Tolerance	% mass	≤ 20% retained by sieve	
Moisture Content	%	Minimum 25	42
Self Heating	°C	≤ 40	< 40
Pathogen Test		Grade P1 Biosolids	
Phytophthora		N/A	Not detected
Pythium		N/A	Not detected
Salmonella sp.		< 1 Salmonella per 50g	Negative
Thermotolerant coliforms		< 100 MPN cfu/mL	< 100

Research highlights include:

- The performance of a lab-scale DiCOM™ process, was reproducible at full-scale.
- Higher solid to liquid ratio at full-scale did not result in process instability.
- Propionate degradation was not evident within the full-scale DiCOM™ reactor.
- Propionate was degraded outside the reactor averting buildup during liquor reuse.
- The DiCOM™ specific methane production was comparable to other reported systems.