Model assisted startup of anaerobic digesters fed with thermally hydrolysed activated sludge

D. J. Batstone, C. Balthes and K. Barr

ABSTRACT

This paper presents the use of the IWA ADM1 to predict and interpret results from two full-scale anaerobic digesters fed with thermal hyrolysate (waste activated sludge with a long upstream sludge age) from a Cambi hydrolysis process operating at 165°C and 6 bar-g. The first digester was fed conventionally—though intermittently, while the second was heavily diluted through a substantial component of the evaluation period (110 days). There were a number of important outcomes—related to both model application, and model predictions. Input and inert COD: mass ratio was very important, and was considerably higher than the 1.42 g g^{-1} used for biomass throughout the IWA activated sludge and anaerobic digestion models. Input COD: VS ratio was 1.6 g g^{-1}, and inert COD: VS ratio was 1.7 g g^{-1}. The model succeeded on a number of levels, including effective prediction of important outputs (degradability, gas flow and composition, and final solids), clarification of the substantial data scatter, prediction of recovery times during operationally poor periods, and cross-validation of the results between digester 1 and digester 2. Key failures in model performance were related to an early incorrect assumption of the COD: VS ratio of 1.42 g g^{-1}, and intermittent high acetate levels, most likely caused by inhibition, and rapid acclimatisation to ammonia. The acute free ammonia limit was found to be 0.008 M NH_3–N, while the chronic inhibition constant ($K_{I,NH3,ac}$) was 0.007 ± 0.001 M NH_3–N. Overall, this is a complex system, and application of the model added significant confidence to the initial operational decisions during an aggressive startup on an atypical feed.

Key words | activated sludge, ADM1, ammonia inhibition, degradability, model, thermal hydrolysis

D. J. Batstone (corresponding author)
Advanced Wastewater Management Centre,
The University of Queensland,
Australia
E-mail: *damienb@awmc.uq.edu.au*

C. Balthes
K. Barr
Brisbane Water, Brisbane City Council,
Oxley Creek Water Reclamation Plant,
Donaldson Road,
Rocklea Qld,
Australia
E-mail: *keith.barr@brisbane.qld.gov.au*

INTRODUCTION

Strict nitrogen limits in Australia have forced many large plants to move towards longer sludge ages, extended aeration, and hoarding of available carbon for denitrification. The impact of this is a lack of primary sludge for anaerobic digestion, as well as poor degradability of the activated sludge. While poorly degradable, it is still produced in large amounts, and is not suitably stabilised for agricultural use. At the same time, an emerging popularity for application of properly stabilised biosolids as agricultural fertiliser, and a stronger emphasis on energy recovery have enhanced the attractiveness of anaerobic digestion as a sludge stabilisation option. To consolidate these conflicting pressures, high energy (electrical, mechanical, or thermal) pretreatment methods, such as thermal hydrolysis, thermophilic pretreatment, sonication, and mechanical methods are being used prior to anaerobic digestion. Intensification of existing digesters is also a key reason for installing pretreatment.

Brisbane Water recently commissioned a thermal hydrolysis unit (Cambi AS) at their Oxley Creek Water Reclamation Plant, as part of an overall upgrade. The rationale behind competitive process selection is given in (Barr *et al.* 2008). This doubled the biosolids handling capacity and made reception from other facilities a viable

doi: 10.2166/wst.2010.487

option. The system was effectively moving from 70% primary/30% activated to purely activated sludge, with a doubling of biosolids loading rate and feed concentration. There were a number of key concerns relating to startup and operation:

- Prediction of digester behaviour under high loading conditions.
- High potential in-digester nitrogen concentrations (and ammonia inhibition).
- Performance of thermal hydrolysis on a long sludge-age, purely activated sludge system.
- Identification of mechanical, hydraulic, and electrical faults during startup.

The use of modelling at full-scale, by practitioners and consultants has always been a primary goal in publication of the IWA Anaerobic Digestion Model No. 1 (Batstone *et al.* 2002). It has been widely applied in full-scale, largely for identification of mass-balance issues and inconsistencies (Johnson & Shang 2006). One of the most analysed areas had been use in primary and activated sludge digesters, and most papers have focused on model validation (Siegrist *et al.* 2002; Blumensaat & Keller 2005; Parker 2005), or methods to interface and properly characterise sludge inputs to the ADM1 (Huete *et al.* 2006; Yasui *et al.* 2006; Nopens *et al.* 2009).

One of the most difficult periods to analyse effectively–qualitatively, or using a model–is system startup. The impact of initial conditions makes model-based analysis difficult (though convergence is rapid at high loading). Startup of the Oxley Creek anaerobic digesters–a highly loaded system with high ammonia and pH was relatively new to the Brisbane Water commissioning team. This, combined with an intensive analytical schedule and opportunity for duplication over two digesters provided an opportunity for model application to system startup.

METHODS

Digester operation

Two digesters were operated in parallel, with different feed rates and concentrations for 300 days from startup. Both were 2570 m^3, and mesophilic (37°C) with a gas mixing system, and liquid recirculation. Both digesters were operated from an initial loading rate of 25 $m^3\,d^{-1}$, with a daily 5% increase to the maximum goal of 200 $m^3\,d^{-1}$. The increase was steady for the first 40 days, after which upstream mechanical and electrical issues, as well as an increase of organic acids in digester 1 caused variation of feed to both digesters. Digester 2 also received dilution water between day 70 and day 100. This occurred by accident, but flows could be calculated once identified by mass balance on the cooling circuit. This provided different flows and loads to both digesters.

The sludge treatment train consisted of thickening (or blending) to 13% solids, thermal hydrolysis by the Cambi process (Panter & Kleiven 2005). This involves direct steam injection, and pressure/heat in a two stage process to 165°C, and 6 barg. The process liquid is then flashed, and diluted with final effluent to a final solids concentration of 8–9% for feeding to the anaerobic digesters. The digesters were fed on a cyclic basis, with feed cycles typically from 2–4 m^3/digester.

The measurement and analytical regime were very comprehensive for an industrial process. Samples were taken from all four points (raw feed to the hydrolysis process, digester feed, digester 1, digester 2), and analysed for chemical oxygen demand (COD), total and volatile solids (TS/VS), and Total Kjeldahl Nitrogen (distillation). Digester contents were also analysed for pH, bicarbonate alkalinity, ammonia (distillation), and organic acids (GC-FID).

Model implementation

The biochemical model structure used was the ADM1, with parameters as defined by Batstone *et al.* (2002). The model was implemented with a very standard configuration–except for inputs (see below)–in Aquasim 2.1d (Reichert 1994). Initial conditions were based on actual initial conditions of the digester. The inoculum was stored seed supplemented by a single 30 tonne tanker of fresh inoculum. For modelling purposes, the inoculum was simulated by simulation of the pre-existing digesters, and diluted with water.

Model Inputs. The original activated sludge interface proposed by the AD Modelling taskgroup (Batstone *et al.* 2002) is poor, with imbalances in mass, and unrealistic

outputs. A number of approaches have been proposed (see (Batstone & Keller 2006), but we have used the approach of the IWA Benchmarking Taskgroup (Nopens *et al.* 2009). Inputs were split between particulate inerts, carbohydrates, proteins, lipids, organic acids, ammonia, and bicarbonate, in order to balance particulate and soluble COD, carbon, nitrogen, and charge. The key independent parameter was degradability, which defines the particular organic inerts fraction. All other organic material was assumed to be degradable, and split as explained above.

RESULTS

COD: mass ratio for inputs

Implementation of the inputs involved analysis of the COD: mass ratio throughout the system, as shown in Figure 1. This indicates that:

(1) The material is significantly more energy rich (higher COD: Mass) than the IWA models suggest.
(2) Hydrolysis causes an apparent, though not significant increase in COD: Mass ratio, probably due to a chemical release of VFA during hydrolysis. VFA is not measured by the volatile solids analytical method, as it is lost during drying. This value of 3000 mg L^{-1} was as estimated by the difference in COD: VS ratios, and confirmed by direct measurement.
(3) There is an apparent, though not significant rise in COD: mass ratio throughout the system, with the inert organic material left after digestion having the highest specific chemical energy.

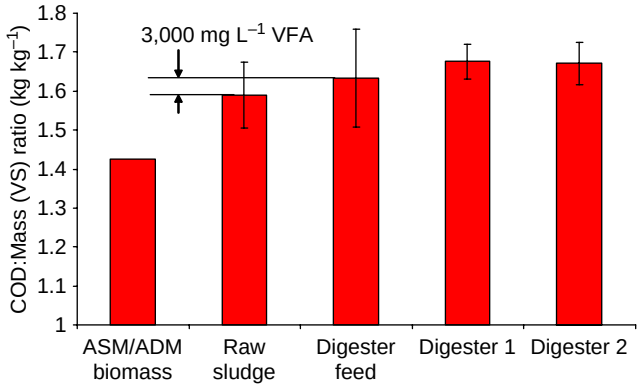

Figure 1 | COD: mass (as measured by VS) ratios throughout the system (error bars indicate 95% confidence in mean values).

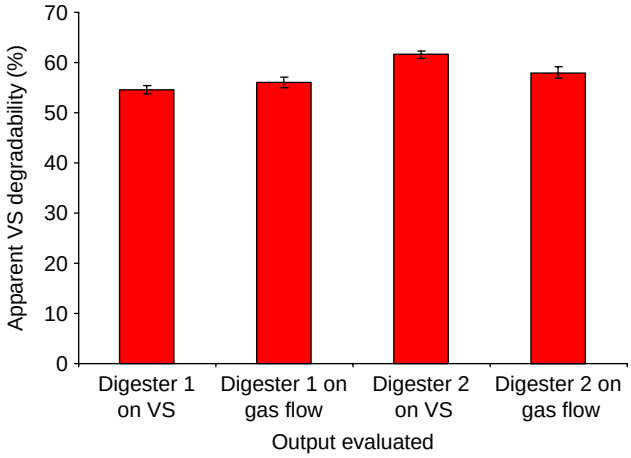

Figure 2 | Apparent degradability as measured by digested sludge VS, and gas flow.

Parameter identification

The key parameter was degradability, or the fraction of organics that goes to particulate carbohydrate, lipid, and protein inputs (X_{ch}, X_{li}, X_{pr}), vs inerts (X_I). It can be equated to potential organics destruction. Degradability can be estimated from either gas flow, or volatile solids measurement. Degradability in the two digesters, as measured by these two independent outputs is shown in Figure 2. This gives an aggregate degradability of 58% ± 1%. A degradability of 58% equates to a VS destruction of 53%, since approximately 10% of the degraded material is used in anabolic growth (Figure 3). This anabolic growth produces additional biomass in the effluent, and hence, reduces

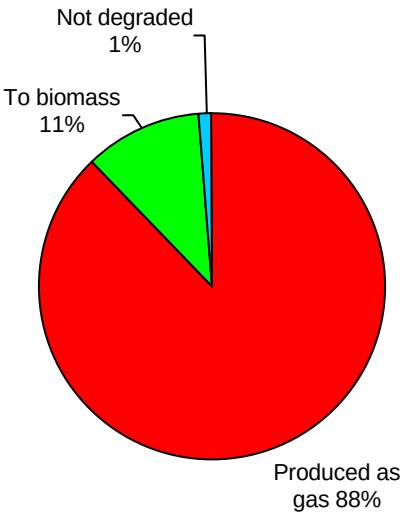

Figure 3 | Sinks for degraded material.

apparent VS destruction. This does not include the organic acid component of 3,000 mg L^{-1}, not measured as VS, which would add an additional 1% to degradability across the whole system. The higher apparent degradability in Digester 2 is due to un-accounted for dilution, which drives apparent VS destruction upwards.

The localised (per month) degradability for these systems has been further discussed in (Batstone *et al.* 2009), and gas flow for optimum degradability is shown in this paper. However, apart from the data shown in this paper, the effluent solids is a very important output as shown in Figure 4. Note particularly the impact of uncontrolled dilution in Digester 2, which was identified based on discrepancies noted during the modelling work. Uncontrolled dilution was caused by leaking from the hydraulic seals in the gas recirculation compressors, particularly in Digester 2, and could not be measured, or included in the model, apart from the period of 80–100 days. It was originally identified around day 90, based on discrepancies between model and actual results, and subsequently fixed. It again occurred later from approx. day 150, again based on model-system discrepancies. Apart from this period, fit is generally good, and varies above and below the data points during specific periods. This is likely due to fluctuations in the degradability of the material as analysed in detail in (Batstone *et al.* 2009).

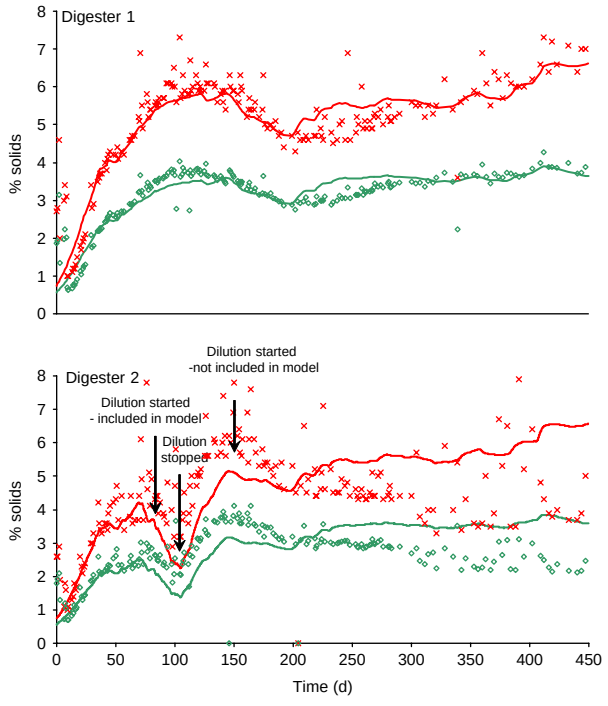

Figure 4 | Measured (points) and model (lines) total solids (TS—'×') and volatile solids (VS—'o') in digester 1 (top) and digester 2 (bottom). Note the impact of dilution in digester 2 (bottom).

The other interesting phenomena was response to free ammonia. The long-term $K_{I,NH3,ac}$, based on fitting acetate in both reactors (450 days) was 0.007 M ± 0.001 M NH_3. This is in agreement with the thermophilic example evaluated by (Siegrist *et al.* 2002), but far less inhibited by the mesophilic example of the same authors (0.0012 M). This may indicate that free ammonia inhibition coefficients are higher than previously believed, and also more consistent between thermophilic and mesophilic systems. However, apart from this long-term chronic inhibition, causing acetate levels of 200–300 mg COD L^{-1} as acetate, we also observed an acute response as the system moved through the threshold of approx. 0.008 M NH_3 (110 mg NH_3–N L^{-1}). This resulted in a very high acute response to 3000 mg COD L^{-1} acetate in digester 1, where flow was decreased in response, or up to 4500 mg COD L^{-1} acetate in digester 2, where flow was not decreased. This indicates that the acute threshold is very similar to the long-term chronic inhibitory parameter.

DISCUSSION

Where the model succeeded

Clarification of measurements. Measuring and analysing a full-scale system at > 5% solids, with imperfect mixing, and high ion activity is relatively difficult, and subject to a high noise. The model was extremely useful in separating the "true" measurement from apparent noise. This can be seen in later TS/VS measurements in digester 2 (Figure 4), but is especially apparent in ammonia measurements (Figure 5).

Prediction of recovery, and key controlling factors. As explained further below, there were two occasions in digester 1, where acetate climbed to very high levels (Figure 6). While the model could not adequately explain the reasons behind the acetate increases, it was effective in predicting recovery times–3–5 days without feed, or 7–10 days with normal feed levels (Figure 6). It could also be used to identify the increased ammonia as a key controlling

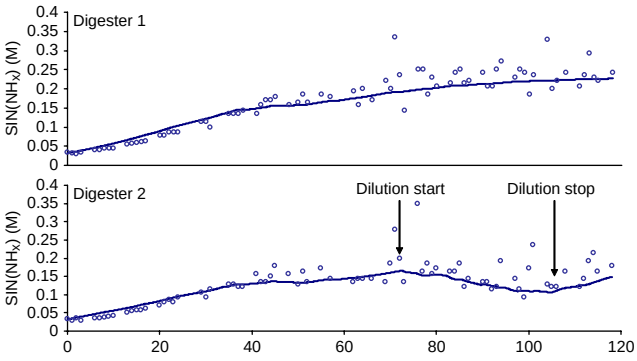

Figure 5 | Soluble inorganic nitrogen ($NH_4^+ + NH_3$) in the two digesters, showing measurement scatter. Shown is measured data (o), and model (-).

factor behind the acetate increases, and slightly increased recovery times. Model predictions were one of the reasons why feed was not stopped during the second increase in acetate. A dilution management strategy was also proposed, using the model, in order to minimise inhibition by ammonia (this was tested incidentally, in digester 2).

Evaluation of a complex, multi-input system (digester 2). It was always intended that both digesters be operated identically to improve confidence in the results. However, increased acetate levels in digester 1, complications in the digester 1 cooling circuit, and dilution water entering digester 2 caused divergence of the operation. While digester 1 could be evaluated by normal mass balance tools, it proved very difficult to evaluate digester 2 this way, due to the additional water stream. The more complex model was very useful in consolidating the wide range of input/output data, and conclusively identifying the dilution water flow to digester 2 as the key reason for divergence in performance data.

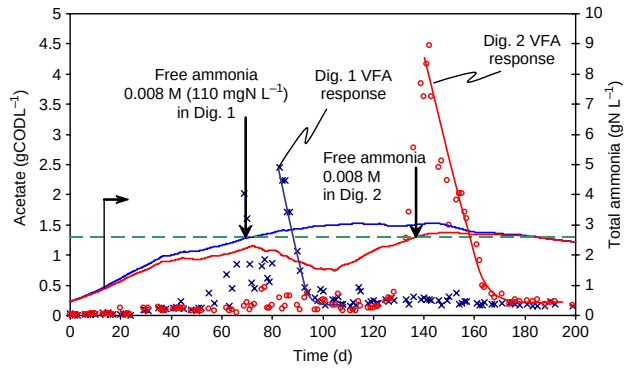

Figure 6 | Acute response from both reactors, as the digester moves through 0.008 M free ammonia (110 mg NH_3–N L^{-1}), or approx. 2600 mg N L^{-1} as total ammonia.

Where the model failed

Preliminary predictions of degradability were high. This was based on the ASM/ADM COD: mass ratio of 1.42 g g^{-1}, and given the true mass of 1.6–1.7 g g^{-1}, caused an over-estimation of degradability initially. This was further complicated by the large amounts of measurement noise, particularly in COD measurements, and the true picture only became visible as large amounts of COD data were collected, thus reducing error bars in Figure 1.

We believe a major factor in the high COD: VS ratio is the high inerts levels in the activated sludge, largely caused by a long activated sludge age (>20 days at > 25°C). Additonally, the startup period was through the Australian summer, with high water temperatures, and sludge ages have been extended during periods of low digester feed capacity. Information on the dynamic sludge age and its further impact on sludge degradability are given in (Batstone *et al.* 2009).

Predictions of acetate increases. The model could not predict the acute response to free ammonia as shown in Figure 6, but was effective in predicting decreases once the inhibitor was alleviated. It was also possible to include long-term chronic inhibition via the normal functions. The main method for including acclimatisation in models is probably replacement of the non-acclimated population with an acclimated population (Ramirez & Steyer 2008), but the acclimatisation observed here appears to be more rapid, with levels dropping rapidly within 10–20 days. Free ammonia levels as observed here are still low as compared to other systems such as manure digesters, and further long-term acclimatisation is likely at higher ammonia levels, for example, by a shift to *Methanosarcinaceae* (Karakashev *et al.* 2005). However, the acute, short term response is very important, as it does not appear to cause long-term impacts, and in the case of Digester 2, we were able to safely ignore it (mainly due to ammonia buffering).

CONCLUSIONS

The model was effective and added value in operational decisions during a relatively aggressive startup, with an atypical feed. The most important limitation was related to

assumed COD: mass ratio of the feed. The default ASM/ADM COD: mass of $1.42\,g\,g^{-1}$ was used–intended for active biomass, but this was unrealistic in our system, where the feed COD: mass was $1.6\,g\,g^{-1}$, and digested sludge inert COD: mass was $1.7\,g\,g^{-1}$. We believe this is related to the long sludge ages, and relatively high temperatures of the upstream activated sludge process. Where the model worked best was prediction of recovery during overload situations, clarification of measured results (noise levels were high), extraction of principal information (e.g., degradability), and analysis of the second digester, which had unintended dilution water being added through a hydraulic seal.

ACKNOWLEDGEMENTS

Brisbane City Council funded this study, and provided analytical results, and engineering resources. BCC financial funding was contributed within the frame of the Neptune FP6 EU project Contract-No. 036845, and the associated Australian Federal Government DIISR International Science Linkage (ISL) project CG110007.

REFERENCES

Barr, K. G., Solley, D. O., Starrenburg, D. J. & Lewis, R. G. 2008 Evaluation, selection and initial performance of a large scale centralised biosolids facility at Oxley Creek water reclamation plant, Brisbane. *Water Sci. Technol.* **57**(10), 1579–1586.

Batstone, D. J. & Keller, J. (eds). 2006 Anaerobic digestion model no. 1–developments and applications–water science and technology 54(4)–2006. IWA Publishing, London.

Batstone, D. J., Keller, J., Angelidaki, I., Kalyuzhnyi, S. V., Pavlostathis, S. G., Rozzi, A., Sanders, W. T. M., Siegrist, H. & Vavilin, V. A. 2002 Anaerobic digestion model no. 1 (ADM1). *IWA Task Group for Mathematical Modelling of Anaerobic Digestion Processes*. IWA Publishing, London.

Batstone, D. J., Tait, S. & Starrenburg, D. 2009 Estimation of hydrolysis parameters in full-scale anerobic digesters. *Biotechnol. Bioeng.* **102**(5), 1513–1520.

Blumensaat, F. & Keller, J. 2005 Modelling of two-stage anaerobic digestion using the IWA anaerobic digestion model no. 1 (ADM1). *Water Res.* **39**(1), 171–183.

Huete, E., de Gracia, M., Ayesa, E. & Garcia-Heras, J. L. 2006 ADM1-based methodology for the characterisation of the influent sludge in anaerobic reactors. *Water Sci. Technol.* **54**(4), 157–166.

Johnson, B. R. & Shang, Y. 2006 Applications and limitations of ADM1 in municipal wastewater solids treatment. *Water Sci. Technol.* **54**(4), 77–83.

Karakashev, D., Batstone, D. J. & Angelidaki, I. 2005 Influence of environmental conditions on methanogenic compositions in anaerobic biogas reactors. *Appl. Environ. Microbiol.* **71**(1), 331–338.

Nopens, I., Batstone, D. J., Copp, J. B., Jeppsson, U., Volcke, E., Alex, J. & Vanrolleghem, P. A. 2009 An ASM/ADM model interface for dynamic plant-wide simulation. *Water Res.* **43**(7), 1913–1923.

Panter, K. & Kleiven, H. 2005 *Ten years experience of full-scale thermal hydrolysis projects*. Wakefield, UK.

Parker, W. J. 2005 Application of the ADM1 model to advanced anaerobic digestion. Bioresource Technology In Press, Corrected Proof.

Ramirez, I. & Steyer, J. P. 2008 Modeling microbial diversity in anaerobic digestion. *Water Sci. Technol.* **57**(2), 265–270.

Reichert, P. 1994 AQUASIM–a tool for simulation and data analysis of aquatic systems. *Water Sci. Technol.* **30**, 21–30.

Siegrist, H., Vogt, D., Garcia-Heras, J. & Gujer, W. 2002 Mathematical model for meso and thermophilic anaerobic sewage sludge digestion. *Environ. Sci. Technol.* **36**, 1113–1123.

Yasui, H., Sugimoto, M., Komatsu, K., Goel, R., Li, Y. Y. & Noike, T. 2006 An approach for substrate mapping between ASM and ADM1 for sludge digestion. *Water Sci. Technol.* **54**(4), 83–92.