Size and structure effects on centrifugal dewatering of digested sewage sludge

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Abstract The application of light scattering over small angles for the determination of digested sludge floc size and structure and its relationship with dewaterability is investigated. It appears that improved dewatering corresponds with lower floc fractal dimension (a more open structure) and a smaller proportion of fine particles. The initial increase in fractal dimension with increasing polymer dose for the digested sludge is most likely due to more efficient aggregation of the finer particles and the resulting formation of denser particle aggregates. A large colloidal fractal of the digested sludge (< 10 μm) appears to be less negative than the bulk digested sludge. This suggests that the fine particles will react differently and possibly less aggressively to the cationic polymer than the larger and more negative particles. The higher negative charge associated with the larger particles might be related to greater levels of highly negatively charged extracellular polysaccharides (EPS) adsorbed to the flocs or could result from the association of FeS with the finer fraction. The appearance of much greater levels of fine particles after digestion suggests that the flocs have undergone disintegration. Whether this is due to reduced biological efficiency within the digestor or iron reduction under the anaerobic conditions is not known for certain, although no indication of prolonged stress in the digesters could be found from plant performance data.

Keywords Floc; centrifuge; wastewater; polymer; light scattering

Introduction

Wastewater treatment processes invariably use coagulants to remove fine particles with the resultant formation of larger particle assemblages or "flocs". The size and structure of these bacterial aggregates are recognised to be important determinants of the efficiency of various treatment processes including sedimentation, thickening and sludge dewatering. Recent years have seen considerable progress in characterizing the structure of bacterial flocs with the recognition that such flocs exhibit mass fractal properties (Mandelbrot, 1983; Li and Ganczarczyk, 1989). That is, the mass (M) of these aggregates may be related to their radius of gyration R_g (the standard deviation of the particles from the centre of mass) by the relationship:

$$M \propto R_g^{D}$$

For linear, planar and three-dimensionally compact objects, the exponent D will have values of 1, 2, and 3, respectively. For porous aggregates (such as wastewater flocs), D may take on fractional values and, in this event, is termed the "fractal dimension" (denoted hereafter as D_f). Compact aggregates have D_t values typically ranging from 2.3–2.5 whereas more open "loose" aggregates have fractal dimensions in the range of 1.7–1.8 (Lin *et al.*, 1990; Amal *et al.*, 1990). In this work, forward light scattering is used to determine both the size and structure of flocs with insight into the floc structure (as described by the assemblage fractal dimension) gained by analysis of scattered light intensity as a function of scattering angle in a manner identical to that described by Guan *et al.* (1998).

Factors responsible for the centrifugal dewaterability of digested sewage sludges are investigated in this paper with particular attention given to the changes in floc characteristics and sludge dewaterability as a function of cationic polymer dose and the presence (or

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absence) of ferric chloride dosing for the purpose of improving effluent quality (i.e. through chemically assisted sedimentation (CAS)). While most attention is focused on digested primary sludge from one source (Malabar), the properties of sludges from various sources in Sydney are included for comparative purposes.

Methods

Bi-weekly digested sludge samples were collected from the Malabar Sewage Treatment Plant (Sydney, Australia) from the 13th January 1999 to the 11th February 1999. An earlier sample was also collected on the 10th December 1998. This sampling period covers the cessation of chemically assisted sedimentation (CAS) on 16th December 1998 and its re-implementation on the 28th January 1999. Total solids (TS) and volatile solids (VS) determination followed AWWA Standard Methods 2450B and 2450E, respectively. While characterisation of unconditioned biosolids was undertaken, the size and structure of samples on addition of cationic polymer (Zetag 92 or Zetag 7650, CIBA Speciality Chemicals) were also determined as these polymers are essential dewatering aids at Malabar. In these cases, a known amount of polymer was added to the biosolids sample and the mixture rapidly stirred (approx. $G = 390/s$) for 0.5 minutes then gently stirred for 1 minute. the sample was then left to reach equilibrium for around 2 to 5 minutes after which size and structure characterisation commenced. Size distributions and structural information of bacterial assemblages were determined using a Malvern Mastersizer/E which ascertains size by analysis of forward scattered light. The Mastersizer consists of a 2 nW He–Ne Laser (632.8 nm wavelength) with 18 mm beam expansion, collimation and spatial filtering for TEM_{00} mode transmission. Further details of the equipment and technique may be found elsewhere (Guan *et al.*, 1998). Streaming current and zeta potential of samples was determined in some instances using a Milton Roy SC4200 streaming current detector (SCD) and a Coulter Delta 440 zeta potential meter. Following characterisation for size and structure, the dewaterability of the samples was assessed in a Spintron GT-1755 laboratory centrifuge. Samples were centrifuged at 3000 rpm for 10 mins, the supernatant removed and the pellet analysed for total solids. As mentioned earlier, while most attention is given to samples from the Malabar Sewage Treatment Plant in this paper, samples were also obtained from a variety of other treatment plants in Sydney and processed in a similar manner to that described above. Selected results of these tests are included here when considered useful for comparative purposes.

Results

The dependence of the fractal dimension (D_f) on polymer dose for biosolids from a variety of different sources is given in Figure 1. A D_f value of 2 is indicative of a relatively open floc whereas higher values describe more compact structures. For virtually all wastewater samples from the many different treatment plants, there is generally a trend where the initially compact sludge increases in openness with increasing polymer dose. More open floc structures (dF 1.8–2.0) have been found to be more amenable for dewatering. Malabar sludge reacts quite differently to most of the other sludges with an initial increase in the fractal parameter on increasing polymer dose. This result indicates that Malabar sludge forms more compact structures on initially dosing with polymer.

As clearly shown in Figure 2, the fractal dimension peaks at a polymer dose of 0.3g/100g dry weight (dw) and then decreases with further polymer dose, indicating the formation of a more open structure at the higher polymer doses. The sludge at a polymer dose of 1.2g/100 dw is clearly overdosed as evidenced by the formation of large floc clusters and the clear separation of liquid and solid phases. The large variation in D_f at this polymer dose is most likely a consequence of the heterogeneity of the sludge sample resulting from the separation of solid and liquid phases.

Laboratory centrifugability studies revealed on average, a 1.4% improvement in dewaterability with the onset of CAS at a polymer dose of 0.8g/100 dw (Figure 3). Lower variation in cake solids content (CSC) is observed for the CAS off than for the CAS on scenario. There is also an observed increase in the dewaterability of the CAS modified sludge even without polymer addition, i.e. with CAS off the CSC ranges from 10–12% and with CAS on the CSC is closer to 14%. Interestingly, only a very minor improvement in dewatering is observed for the dewatering plant for the corresponding period and since July 1998 there has been little difference at all in the dewatering plant TSR results depending on whether CAS is on or off. This discrepancy between laboratory and plant dewatering results may reflect the small sample size for the laboratory experiments but more likely can be attributed to the presence of other process-related factors contributing greater variability than induced by polymer addition.

The reason for the effect of ferric chloride addition on dewaterability is unclear but may be related to the generation of flocs of slightly different structure. In Figure 4, poorer dewatering appears to be associated with ranges of dF values in the higher spectrum while a general increase in CSC is observed where fractal dimensions are limited to lower values. This follows previous observations in other wastewater treatment plants where a more open floc

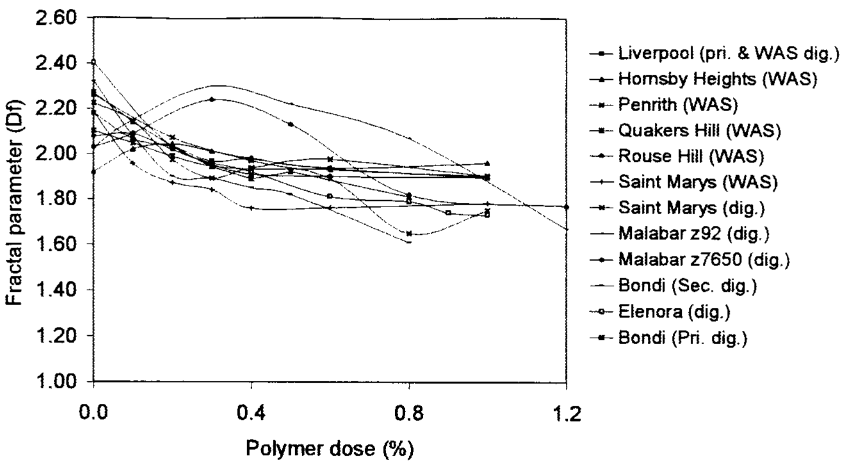


Figure 1 Average fractal dimension (dF) behaviour with respect to polymer dose for different wastewater sludges in Australia (anaerobic and aerobic) (WAS = waste activated sludge; pri. = primary digestion; dig. = digestion; sec. = secondary digestion)

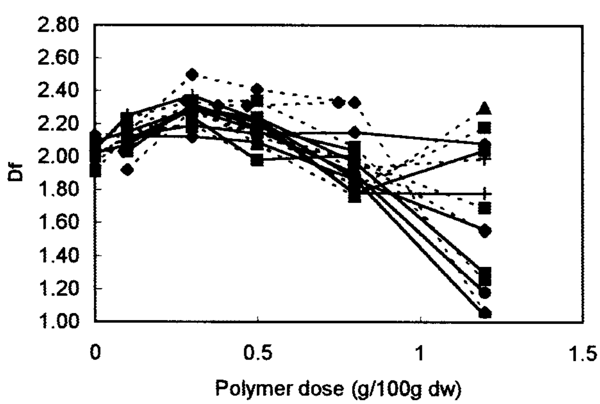
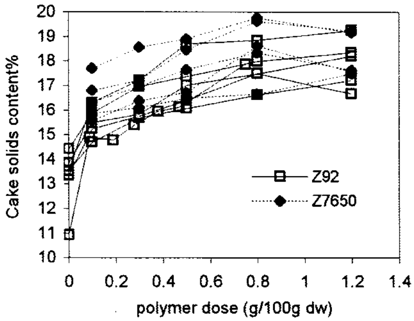


Figure 2 The scattering exponent as a function of polymer dose for Malabar digested sludge (dotted lines refer to Z92 and solids lines to Z7650)



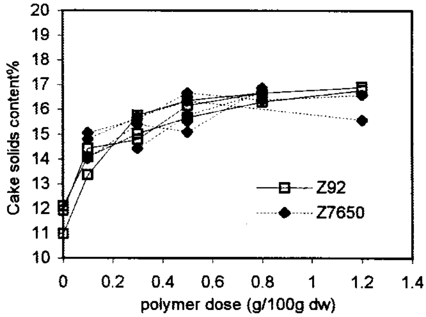


Figure 3 Cake solids content as a function of polymer dose and type with (a) and without (b) chemically assisted sedimentation

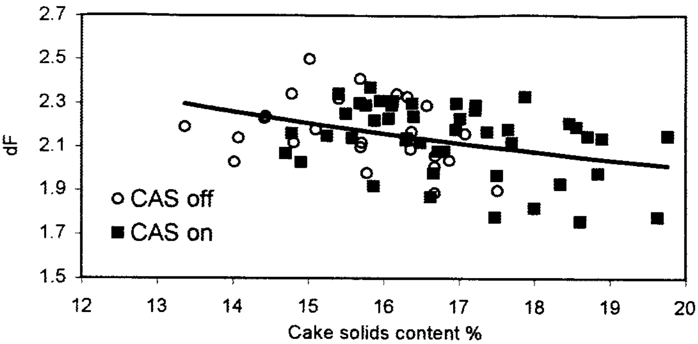


Figure 4 Apparent improvement in dewatering with CAS (Z92 & Z7560, polymer 0.1–0.8 g/100 dw)

structure provides an opportunity for greater dewatering. Whether CAS is on or off makes little difference to the size of flocs at a particular polymer dose as shown in Figure 5. The digested sludge without polymer addition has approximately the same size, independent of whether CAS is on or off (data < 100 µm in Figure 5), yet with CAS on appears to show improved dewatering. The digested sludge has an unusually high negative streaming current ranging from –88 to –107 mV, independent of fractal dimension and CAS. On increasing polymer dose the streaming current becomes less negative until at the highest polymer dose the potential moves into the positive region. This did not appear dependent on CAS.

In accord with the results of other workers (Olböter and Vogelpohl, 1993; Lotito *et al.*, 1993; Lawler *et al.*, 1986), improved dewaterability does appear to correlate with a

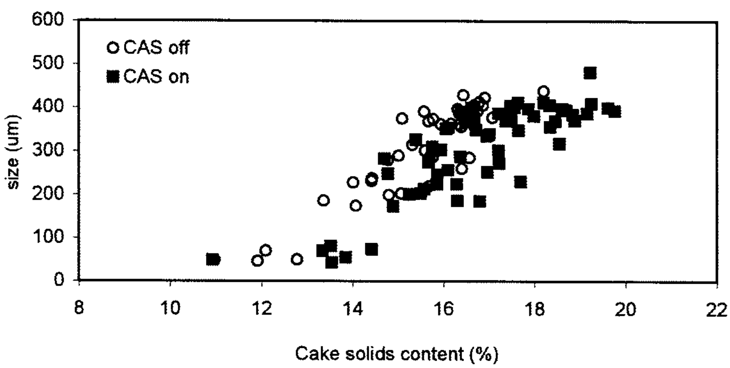


Figure 5 Cake solid content dependence on floc size both Z92 and Z7650

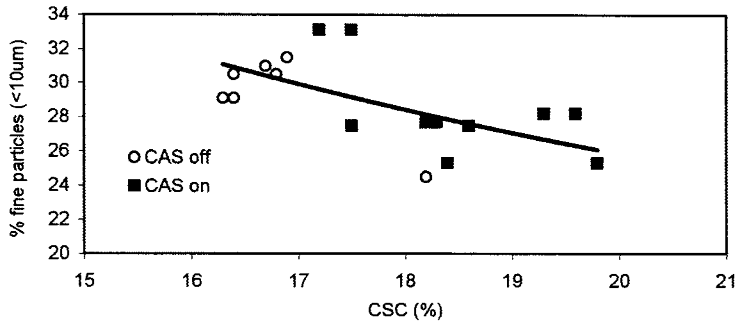


Figure 6 Maximum laboratory cake solids content and its dependence on initial percentage of fines (< 10 μm) in digested sludge for both Zetag 92 and Zetag 7650

reduction in the fraction of fine particles present (Figure 6). This relationship is independent of polymer type and shows only minor dependence on whether CAS is on or off in the case of Malabar digested sludge.

Discussion

It is interesting that while a small improvement in dewatering is observed with CAS on for the laboratory centrifugability studies (Figure 3) there is only a minor corresponding improvement in dewaterability at the plant. No improvement in dewaterability is apparent over a longer time period. Whether CAS is on or off makes little difference to polymer undosed or dosed streaming current measurements but does have a marginal effect on cake solids content dependence on size (Figure 5) and fractal dimension (Figure 4). All these relationships are rather weak but the limited data collected during this short study suggests that CAS has little effect on dewatering at the plant. The most important factor influencing the dewaterability of sludge is particle size (Spellman, 1997). As the average particle size decreases, the surface area of the sludge increases, which leads to the following effects: (1) increased frictional resistance to the movement of water; 2) increased attraction of water to the particle surface due to more adsorption sites; and (3) greater electrical repulsion between sludge particles due to a larger area of negatively charged surface.

Dewaterability is generally linked to the performance of the anaerobic digestor and, as shown in this study, anaerobic digestion changes the particle size distribution of sludges. The change observed at Malabar, however, is one for the poorer. When digestion works well, particles of all sizes are destroyed, but there is preferential removal of particles of small sizes, a consequent loss of specific surfaces area, and therefore an improvement in dewaterability (Lawler *et al.*, 1986). When digestion does not work well, large particles are destroyed resulting in the creation of small particles with a consequent gain in specific surface area, and therefore worsening dewaterability. One problem when trying to assess if a digestor is not performing satisfactorily for dewatering is that there may not be a perceptible change in overall performance indicators (e.g. pH, %VS and %TS reduction). In the study by Lawler *et al.* (1986), digestion did not work well under high influent TS% concentration, at low temperature and when treating 100% activated sludge (compared to primary sludge). None of these are an issue for the Malabar digesters, though, since the influent TS concentration, temperature and sludge source has remained steady. This suggests that the changes in particle size are not due to the digesters being stressed, especially considering digestion appears to be performing satisfactorily (e.g. VS reduction average ~60%).

The simplified schematic diagram given in Table 2 attempts to clarify how the presence of large numbers of these fine particles might be related to the observed values for fractal dimension with increasing polymer dose and decreasing negative charge. As mentioned earlier, the initial streaming current (SC) measurement for the Malabar digested sludge was very negative and this corresponds with similar measurements for Bondi primary digested sludge. Bondi primary digested sludge had a streaming current value of –94 mV and Malabar digested sludge ranged from –88 to –104 mV whereas Bondi secondary digested sludge and St Mary's mixed liquor sludge both had less negative initial SC values of –65 mV. This high negative charge effectively stabilizes the particles in the sludge and prevents aggregation via charge repulsion effects. The initial high negative charge of the particles and the resulting structure response with increasing cationic polymer gives some hint as to what might be happening during polymer dosing. Since the colloidal and near-colloidal particles (i.e. those < 10 μm) occupy on average 29% of the total volume of Malabar particles, a particle number distribution would be heavily biased towards this colloidal range.

As low concentrations of the cationic polymer are added to the sludge, the polymer would be expected to initially bind to the surface of the highly negatively charged particles and form strong interparticle aggregation. The high surface area of the finer particles enables them to bind strongly to the polymer and form tight, packed aggregates. Such densification in structure would lead to an increase in the fractal dimension, as observed in Figures 1 and 2 and shown diagrammatically in Table 1 (0.3% dose). An increase in fractal dimension with increasing polymer dose is also found when aggregating fine coal particles (pers. comm. R. Amal). This change in structure, however, differs from that observed for Bondi primary digested sludge. Indeed, for Bondi sludge there is a reduction in fractal dimension with increasing polymer (as is typical of most sludges) (Figure 1). The difference is understandable if one considers that Bondi's primary digested sludge consists of 6% by volume of < 10 μm particles – compared to Malabar's 29%. The smaller number of fine particles and greater average particle size ensures that the surface area for the Bondi sludge is lower and aggregate packing upon polymer dose is not as efficient. Subsequently, as cationic polymer is added to the similarly negative Bondi sludge, the polymer would be expected to form equally strong interparticle aggregation but the reduced surface area and predominance of larger particles prevents the formation of tightly packed structures. Hence, the fractal dimension would decrease, reflecting a more open structure, as shown in Figure 1 and diagrammatically in Table 1 (0.3% polymer dose). This decrease in fractal dimension continues with increasing polymer dose for the Bondi sludge.

For Malabar digested sludge, once the high negative charge is initially "quenched", many of the negative surface sites are occupied with polymer and the fine particles aggregated. Table 2 shows the disappearance of the < 10 μm particles with increasing polymer dose. At 0.1% dose, there are still many fine particles remaining and it is not until 0.3% dose that almost all the fine particles are removed. This coincides with a peak in the fractal dimension and the highest degree of compaction in structure (Table 2, 0.3% dose). With doses greater than 0.5%, no fine particles are detected and so further dosing of cationic polymer results in bridging of the densely packed aggregates, resulting in a more open structure and corresponding lower fractal dimensions. This decrease in fractal dimension continues with increasing polymer dose (Figure 2) with further aggregation of flocs. Streaming current measurements (SCM) show that zero change is reached at a polymer dose of approximately 0.8%. Above this, the SCM is positive forcing the flocs to destabilize and come out of suspension (there is a clear separation of large floc clusters and clear liquid).

Why is it so difficult to aggregate Malabar sludge with cationic polymer when it has such an apparently high initial negative charge is not clear, but it may be related to the fact that the fine particles are actually not as negatively charged as the bulk sludge. The large number of small particles means much higher concentration of surface sites and thus much

Table 1 Possible evolution in structure for digested sludges upon cationic polymer dosing

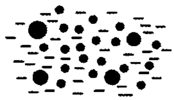
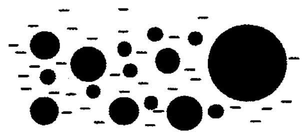
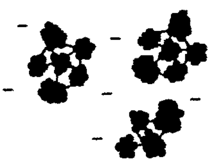
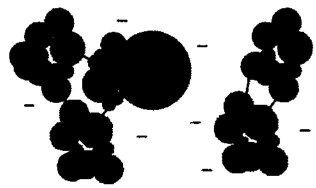
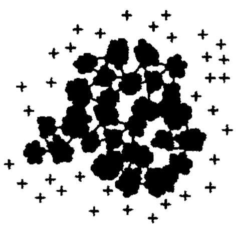
	Malabar digested sludge	**Bondi primary digested sludge**
% cationic polymer dose		
0.3% cationic polymer dose		
0.5% cationic polymer dose		
1.2% cationic polymer dose		

Table 2 Examples of colloidal response to polymer dosing

Polymer dose g/100g dw	**Malabar (15/1/99) % < 10 μm**	**Bondi (29/1/99) % < 10 μm**
0	31.5	6.1
0.1	4.4	0.8
0.3	0.4	0
0.5	0.3	0
0.8	0	0
1.2	0	N/A

more capacity for polymer adsorption (assuming charge interactions are favourable). Isolation of the majority of fine particles (average particle size 25 μm, 50% < 10 μm) from the digested sludge sees reduction in SCM from approximately –100 mV to –50 mV, indicating that the fines are less negatively charged. SCM measures the charge between particles but is not capable of determining the charge for small volumes of fine particles. For these finer particles the zeta potential is used to determine charge. This is still a measurement of surface charge but is different to SCM as it measures charge closer to the actual surface of a particle. Further size separation of the particles leads to a fine suspension (average particle size 9.5 μm, 75% < 10 μm) that has a mean zeta potential of –5.1 mV suggesting that the fine particles are less negatively charged than the bulk digested sludge. It could be possible to exploit this difference in charge by using difference polymer dosing schemes, i.e. a less positive polymer to aggregate the less negative fine particles used in conjunction with highly cationic polymer to flocculate larger aggregates.

These fine particles primarily consist of organic material (VS = 63 w/w%), similar to that of the bulk digested sludge (VS ~ 0 w/w%) but lower than the raw sludge (VS ~ 80 w/w%). This suggests that the fraction of fines mostly consist of small bacterial aggregates. Further studies are required to determine the composition of the inorganic material but it is anticipated that a significant fraction might consist of FeS particles. Ferric chloride (added to assist sedimentation), ferrous sulfate (added as “Odorlock” to reduce sulfide levels) and the raw wastewater are all sources of iron in the anaerobic digester. Calculations indicate that the iron contribution solely from CAS leads to an average Fe concentration in the digester of approximately 900 mg/L. This ignores iron already present in the raw wastewater and contributions from Odorlock. Under the anaerobic, reducing conditions of the digester, Fe-III will be reduced to Fe-II and the potentially high levels of Fe and S in the digester could lead to the formation of FeS. One problem associated with fine FeS particles is that flocs incorporating these particles perform poorly under shear. Studies have shown that the reduction of Fe-III to Fe-II and the formation of FeS in activated sludge leads to floc disintegration and release of bacteria, organic colloids and dissolved extracellular polymeric substance (EPS) (Nielsen and Keiding, 1998). Higher levels of EPS have been found to increase cationic polymer use due to their highly negative surface charge (Kopp and Dicht, 1998). Further indication that Fe might be an important influence in the digester is that the raw sludge consists of substantially fewer fines than the digested sludge (i.e. ~10% compared to 29) and has a much larger average particle size (~130 μm compared to ~50 μm for digested sludge). This result is consistent with floc disintegration. Whether this is due to the digester being stressed from a biological viewpoint or iron reduction within the reactor is unknown, though plant operational data suggests that it might not be the former. Further work is required to determine the predominant inorganic particles present and the level of iron in the digested sludge in order to fully assess the importance of iron in the digester and its relationship to sludge dewaterability.

Conclusions

1. Improved dewatering corresponds with lower floc fractal dimension (a more open structure) and a smaller proportion of fine particles. The initial increase in fractal dimension with increasing polymer dose for the Malabar sludge is most likely due to more efficient aggregation of the finer particles and the resulting formation of denser particle aggregates.
2. The large colloidal fraction of the digested sludge (<10 μm) seems to be less negative than the bulk digested sludge. This suggests that the fine particles will react differently and possibly less aggressively to the cationic polymer than the larger and more negative particles. The higher negative charge associated with the larger particles might be related to greater levels of highly negatively charged EPS adsorbed to the flocs.
3. The appearance of much greater levels of fine particles after digestion suggests that the flocs have undergone disintegration, one way or another. Whether this is due to reduced biological efficiency within the digester or iron reduction under the anaerobic conditions is not known for certain, although no indication of prolonged stress in the digesters could be found from plant performance data.

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References

Amal, R., Raper, J.A. and Waite, T.D. (1990). *J. Colloid Interface Sci.* **140**, 158–168.
Amal, R. (1999). Personal communication. School of Chemical and Industrial Engineering, University of New South Wales, Sydney, Australia.
Guan, J., Waite, T.D. and Amal, R. (1998). Rapid structure characterization of bacterial aggregates. *Environ. Sci. Tech.* **32**, 3735–3742.
Kopp, J. and Dicht, N. (1998). Influence of surface charge and exopolysaccharides on the conditioning characteristics of sewerage sludge. In: *Chemical Water and Wastewater Treatment V.* H.H. Hahn, E. Hoffmann and Ødegaard (eds), Springer-Verlag, Berlin, pp. 265–296.
Lawler, D.F., Chung, Y.J., Hwang, S.-J. and Hull, B.A. (1986). Anaerobic digestion: Effects on particles size and dewaterability. *J. Wat. Poll. Contr. Fedn.* **58**(12), 1107–1117.
Li, D.-H. and Ganczarcrzyk, J.J. (1989). *Environ. Sci. Tech.* **23**, 1385–1389.
Lin, M.Y., Klein, R., Lindsey, H.M., Weitz, D.A., Ball, R.C. and Meakin, P. (1990). *J. Colloid Interface Sci.* **137**, 263–280.
Lotito, V., Mininni, G., Spinosa, L. and Lore, F. (1993). Developments in laboratory evaluation of sewage sludges dewaterability. *Wat. Sci. Tech.* **28**(1), 103–108.
Meldelbrot, B.B. (1983). *The Fractal Geometry of Nature*, W.H. Freeman and Co., New York.
Neilsen, P.H. and Keiding, K. (1998). Disintegration of activated sludge flocs in presence of sulfide. *Wat. Res.* **32**, 313–320.
Olböter, L. and Vogelphol, A. (1993). Influence of particle size distribution on the dewatering of organic sludges. *Wat. Sci. Tech.* **28**(1), 149–157.
Spellman, F. (1997). *Dewatering Biosolids.* Technomic Publishing Company, Inc., Lancaster PA, USA.