Journal of Water Supply: Research and Technology—AQUA | 53.1 | 2004

Fundamental dewatering characteristics of potable water treatment sludges

Peter J. Harbour, Nevil J. Anderson, Ainul A. A. Aziz, David R. Dixon, Peter Hillis, Peter J. Scales, Anthony D. Stickland and Martin Tillotson

ABSTRACT

Capillary suction time (CST) and specific resistance to filtration (SRF) measurements are commonly used in the characterisation of sludges and for the prediction of efficient operation of dewatering processes such as centrifugation and filtration. Whilst occasionally useful in predicting trends, they do not assist in the design and optimisation of dewatering devices from first principles. Recent work in our laboratories has led to the development of a technique for the fast measurement of the permeability and compressibility of sludge. The use of a single volume-fraction dependent parameter, namely the solids diffusivity, $D(\varphi)$, calculated from permeability and compressibility, enables full characterisation of the dewaterability of sludge. This allows different sludges to be compared in an unequivocal fashion. Data is presented for a range of sludges from different sources showing vastly different dewatering properties. The dewaterability of the different sludges is easily compared and the true role of flocculants in dewatering is highlighted.

Key words | dewaterability, potable water treatment sludges, pressure filtration

Peter J. Harbour (corresponding author)
Nevil J. Anderson
CSIRO Molecular Science,
Private Bag 10,
Clayton South MDC,
Victoria,
Australia 3169
E-mail: *peter.harbour@csiro.au*

Ainul A. A. Aziz
David R. Dixon
Peter J. Scales
Anthony D. Stickland
Particulate Fluids Processing Centre,
Department of Chemical and Biomolecular Engineering,
University of Melbourne,
Victoria,
Australia 3010

Peter Hillis
United Utilities,
Lingley Mere, Lingley Green Avenue,
Warrington WA5 3LP,
UK

Martin Tillotson
Yorkshire Water,
Halifax Road,
Bradford BD6 2LZ,
UK

INTRODUCTION

Until now, characterisation of water treatment plant (WTP) sludges has commonly been performed using techniques such as capillary suction time (CST) and specific resistance to filtration (SRF). It is largely accepted that CST is dependent on a number of non-fundamental parameters (Vesilind 1988; Dentel 1997; Novak *et al.* 1999), which render it unsuitable for the purposes of comparing different sludge types or sludges from different plants (Vesilind 1988). This technique is also highly dependent on initial solids concentration. Efforts to avoid these problems, by taking measurements at different sludge solids or by calculating 'filterability', result in characterisations based on empirical or semi-empirical factors which cannot be universally applied. SRF may give useful information regarding the permeability of non-compressible sludges but with compressible materials such as water treatment sludges the measurements are highly dependent on the applied pressure and the initial solids fraction. To fully characterise such materials requires testing at a range of pressures up to the required operating pressure covering a range of initial volume fractions. The standard method for SRF determination uses a vacuum to establish the pressure gradient and the accuracy of the measurement is limited by difficulties in maintaining a constant pressure differential throughout the entire filtration. Neither of these techniques gives information regarding the extent of filtration or the maximum solids achievable (Novak *et al.* 1999).

Whilst it is noted that, for determining the effects of flocculants on a specific sludge, both these methods may be qualitatively and practically useful, it should also be recognised that both measurements are lacking with respect to fundamental material characterisation and as parameters for use in the modelling and prediction of filtration processes on a fundamental basis.

The filtration characteristics of water treatment sludges can vary for a number of reasons including changes in the quality (turbidity or colour) of the raw water being treated or more commonly changes in the type and dose of primary coagulant being used. The standard primary coagulants used in water treatment are alum (aluminium sulphate), ferric salts (sulphate or chloride), poly (aluminium chloride) (PACl) and cationic polyelectrolytes. In the case of metal salts, the filtration characteristics of the sludge may also be dependent on the formation conditions of the metal hydroxide precipitate. Temperature, pH adjustments and mixing conditions may all affect the nature of the solids produced. In addition, each plant may operate with different coagulation mechanisms that can vary from charge neutralisation to sweep floc or enhanced coagulation. The varying nature of water treatment sludge and the lack of robust fundamental characterisation techniques highlight the requirement for an improved measure of filterability and dewaterability.

The solid–liquid separation theory, introduced by Buscall & White (1987) and developed for pressure filtration by Landman & White (1997) and Landman *et al.* (1999), uses the fundamental parameters of permeability (rate of filtration) and compressibility (extent of filtration) to characterise materials with regard to dewaterability. These two parameters show strong non-linear dependencies on solids volume fraction, φ, and are fundamental properties of the sludge. In this instance, the volume fraction of solids is defined as the volume of solids divided by the total volume. For full details of the derivation of the filtration theory, refer to Landman & White (1997). In summary, the rate of filtration is quantified by the hindered settling function, $R(\varphi)$, which is inversely proportional to the permeability of the cake and related to specific cake resistance. The compressibility, $P_y(\varphi)$, is a measure of the pressure required to achieve a specific equilibrium solids volume fraction or, alternatively, it is the equilibrium solids achieved at a given applied pressure. These two fundamental parameters are combined when solving the theoretical governing equations to give the diffusivity, $D(\varphi)$.

$$D(\phi)=\left[\frac{\delta P_y(\phi)/\delta(\phi)}{R(\phi)}\right](1-\phi)^2 \quad (1)$$

$D(\varphi)$ is a material characteristic which is independent of both initial volume fraction and the type of filtration process. This relationship describes the interplay between the compressibility, the solids volume fraction and the permeability of the cake. It should be noted that as a measure of dewaterability, SRF describes a particular set of circumstances and is a finite subset of $D(\varphi)$.

The use of this approach has been practically demonstrated using stepped pressure filtration techniques (de Kretser *et al.* 2001*a*, *b*; Usher *et al.* 2001) to fully characterise mineral slurries. It has also been shown that this parameter can be used, in conjunction with a suitable physical model of the filtration device, to model filtration processes and predict filtration performance in terms of filtration time and final solids. A model describing the performance of plate-and-frame filter presses has been developed and validated at full-scale (Eberl *et al.* 1995).

In addition, for constant pressure filtration, the diffusivity can be related to the true time of dewatering, t_f, by the following equation (Landman & White 1997):

$$\mathrm{t_f}=h_0^2\left(\frac{\phi_0}{\phi_\infty}\right)^2\frac{T_f}{D_\infty} \quad (2)$$

where h_0 is the initial sludge height or cake thickness, φ_0 and φ_∞ are the initial and the final volume fraction respectively, Tf is the scaled (dimensionless) time to filtration and D_∞ is $D(\varphi)$ at φ_∞. T_f is dependent on φ_0, φ_∞ and $D(\varphi)$ and calculated based upon the solution of Landman & White (1997). However, the changes in T_f are relatively small compared with the changes in D_∞. From Equation (2) it is clear that D_∞ is inversely related to the true filtration time and is therefore, a parameter which can be used to describe dewaterability as a function of solids volume fraction. Simple comparisons of dewaterability can be made by comparing $D(\varphi)$ on an equal volume fraction basis, i.e. by plotting $D(\varphi)$ versus φ. In previous papers, Aziz *et al.* (2000) and Harbour *et al.* (2002) used this parameter to describe and compare the dewatering properties of mineral slurries, selected inorganic sludges and WTP sludges.

The aim of this paper is to compare the dewaterability of a number of WTP sludges formed using a range of different coagulants. This will be done using the stepped pressure filtration technique to characterise the sludges based on diffusivity.

EXPERIMENTAL

A range of sludges was obtained from a number of WTP plants in both the UK and Australia. In addition, some sludges were prepared using a WTP pilot plant. The raw water quality and clarification conditions for each sludge are shown in Table 1. Polymeric flocculants (LT-22, -24 and -35) supplied by Ciba Specialty Chemicals were used in some circumstances.

The compressibility and permeability of each sludge sample were measured on a pressure filtration rig using a stepped-pressure technique as described in detail elsewhere (de Kretser *et al.* 2001; Usher *et al.* 2001). The permeability and compressibility were measured in two separate filtration runs. In each run, a constant pressure was applied until a certain 'stepping criteria' was achieved. At this point the pressure was stepped (increased) to a new pressure and the stepping criteria was again applied. This was performed for a number of pressures (usually five or six) in the range 20–300 kPa, which covers a large number of filtration processes. The main difference between the two measurements was the pressure stepping criteria. For the permeability measurement the stepping criterion was based on the linearity of a plot of time, t, versus V^2, where V is the specific volume of filtrate. The gradient of this graph at each pressure was then used to calculate $R(\varphi)$. The pressure stepping criterion for the compressibility measurement was based on lack of movement of the piston over a time span proportional to the filtration measurement. This criterion has previously been validated against laboratory centrifuge experiments (Green *et al.* 1998). The compressibility is defined by the equilibrium volume fraction (φ) achieved at each applied pressure, P_y. For each material, a higher applied pressure resulted in a higher equilibrium volume fraction. The final solids fraction was measured at the end of the compressibility run by oven drying the pressed cake at 105°C overnight. The volume fractions at all pressures were then back calculated using the liquid and solids densities and volume of the cake at each equilibrium point. $R(\varphi)$ and $P_y(\varphi)$, as measured at each pressure, were then combined using Equation (1) to give $D(\varphi)$.

RESULTS AND DISCUSSION

Sludges from a variety of WTPs in South Eastern Australia and the UK have been collected and analysed. Sludges from a number of controlled pilot plant trials performed on water from Auravale Lake, near Melbourne, Australia, are also presented. The quality of the raw waters and the treatment conditions applied are shown in Table 1. Australian plants are indicated by 'A', UK plants by 'UK'; each different number corresponds to a different plant. If sludges were produced at the same plant under different conditions these are indicated by an additional lower case letter.

In most cases the sludges were not conditioned with polymer flocculant prior to dewatering; however, in the case of the UK sludges extra polymer had been added to condition them prior to dewatering. For the A2 sample, conditioning was performed in the laboratory and results for both conditioned and unconditioned are shown.

The exact chemical composition and particulate structure of each sludge was not known. However, given that colour and turbidity are gross measures of the dissolved organic content and particulate loading of the raw water, respectively, and that each treatment plant was optimised to remove colour, it is reasonable to infer the composition of these sludges from the colour, turbidity and treatment conditions listed in Table 1.

The data are presented as graphs of diffusivity, $D(\varphi)$, versus solids volume fraction, φ, which, as outlined above, give a direct indication of the time to filtration. Sludges with a higher $D(\varphi)$, at a given volume fraction, will filter faster and exhibit shorter filtration times to attain the same

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Table 1 | Raw water quality and treatment conditions for WTP sludges

	Raw water quality		**Clarification conditions**	
	Colour (PCU)	**Turbidity (NTU)**	**Coagulant (dose)* ($mg\ l^{-1}$)**	**Flocculant (dose) ($mg\ l^{-1}$)**
A1a†	60	5	Alum (3.5)	N/A
A1b†	60	5	Alum (5.1)	N/A
A1b† (B/W)	60	5	Alum (5.1)	N/A
A1c†	60	5	Alum (8.0)	N/A
A1d†	60	5	PACl (3.1)	N/A
A1e†	60	5	PACl (4.1)	N/A
A1f†	60	5	$FeCl_3$ (18.6)	N/A
A1g†	60	5	$FeCl_3$ (24.1)	N/A
A2a	47	193	Alum (8.5) + LT 35 (4.2)	LT 22 (0.2)
A2b	26	83	LT35 (1.8)	LT 22 (0.2)
A3	40	3	Alum (4.0)‡	LT24 (0.2)‡
A4	105	6	$FeCl_3$ (2.4)	(not known)
A5	30–40	3–27	PACl (1.3)	N/A
A6	30‡	70‡	Polymer (2.0)‡	Polymer (0.2)‡
UK1	60	4	Alum (2.5)	Polymer (0.2)
UK2	20	15	Alum (5.0)	Polymer (0.25)
UK3	188	3	$Fe_2(SO_4)_3$ (15.4)	Polymer (0.45)
UK4	20	3	Fe/Al (not known)	(not known)
UK5	70	15	Fe (6.0)	N/A
UK6	80	5	Fe (5.8)	Polymer (0.1)

*Doses of alum, PACl and ferric salts are presented as the metal.
†Pilot plant trials.
‡Values have been estimated.
(B/W), filter backwash.
'Polymer' is indicated where the exact polymer type is not known.

solids, when compared with materials with lower $D(\varphi)$. As stated earlier, this assumes that all other conditions are equal. The diffusivity is plotted on a log scale so each division on the y-axis is a factor of ten in terms of filtration time.

Under certain conditions it is also possible to get an indication of the compressibility of the sludges from these graphs. For a strict comparison of compressibility as defined above, it is necessary to compare the equilibrium volume fractions achieved for each material at a given pressure. In that case, a higher volume fraction indicates a higher compressibility. In these graphs the diffusivity, measured at a number of pressures, is plotted against the equilibrium volume fraction. As each sludge was measured over similar pressure regimes (applied pressures between 20 and 300 kPa), the volume fractions achieved for each dataset, as indicated on the x-axis, give an indication of the compressibility of the sludges. In the context of our work, sludges that show higher diffusivities will more readily form a cake at the equilibrium solids volume fraction.

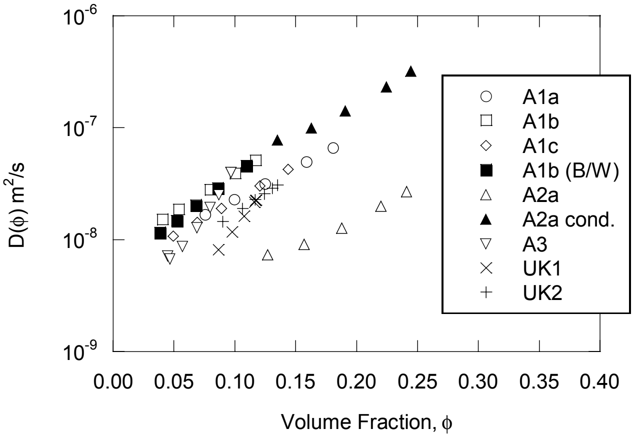

Figure 1 | Diffusivity versus solids volume fraction for a range of alum sludges.

Alum sludges

Alum is a common coagulant and one of the main chemicals used in water treatment, especially for water supplies from areas of low colour. Alum has quite a complex chemistry with respect to its formation of hydroxides, and many different species, including polymeric aluminium species, are known to form under various conditions. Factors such as dose of alum, pH, rate of addition of hydroxide, OH/Al ratio and time (i.e. ageing) are all known to affect the type of products formed. This complexity means alum is also very versatile and conditions can be manipulated to attain different coagulation mechanisms, including charge neutralisation, precipitation, sweep floc and enhanced coagulation.

Auravale Lake water was treated using a pilot plant at various doses of alum and the characteristics of the sludges produced (A1) are presented in Figure 1. These treatment conditions are likely to span the range of coagulation conditions outlined above although each coagulation mechanism may not be mutually exclusive. Sample A1b was formed at a dose of 5.1 mg Al l^{-1}, which is approximately the optimal dose that would be used in conventional water treatment. Figure 1 shows that the pilot plant data are closely grouped. Interestingly the sludge formed at the optimal treatment dose (A1b) is less compressible (reached a lower volume fraction of solids) than the other samples (A1c and A1a), formed at higher and lower doses, respectively. The sludge formed at the lowest alum dose (A1a) shows the highest compressibility. However, sludge A1b has a higher diffusivity at a given solids volume fraction and will filter faster to a specific volume fraction. The filter backwash water (A1b (B/W)) has similar properties to the settled sludge (A1b). The data for the full-scale WTP alum sludge (A3) also lie in this area indicating that the treatment conditions used on the pilot plant were similar to those applied on the full-scale plant.

Figure 1 also shows data from a full-scale plant (A2a) where the primary coagulant was a combination of alum and polyelectrolyte. The A2a sample shows much higher compressibility than the conventional alum plant sludge but lower diffusivity indicating a lower permeability. This may be due to the formation of more compact flocs when polymer is used in conjunction with alum. The high turbidity of the raw water indicates a high proportion of particulate solids, which may also act to increase the compressibility. Adding extra polymer to condition the A2a sludge in the laboratory (A2a cond.) increased the permeability and hence the diffusivity of the sludge,

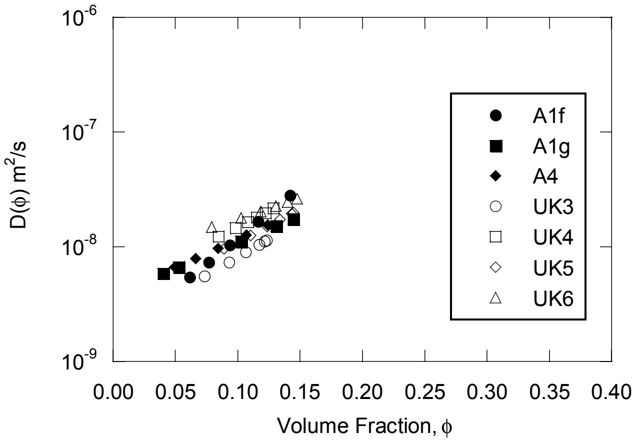

Figure 2 | Diffusivity versus solids volume fraction for various ferric sludges.

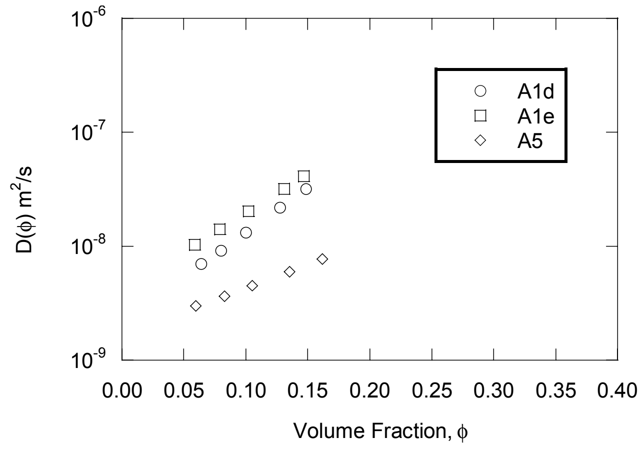

Figure 3 | Diffusivity versus solids volume fraction for various poly(aluminium chloride) sludges.

thereby improving the rate of filtration. The UK sludges appear to have similar diffusivities to each other and the pilot plant sludges, A1a and A1c; however they are less compressible than the latter.

Ferric sludges

Ferric coagulants are often used to treat waters of low turbidity and high colour. The high colour requires high coagulant doses and the sludges produced generally have low particle to coagulant ratios. The diffusivity of some ferric sludges is shown in Figure 2. The data are very closely grouped indicating that these sludges have similar properties even though they are formed under different conditions in terms of raw water quality and clarification protocols. In general, these sludges have low compressibility, <15% solids by vol. at 300 kPa, compared with alum sludges which attain solids volume fractions of up to 25%. (These fractions are higher in mass terms due to the density of the solids; 15% v/v equates to 25–30% w/w.)

The pilot plant runs using lower ferric dose resulted in sludge (A1f) with lower diffusivity at low volume fraction but an increased diffusivity at higher volume fraction. This implies that the improvement in filtration rate obtained by applying higher pressures will be slightly greater for sludges formed at low doses. This type of behaviour also points to the presence of an optimum dose that is dependent on the dewatering device and the ultimate treatment for which the sludge is destined. The compressibility of A1f was slightly better than A1g at low volume fraction (corresponding to low pressure) but the data do converge to the same compressibility at high volume fractions (corresponding to high pressures).

Overall, differences in the behaviour of these sludges are small and they effectively have similar filtration characteristics. The small variations in behaviour mean good operational stability for a dewatering plant, with respect to processing times and final solids, assuming constant solids feed concentration. Differences in plant performance are often observed for ferric sludges; however these results suggest that they are more likely caused by changes in operational parameters such as initial solids concentration or solids loading. The main drawback with ferric sludges is the low compressibility and hence low final solids achievable.

Poly(aluminium chloride) sludges

PACl is used in water treatment to enable better control of the aluminium species present. It is often claimed to be a more efficient coagulant and when used in place of alum allegedly produces less sludge. The filtration results for the pilot plant trials on Auravale (A1d and e) and for full-scale plant data (A5) are presented in Figure 3. The filtration

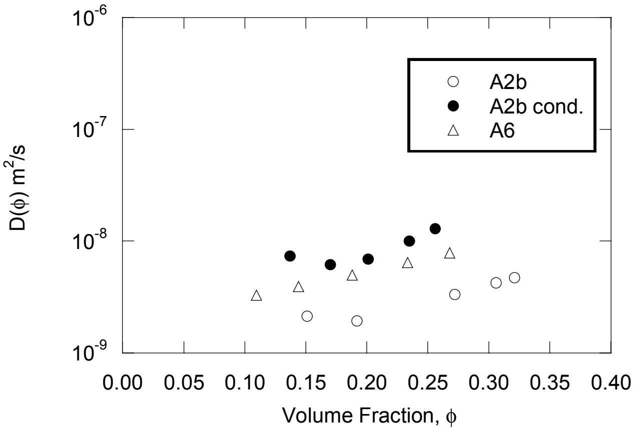

Figure 4 | Diffusivity versus solids volume fraction for various polyelectrolyte coagulated sludges.

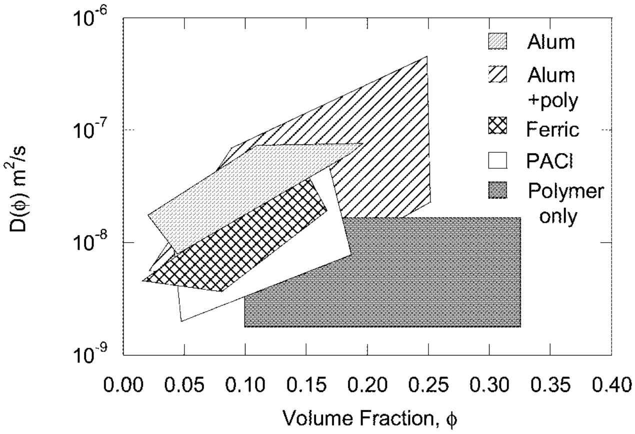

Figure 5 | An overview of the diffusivity for various water treatment sludges.

behaviour of the two pilot plant sludges is similar but the higher dose PACl sludge had slightly faster filtration properties. The A5 sludge was much slower to filter but has similar compressibility to the pilot plant sludges. The compressibility of the PACl sludges is also similar to the ferric sludges, achieving approximately 15% volume fraction at 300 kPa. When compared with the other pilot plant sludges produced using alum, the diffusivity of the PACl sludges was similar to the overdosed and underdosed alum sludges. At the optimum dose PACl sludges are more compressible but less permeable than alum sludges.

Polymer only sludges

Some plants choose to use cationic polyelectrolytes as the primary coagulant for drinking water treatment. Figure 4 shows data for two sludges, A2b and A6. In the case of A2b, a portion of the sample also had extra polymer added to examine the effect of polymer conditioning. In this case, it resulted in a less compressible but more permeable (higher diffusivity) sludge. The A6 sample had similar diffusivity to the conditioned A2b sludge. All these sludges had greater compressibility compared with the sludges previously examined and therefore attained higher solids under the same applied pressures. The diffusivity for a given volume fraction is much worse than for the other sludges so whilst better solids were achieved this material took much longer to filter/dewater. This is possibly due to the replacement of the open gelatinous nature of the metal hydroxide flocs with less permeable, more compact polymer flocs. However, it may also be due to the higher turbidity of the source waters in both cases.

SUMMARY

Characterisation results have been presented for a number of water treatment plant sludges from both the UK and Australia treating a variety of raw waters. These sludges also represent a number of different clarification conditions using a range of primary coagulants including alum, alum and polyelectrolyte in combination, poly (aluminium chloride) (PACl), ferric chloride and cationic polymer only. Figure 5 shows an overview of the sludge diffusivity separated into the major sludge types. The properties of the ferric sludges are very closely grouped and therefore provide the highest operational stability in terms of solids output and processing time. The lack of variability in ferric sludges compared with the alum sludges is possibly a result of the low turbidity of the raw waters so that the main constituent of the sludge is the ferric hydroxide itself. The alum sludges have similar compressibility to ferric or PACl sludges but have a higher diffusivity and therefore filter faster. The PACl results were similar to the ferric although a greater spread in diffusivity was observed. Compressibility was improved when alum was replaced, either partially or completely, with a polyelectrolyte as the primary coagulant. It is likely that this is due to the lower amount of less compressible metal hydroxide flocs but it may also be a function of the higher turbidity of the source waters. The sludges containing polymer as primary coagulant were more compressible and offer higher solids but have lower diffusivity and therefore take longer to filter.

Polymer conditioning of sludge appears to increase the permeability of the sludge, allowing faster filtration, but does not improve the compressibility of the sludge (i.e. final solids achievable at a given applied pressure), which compares well with results obtained in other studies (Johnson *et al.* 2000). For fixed time dewatering operations, such as belt press filters, the increase in filtration rate will often result in the observation of increased solids; however, this is because these filtration processes rarely approach equilibrium.

ACKNOWLEDGEMENTS

The authors acknowledge the financial support of the Particulate Fluids Processing Centre (funded as a Special Research Centre of the Australian Research Council) and industrial sponsorship from United Utilities, Yorkshire Water and Brisbane City Council. Financial support for A. D. Stickland is supplied by an Australian Post-graduate Award from the Australian Research Council, and for A.A.A. Aziz by the Malaysian Government 'Excellent Student Scheme'.

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First received 19 December 2002; accepted in revised form 30 May 2003