Journal of Environmental Science and Technology

ISSN 1994-7887

Journal of Environmental Science and Technology 3 (1): 35-46, 2010
ISSN 1994-7887

Assessment of Using Chemical Coagulants and Effective Microorganisms in Sludge Dewaterability Process Improvement

M.S. Shihab
Department of Civil Engineering, University of Mosul,
College of Engineering, Mosul, Iraq

Abstract: The feasibility of using a combination of Effective Microorganism (EM) and conventional chemical conditioner was evaluated in this study to assess and discern the dewatering properties of the secondary sludge which was produced from wastewater treatment plant of the medical assembly in Mosul City. Conventional coagulants such as lime, alum and ferrous sulfate, of six doses for each coagulant type i.e., 5-30, 10-60 and 25-150 mg L^{-1}, respectively, were used in the sludge conditioning processes for the enhancement of the sludge dewatering capacity. The characteristics of the conditioned sludge at each dose, such as Specific Resistance to Filtration (SRF), were determined. Experimental results indicated that EM seeds had a passive effect on SRF value which was about 71.4 and 75% in lime and ferrous sulfate respectively. While in the alum conditioning process, a significant effect on SRF reduction was accomplished which was about -47.9 and -32.8% for EM and alum dose increments, respectively. The optimum alum dosage, which gave minimum SRF 0.98348×10^{12} m kg^{-1}, was 60 mg L^{-1} at 1% EM. However, a suitable linear relationship has been found to describe the variation of EM and coagulants dosage as a function of SRF.

Key words: Sludge dewatering, sludge chemical conditioning, specific resistance to filtration, effective microorganism

INTRODUCTION

There are various methods of sludge dewatering and these include lagoons, sludge drying beds, filter pressing, vacuum filtration and filter belt presses. The design and operation of any of these systems are based on the volume of the wet sludge as well as dry solids weight. Mechanical dewatering has found extensive industrial applications in solid-liquid separation mainly due to its low energy consumption in comparison with classical thermal methods (Berktay, 1998; Eckenfelder, 2000).

Usually the addition of coagulants to the waste activated sludge, sludge conditioning processes, before the mechanical pressure is applied to force out the water are essential that may increase the dry solid by 20-30% using iron salts and lime (Eckenfelder, 2000).

Chemical stabilization of sludge is aimed not to reducing the quantity of biodegradable organic matter, but to creating conditions that inhibit microorganisms in order to retard the degradation of organic materials and prevent odors. The most common chemical stabilization procedure is to raise the pH of sludge using lime or other alkaline materials, such as cement kiln dust (Weber, 1972).

The methods selected for pre-treatment; dewatering and disposal must depend not only on the availability and practicability of certain processes but also on the economics of the situations. Dewatering and disposal costs for a medium-sized activated sludge plant represents as much as 50% of the initial capital and 65% of the operating costs (Calcutt and Moss, 1984).

Berktay (1998) studied the effect of pressure on settleability and filterability of the sludge produced from a pressurized biological wastewater treatment unit in comparison with an equivalent unit held at atmospheric pressure but otherwise operating under identical conditions. The specific resistance to filtration was found to be moderately stable with a variation between 31×10^{13} and 87×10^{13} m kg^{-1} (mean 51.9×10^{13} m kg^{-1}) at 49 kN m^{-2}. The sludge tested from the reference unit with the same vacuum gave values from 51×10^{13} m kg^{-1} to as high as 217×10^{13} m kg^{-1} (mean 123×10^{13} m kg^{-1}). The indications from the tests carried out were that the sludge from the pressurized unit would be rather more readily dewatered than that from the reference unit.

The filtration-compression of suspensions has received an extensive theoretical and experimental investigation. Lee and Wang (2000) have proposed a review of the various theories dealing with filtration and compression and their implications to sludge dewatering. However, in the case of activated sludge, experimental data are still generally lacking due to the variability and the complexity of this kind of material (Ghiaus *et al.*, 2000).

Chia-Hung and Kung-Cheh (2003) presented Capillary Suction Time (CST) and Specific Resistance to Filtration (SRF) measurements for the organic activated sludge conditioned with fly ash and polymer. The SRF slightly fell from 1.33×10^{12} to 1.00×10^{12} m kg^{-1} and the CST slightly decreased from 20.9 to 20.0 sec. Conversely, the sludge viscosity increased from 2.81 to 3.01 cp when the fly ash was increased from 0 to 12% showing that physical conditioning with fly ash does not significantly improve the dewaterability of organic activated sludge. While the measurements of the SRF and CST for Polymer have been markedly affected the sludge dewaterability. The SRF declined from 1.33×10^{12} to 0.13×10^{12} m kg^{-1} and the CST decreased from 20.9 to 15.0 sec at polymer dosages from 0 to 25 mg L^{-1}, suggesting that the optimal dosage is around 15 mg L^{-1}.

Buyukkamaci and Kucukselek (2006) investigated the improvement of dewatering properties of oily sludge produced from a petrochemical industry. Conventional chemical conditioners such as alum, lime and polyelectrolyte and less conventional ones like fly ash, gypsum and bentonite were used in the sludge conditioning studies. Alum was one of the best conditioners in this study and the optimum alum dosage for both SRF and CST was 8%. SRF decreased from 0.5×10^{16} to 2×10^{15} m kg^{-1} (E = 92%) while CST decreased from 2000 to 600 sec (E = 70%) in this application. Approximately, 38.5% CST (CST min = 1232 sec) and 80% SRF (SRF min = 1.1×10^{16} m kg^{-1}) reductions were achieved using 8 and 12% lime, respectively. However, cationic polyelectrolyte applications gave better results than other conditioners studied. With polyelectrolyte conditioning, it was possible to obtain up to 95% CST and SRF reductions. CST decreased to 22.4 sec (E = 99%) and SRF was reduced to 0.28×10^{16} m kg^{-1} (E = 95%) by the addition of cationic polyelectrolyte.

Disposal bio-solids from west Ahwaz wastewater Treatment Plant (WWTP) were usually used for agricultural activities in the lime addition reactor, pH amounts and FC removal efficiency increases with increasing the lime dosage from 0.2 to 0.38 g $Ca(OH)_2$ g^{-1} of dry solids of bio-solids. However, the results indicate that there is a little benefit in the operating of reactor beyond the ratio of 0.265. Therefore, this amount can be considered as an optimum ratio of lime addition for Ahwaz WWTP bio-solids stabilization (Farzadkia *et al.*, 2007).

The main objectives of this study are to compare the performance of Effective Microorganism (EM) with lime, alum and ferrous sulfate in terms of Specific Resistance to Filtration (SRF), assess the effectiveness of EM seed and estimate its optimum dose for the conditioning of sludge produced from the wastewater treatment plant of the medical assembly in Mosul City.

MATERIALS AND METHODS

Chemical Conditioners and Sewage Sludge

Chemical conditioning is the more widespread method used in sludge dewatering. Alum, lime and ferrous salts are commonly used chemical conditioners (Eckenfelder, 2000). The combination of activated sludge with Effective Microorganism (EM) was accomplished in this study. EM comprises beneficial bacteria belonging to three main generations: phototrophic bacteria, lactic acid bacteria and yeast. The effective microorganisms substances such as vitamins, organic acids, chelated minerals and antioxidants are beneficial when in contact with organic matter (Zimmermann and Harrison, 2004). This beneficial consortium can be used in a wide variety of applications, but for the purposes of this research, sludge dewaterability will be the main topic addressed.

Raw sewage sludge used in this work was secondary activated sludge collected from the recycling line of the wastewater treatment plant in the medical assembly of Mosul city. The period of conducted study was ranged from the mid of April till end of June 2009. The bio-sludge was pretreated with EM solution with the proposed doses illustrated in Table 1 for 24 h with continuous stirring as thickening process, these processes were conducted in Environmental Engineering Laboratory of Civil Engineering Department in Mosul University. The following Table in the experiment design was conducted in the experimental work, where (SRF) was obtained for each of the references dose. However, the Jar test was used to evaluate (SRF); six doses from each of alum, lime and ferrous sulfate were used, as depicted in Table 1. The variables of jar test were fixed as follows (A 1.0 L cylindrical glass beakers):

- Rapid mix 100 rpm for 1.5 min
- Slow mix 40 rpm for 20 min
- Settling time 30 min

Specific Resistance to Filtration (SRF)

In sludge dewatering practice the descriptive parameter is the specific resistance of Sludge (SRF) test, which is known as the Buchner funnel test. It is one of the most commonly employed tests for the evaluation of wastewater sludge dewaterability or filterability. High specific resistance indicates poor dewateraility.

Table 1: Procedure of experimental work

	EM (%)			
Conditioner	0	0.5	1	2
Lime $Ca(OH)_3$ doses (mg L^{-1})	5	5	5	5
	10	10	10	10
	15	15	15	15
	20	20	20	20
	25	25	25	25
	30	30	30	30
Alum $Al_2(SO_4)$ doses (mg L^{-1})	10	10	10	10
	20	20	20	20
	30	30	30	30
	40	40	40	40
	50	50	50	50
	60	60	60	60
Copper as Ferrous Sulfate $FeSO_4.7H_2O$ doses (mg L^{-1})	25	25	25	25
	50	50	50	50
	75	75	75	75
	100	100	100	100
	125	125	125	125
	150	150	150	150

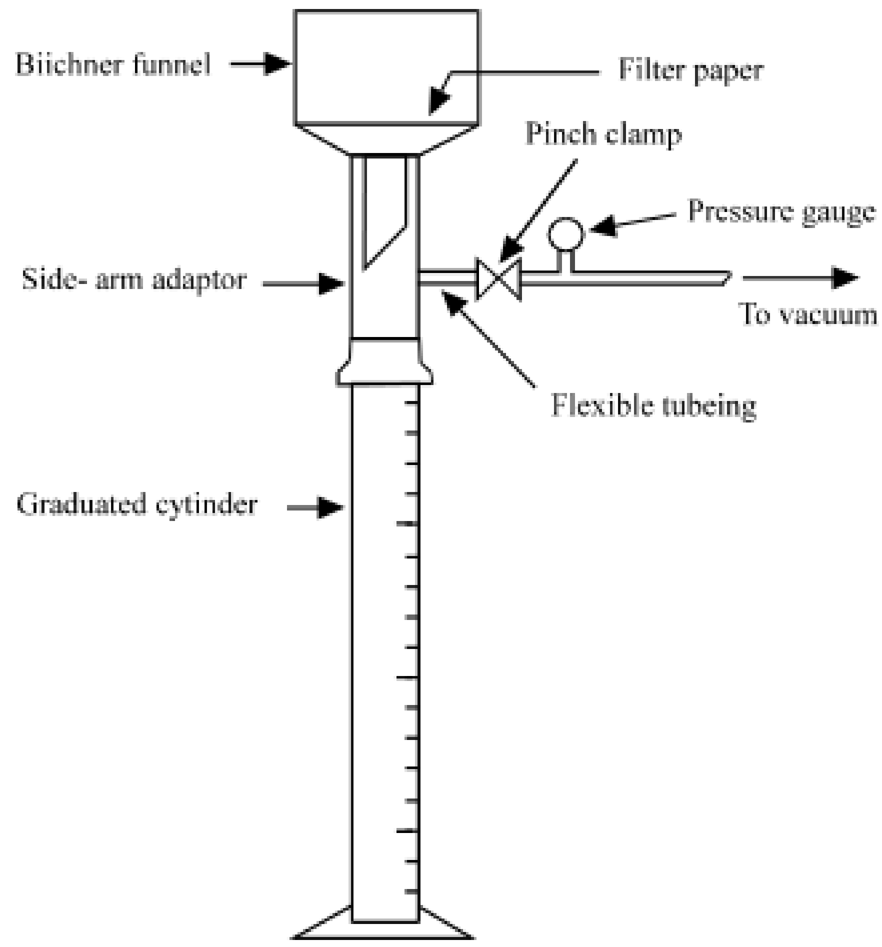

Fig. 1: Sludge specific resistance laboratory apparatuses

The sludge specific resistance test was performed using the laboratory apparatuses shown in Fig. 1 and in the following order (Eckenfelder, 2000; Viessman and Hammer, 1985):

- A wet filter paper (Whatman No. 1 qualitative, 5.0 cm) was placed in Buchner funnel
- The conditioned sludge was poured into Buchner funnel
- Vacuum pressure of (50 kN m^{-2}) was applied
- The initial volume of filtrate was recorded after 2 min
- The volume of filtrate was recorded as a function of time (5 min intervals)
- When vacuum seal breaks, temperature and pH of filtrate were measured

Specific resistance to filtration (SRF or r) was determined using a plot of filtration time/filtrate volume (t/V) versus filtrate volume (V). Using the slope of the line, SRF was calculated from the following Ruth's equation which generally well describes the filtration sub-process (Attar *et al.*, 2005; Viessman and Hammer, 1985).

$$\frac{t}{v} = \left(\frac{\mu r w}{2A^2 P}\right) v + \frac{\mu R_f}{AP} \qquad (1)$$

Where:
r = The specific resistance to filtration (m kg^{-1})
P = The vacuum pressure of filtration (N m^{-2})
μ = The viscosity of filtrate (normally taken as that of water at filtrate temperature) (Ns m^{-2})
V = The volume of filtrate (m^3)
t = The filtration time (sec)
w = The weight of dry solids per volume of filtrate (kg m^{-3})
A = The area of the filter paper (m^2)
R_f = The resistance of filter medium (m^{-1})

This basic formula assumes laminar flow, uniform solids deposition during filtration and constant increase in filtrate flow resistance as the cake increases in thickness.

For compressible sludge, ignoring (R_f) which is very small as compared to the resistance on the sludge cake, Eq. 1 was reduced to:

$$\frac{t}{v} = \left(\frac{\mu r w}{2A^2 P}\right) v = bv \tag{2}$$

Taking the slope of the line as (b) and equating the coefficient of (V), (r) was calculated from the formula:

$$r = \left(\frac{2A^2 P}{\mu w}\right) b \tag{3}$$

RESULTS

Lime

Floc properties can be improved with changes in the cation concentrations. To improve sludge flocculation addition of divalent cations, Ca^{+2} as $Ca\ (OH)_2$ formula, which is traditionally known as lime, was managed. Calcium carbonate and other calcium compounds formed by reaction with constituents of sludge no doubt assist in providing a granular structure for dewatering. Results obtained in Fig. 2 shows the effect of EM and lime increments on SRF at sequential dosages. The statistical analysis showed that EM seeds addition in sludge thickening processes had an intrinsic passive effect on SRF value which was about (71.4%), whereas, minimum SRF value was 1.265×10^{12} m kg^{-1} at 0% EM and 20 mg L^{-1} lime dose. Conversely, SRF value was 7.65×10^{12} m kg^{-1} at 2% EM and the same lime dose (up to 83.5% increments in SRF value). While lime dosages increment had a slight effect (about 0.9%) on SRF curves, excluding 1% EM, that no plausible explanation can be given for this different behavior. A rational relationship for prediction SRF value was derived as follows:

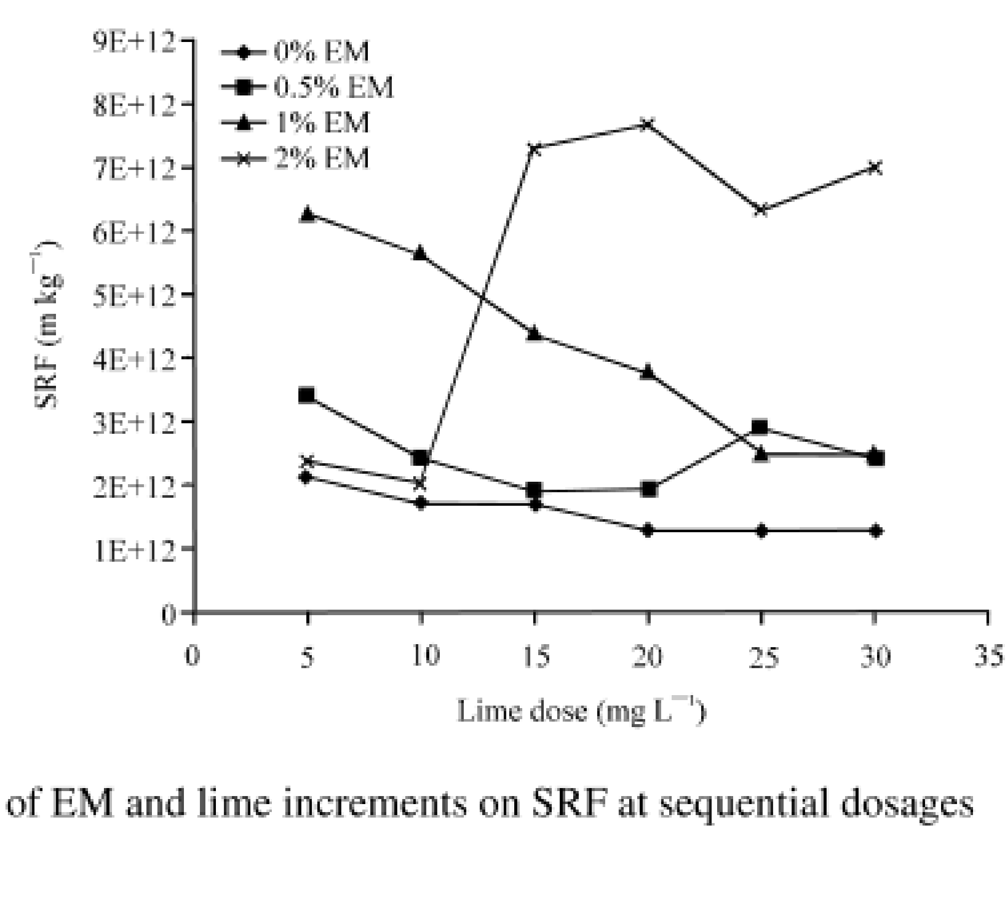

Fig. 2: Effect of EM and lime increments on SRF at sequential dosages

$$SRF = 1.7\times10^{12}+2.0\times10^{12}\times A-202\times10^{9}\times B_1 \quad (4)$$

Where:
SRF = The specific resistance to filtration (m kg^{-1})
A = EM seeds volumetric percent
B1 = Lime dose in mg L^{-1}

Lime addition to sludge raised pH value from (7.45-8.15) at 5 and 30 mg L^{-1} of lime dose, respectively, where the initial pH of the raw sludge was about (6.5±0.3), as shown in Fig. 3. Noticeably, there is a slight effect on filtrate pH dropping at 2% EM which could lend to the deduction that increasing EM concentration might reduce lime used dosages.

Corresponding to Fig. 4, in general, increasing EM concentration produces a relatively high bio-solids concentration in filtrate, with singular variation at 2% EM.

Alum

Alum was one of the best conditioners and the optimum alum dosage, for minimum SRF 0.98348×10^{12} m kg^{-1}, was 60 mg L^{-1} at 1% EM, while at 40 mg L^{-1} and 2% EM, SRF was 1.1087×10^{12} m kg^{-1}. Figure 5 shows markedly the results of sludge conditioning studies when alum was used. The specific resistance of the sludge declined with the increases in biomass load of the sludge, whereas solids content was ranged from 48 at 0% EM to 62% at 2% EM.

The linear regression of the experimental data indicates that the significant effect of EM and alum dose were about 47.9 and 32.8%, respectively. A linear relationship also has been found to be suitable to describe the variation of the EM and Alum dose as a function of SRF:

$$SRF = 3.9\times10^{12}-7.1\times10^{12}\times A-2.1\times10^{10}\times B_2 \quad (5)$$

Where:
B2 = Alum dose in mg L^{-1}

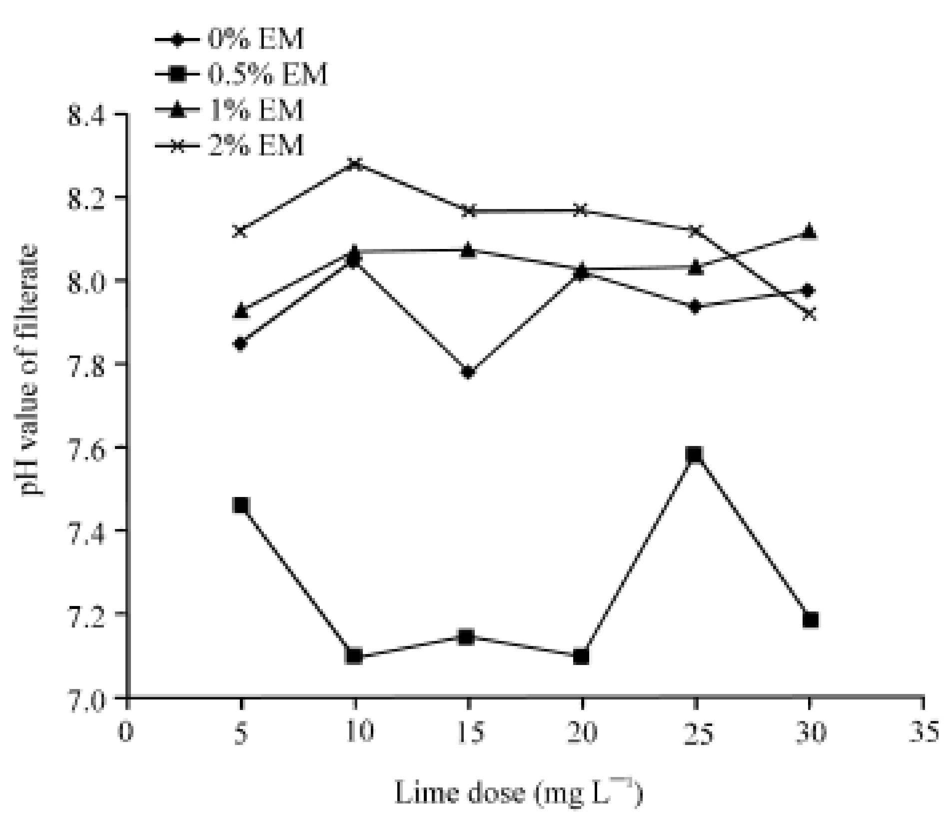

Fig. 3: pH value of filtrate corresponding to EM and lime dosage

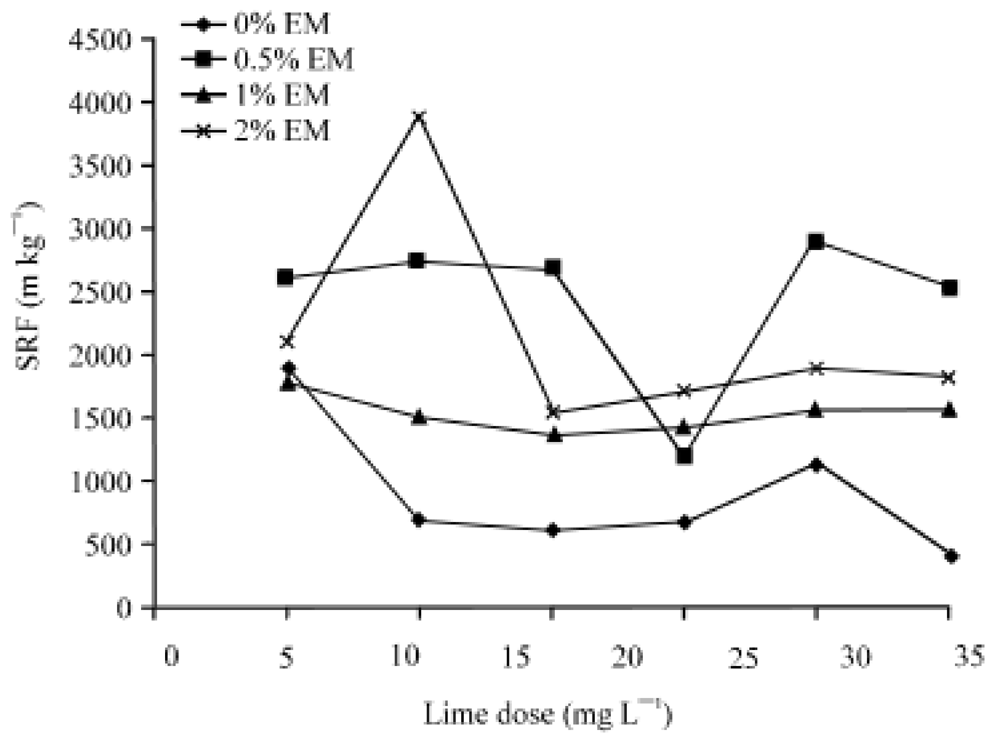

Fig. 4: Variations of total solids concentration in filtrate at sequential dosages of EM and lime

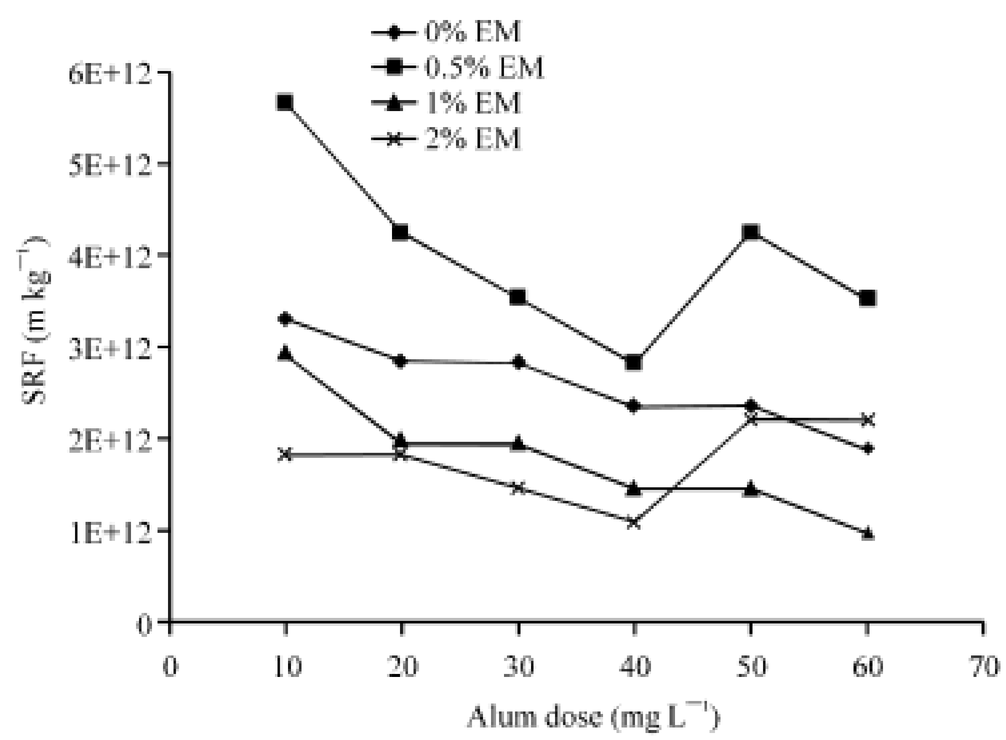

Fig. 5: Effect of EM and Alum increments on SRF at sequential dosages

The operating region for Alum precipitation is for a pH range 5.0 to 7.0 with minimum solubility occurring at pH 6.0 . pH as a factor that influences coagulation is important because the addition of metallic cation (in this case Al^{+3}) automatically lowers pH. Conversely, pictorial results have been depicted in Fig. 6.

Figure 7 shows the variations of total solids in filtrate as a function of EM and Alum dosages, slight variations were noticed with EM and Alum increment dosages, excluding 2% EM dose where a dramatic diminishing in solids was obtained.

Ferrous Sulfate

The basis of activated sludge floc formation lies in the abilities of micro-organisms to stick to each other and to non-biological particles. The mechanisms of flocculation are supposed to happen by bacterial aggregation and adhesion mechanism, charge neutralization and/or bridging mechanisms (Katja and Mika, 2007).

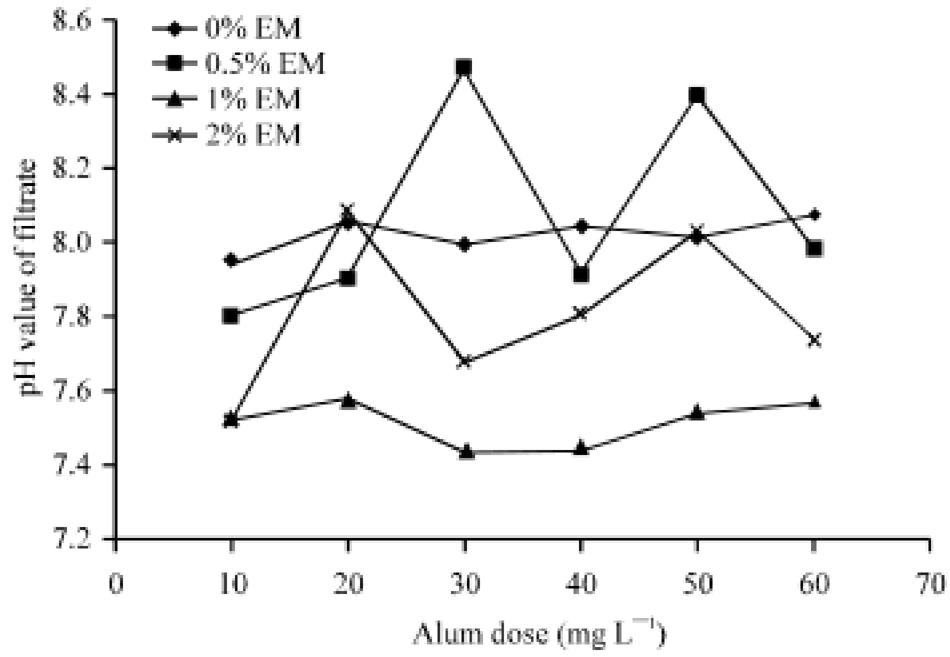

Fig. 6: pH value of filtrate corresponding to EM and Alum dosage

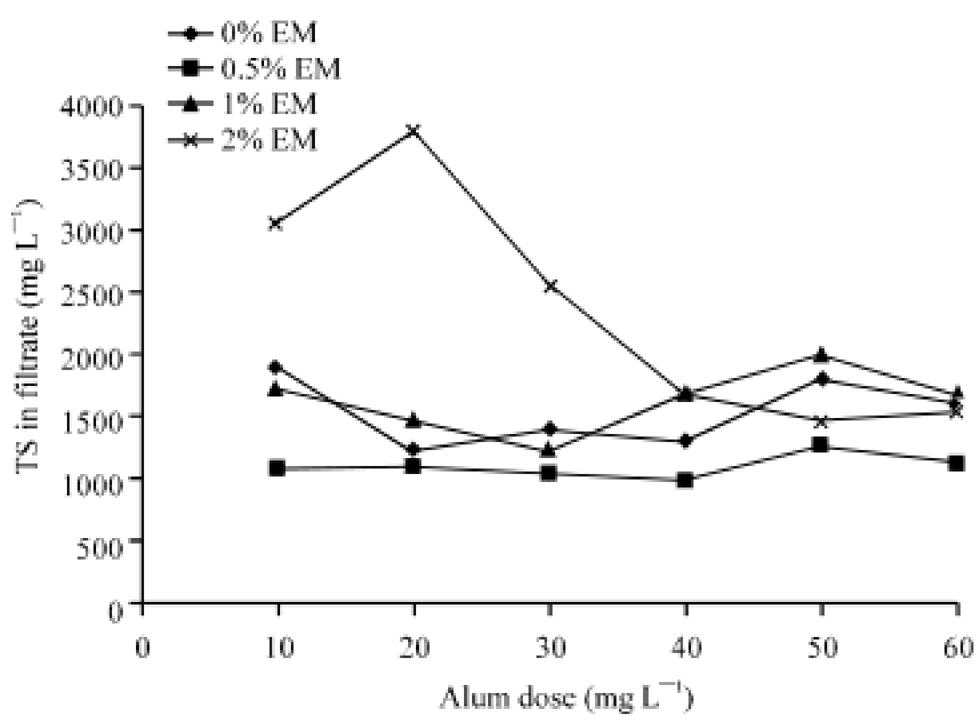

Fig. 7: Variations of total solids concentration in filtrate at sequential dosages of EM and Alum

Remarkably, the behavior of ferrous sulfate in dewatering process was analogous to lime. The solids content increased from (50 to 62.3%) at 0 and 2% EM respectively. The maximum SRF value was 10.85×10^{12} m kg^{-1} and the minimum value was 1.37×10^{12} m kg^{-1} at 2 and 0% of EM (about 99.8% increase in SRF value), for the same coagulant dose which was 100 mg L^{-1}, as illustrated in Fig. 8. Once again as in Lime case, the EM seeds have an imposing passive effect on SRF value, as obtained from the statistical analysis, which was about 75% but ferrous sulfate has less effect (about 8.2%) for ferrous sulfate dosages.

The obtained descriptive relationship was as below:

$$SRF = 8.9\times10^{11}+2.9\times10^{12}\times A+5.6\times10^{9}\times B_3 \quad (6)$$

Where:
B3 = Ferrous sulfate dose in mg L^{-1}

Ferrous sulfate coagulants reduce pH and when they are added to sludge, a pH change occurs along with the formation of the coagulant products, as demonstrated in Fig. 9.

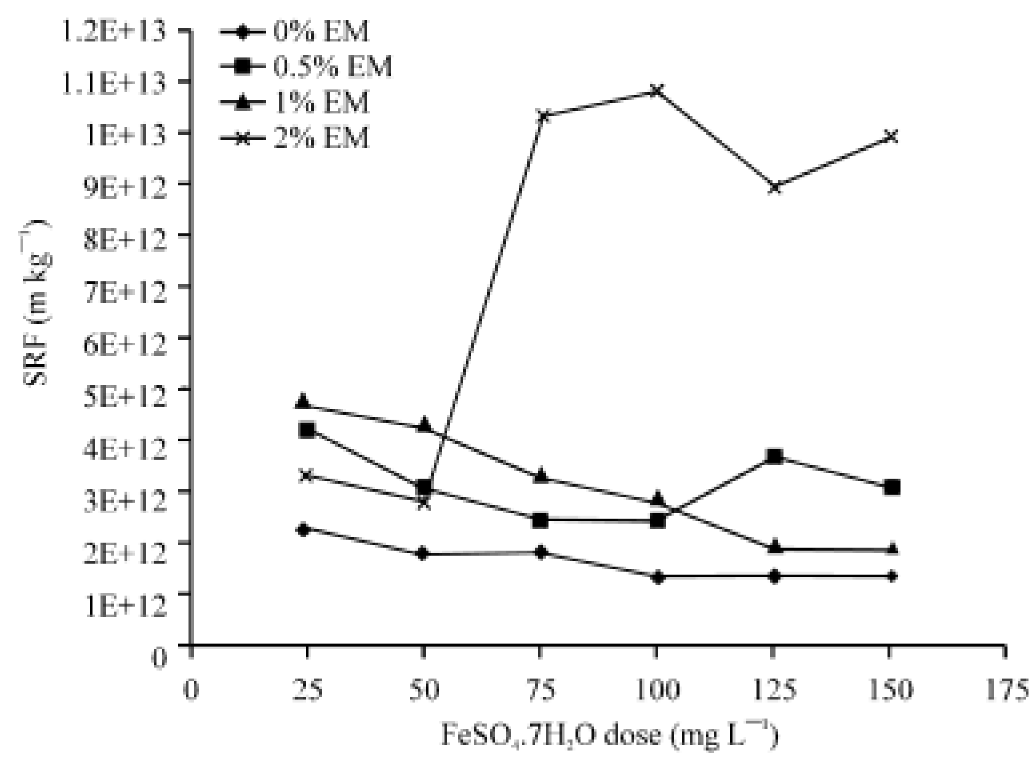

Fig. 8: Effect of EM and Ferrous Sulfate increments on SRF at sequential dosages

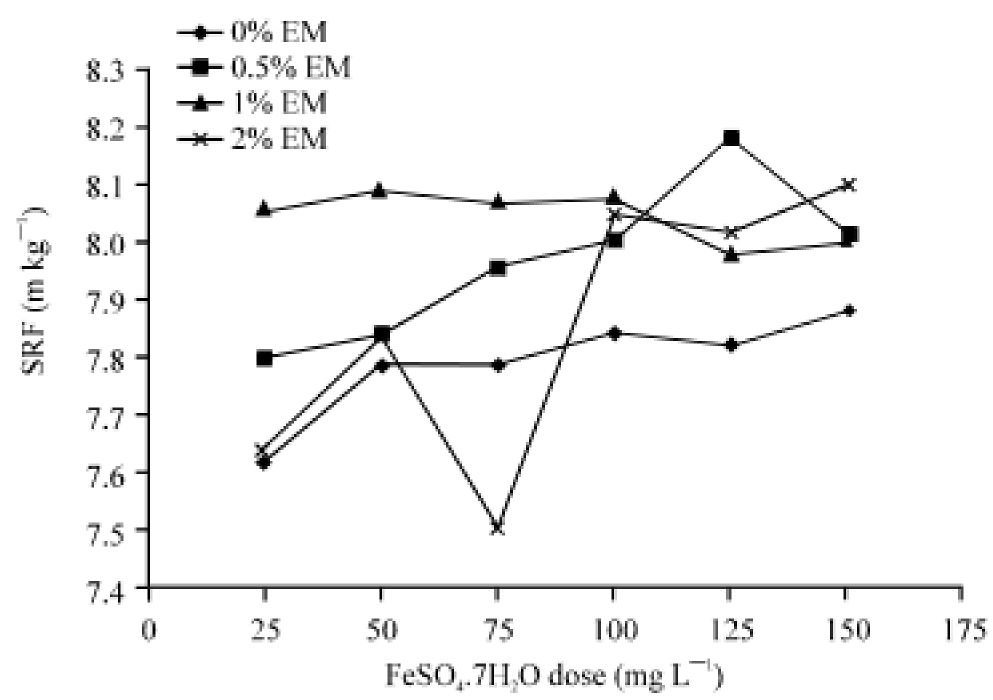

Fig. 9: pH value of filtrate corresponding to EM and Ferrous Sulfate dosage

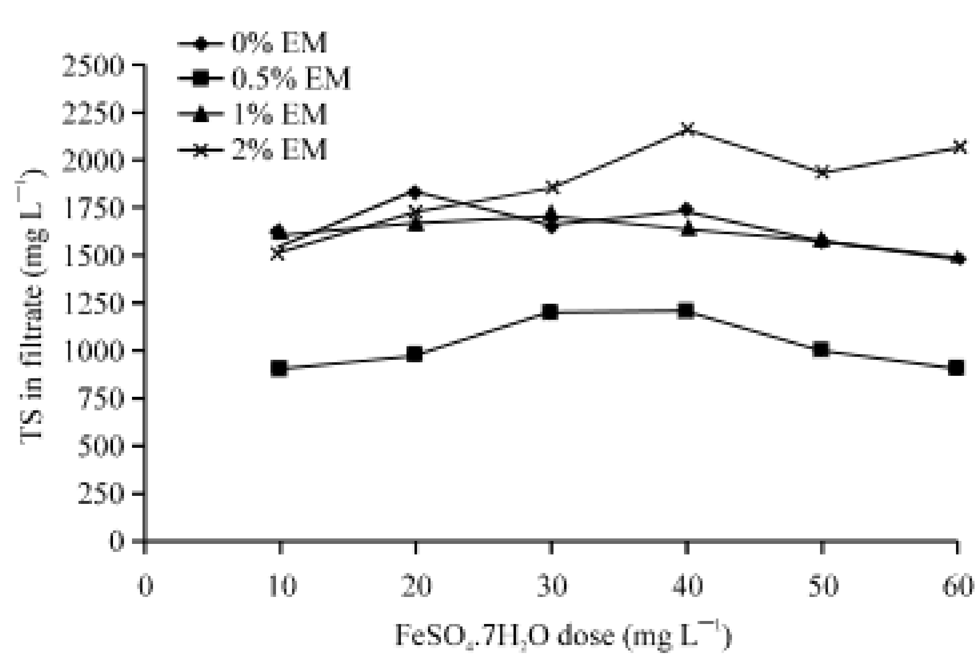

Fig. 10: Variations of total solids concentration in filtrate at sequential dosages of EM and Ferrous Sulfate

Reasonably, increasing bio-solids by EM seeds and coagulant dosages increase solids concentration in filtrate, but as shown in Fig. 10 there are no noticeable changes in filtrate solids, as ferrous sulfate and EM increments in sequential dosage, comparatively to the high bio-solids content in the raw and conditioned sludge that ranged (50-63.2%), respectively.

DISCUSSION

Lime

Divalent Cation Bridging (DCB) theory seems to describe best the major role of cations in bioflocculation, where, at higher pH values, bacteria leave the isoelectric state, resulting an increment in the number of active sites on cells leading to an improvement in the potential for bridging and bioflocculation (Katja and Mika, 2007; Weber, 1972).

The exception course of SRF curve at 2% EM could contribute to the increment in bio-solids production from (54-69%) at 0 and 2% EM concentrations, respectively where this increment demanded high concentrations of the coagulant (lime) that confers positive charges on the sludge particle surface, thus resulting in a charge reversal and re-dispersing the particles and an increase in SRF (Amudaa and Amoob, 2006; Eckenfelder, 2000).

pH as a factor that influences coagulation is important because the addition of lime automatically raises pH, which may cause further reduction in contaminants removal. This pH rising does not render the sludge permanently stable because carbonate precipitates as calcium carbonate resulting in a drop of the pH eventually (Al-Malack *et al.*, 2002). The need to employ higher dose of coagulant may pose a health hazard as a result of residual quantities of excess chemical additives.

The exertion of small flocs, due to re-dispersion, may restrict water penetration throughout the cake thickness, where a sharp diminishing in solids concentration was observed that may clarify the SRF increasing at 2% EM and the referring doses of lime.

Alum

Since, sludge alkalinity ranged from740 to 1350 mg L^{-1}; the gelatinous floc of the insoluble aluminum hydroxide that was formed due to reactions of calcium and magnesium bicarbonate alkalinity serves as a skeleton builder, which settles slowly, sweeping out bio-solids and forming a permeable rigid lattice structure that can remain porous after the cake growth of the filtration, thereby maintaining the size of the micro-passages through which water is expressed (Metcalf and Eddy, 2003; Buyukkamaci and Kucukselek, 2006).

pH raising may contribute to the extent of the dissociation of the hydrolysis products (aluminum hydroxide) producing a negative ions (hydroxide), which has significant effects on raising pH (Metcalf and Eddy, 2003). Furthermore, the increasing in bio-solids production throughout hydrolysis processes releases eventually greater amount of hydroxide ions.

Ferrous Sulfate

Despite that, ferrous sulfate has the same hydrolysis mechanism in chemical reactions that produce metal hydroxyl (ferrous hydroxide) that can be oxidized to ferric hydroxide to form a bulky gelatinous floc similar to the alum floc, but it seems to have an offender course with EM seeds increments, apparently at 2% EM. Possibly, increasing the bio-solids reversed the surface charge of the floc due to the continued adsorption of mono and poly-nuclear hydrolysis species that the colloidal flocs are positively charged and cannot be removed by perikinetic flocculation. Nevertheless, ferrous sulfate dosages augmentation have a minor effect on SRF, excluding 2% of EM scheme (Metcalf and Eddy, 2003; Katja and Mika, 2007).

The presence of anions (Sulfate ion) will alter the range of effective flocculation, increase the acidity and decrease the alkalinity (Metcalf and Eddy, 2003; Eckenfelder, 2000; Weber, 1972). The inferring results demonstrated in Fig. 9 shows contrary course, in as much as the dissociation of ferrous sulfate liberates hydroxyl ions which eventually, raised the pH value. Since, the operating region for iron precipitation ranged from about 7.0 to 9.0, with minimum solubility occurring at a pH of 8.0.

CONCLUSIONS

The SRF value, close to 1.265×10^{12} m kg^{-1} at 0% EM and 20 mg L^{-1}lime dose, was sensibly lower than the ones obtained for 2% EM, which was 7.65×10^{12} m kg^{-1} at the same Lime dose, while Lime dosages increment had a salient effect (about 0.9%) on SRF values. However, the sharp diminishing in bio-solids concentration in filtrate was observed at 2% EM.

Alum was one of the best conditioners, for minimum SRF 0.98348×10^{12} m kg^{-1}. The optimum Alum dosage was 60 mg L^{-1}at 1% EM, while at 40 mg L^{-1} and 2% EM, SRF was1.1087×10^{12} m kg^{-1}. Slight variations in filtrate solids were noticed with EM and Alum dosages increment.

The linear regression of the experimental data indicates that the significant effect of EM and Alum dose, in SRF reduction, was about -47.9 and -32.8%, respectively.

Ferrous sulfate in dewatering process was analogous to Lime. The maximum and the minimum SRF value was 10.85×10^{12} m kg^{-1}, 1.37×10^{12} m kg^{-1} at 2 and 0% of EM, respectively at the same coagulant dose which was 100 mg.L^{-1}.

SYMBOLS

SRF = Sludge Resistance to Filtration (m kg^{-1})
EM = Effective Microorganisms
CST = Capillary Suction Time (sec)
P = The vacuum pressure of filtration (N m^{-2})
μ = The viscosity of filtrate (N sec m^{-2})
V = The volume of filtrate (m^3)
t = The filtration time (sec)
w = The weight of dry solids per volume of filtrate (kg m^{-3})
A = The area of the filter paper (m^2)
R_f = Te resistance of filter medium (m^{-1})

REFERENCES

Al-Malack, M.H., N.S. Abuzaid, A.A. Bukhari and M.H. Essa, 2002. Characterization, utilization and disposal of municipal sludge: The state of-the-art. Arabian J. Sci. Eng., 27: 3-27.

Amudaa, O.S. and I.A. Amoob, 2006. Coagulation/flocculation process and sludge conditioning in beverage industrial wastewater treatment. J. Hazardous Mater., 141: 778-783.

Attar, M.H., B. Bina and K. Moeinian, 2005. Effects of aeration rate and detention time on thermophilic aerobic digestion of mixed sludge and its dewaterability. Int. J. Environ. Sci. Technol., 2: 105-111.

Berktay, A., 1998. Properties of sludge produced from the pressurized wastewater treatment process. Turk. J. Eng. Environ. Sci., 22: 377-385.

Buyukkamaci, N. and E. Kucukselek, 2006. Improvement of dewatering capacity of a petrochemical sludge. J. Hazardous Mater., 144: 323-327.

Calcutt, T. and J. Moss, 1984. Sewage sludge treatment and disposal-the way ahead. WPC, 83: 163-171.

Chia-Hung, H. and L. Kung-Cheh, 2003. Assessment of sludge dewaterability using rheological properties. Zhôngguó Gôngchéng Xuékân, 26: 221-226.

Eckenfelder, W.W., 2000. Industrial Water Pollution Control. 3rd Edn., Mc. Graw-Hill, Boston.

Farzadkia, M., N. Jaafarzadeh and L. Loveimi, 2007. Feasibility study of lime addition method for biosolids stabilization in West Ahwaz wastewater treatment plant. Proceedings in Specialist Conference Supported by Water Environment Federation, Jun. 24-27, Moncton, New Brunswick, Canada, pp: 1129-1133.

Ghiaus, A.G., J. Vaxelaire, M. Roques and D. Anglerot, 2000. Dewatering of activated sludge-a global approach. Proceedings of the 12th International Drying Symposium (IDS 2000).

Katja, H. and S. Mika, 2007. Flocculation in paper and pulp mill sludge process. Res. J. Chem. Environ., 11: 96-103.

Lee, D.J. and C.H. Wang, 2000. Theories of cake filtration and consolidation and implications to sludge dewatering. Water Res., 34: 1-20.

Metcalf and Eddy, 2003. Wastewater Engineering Treatment and Reuse. 4th Edn., McGraw Hill, New York, America.

Viessman, W. and M. Hammer, 1985. Water Supply and Pollution Control. 4th Edn., Harper and Row Publishers, New York, ISBN: 0-06-046821-1.

Weber, W.J. Jr., 1972. Physicochemical Processes for Water Quality Control. Wiley-Interscience, New York, ISBN-13: 9780471924357, pp: 533-596.

Zimmermann, I. and J. Harrison, 2004. Example of an integrated biosystem incorporating effective microorganisms. Proceedings of the Water Management Conference, Jun. 16, Worcester Polytechnic Institute, pp: 1-3.