Enhancement of Biogas Production from Anaerobic Digestion of Disintegrated Sludge: A Techno-Economic Assessment for Sludge Management of Wastewater Treatment Plants in Vietnam

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1 Introduction

1.1 Waste Activated Sludge

The activated sludge process is the most widely used biological wastewater treatment for both domestic and industrial plants in the world (Grady et al. 1999; Tchobanoglous et al. 2003). The basic function of a wastewater biological treatment process is to convert organics to carbon dioxide, water and bacterial cells. The cells can then be separated from the purified water and disposed of in a concentrated form called excess sludge. It must be realized that the excess sludge generated from the biological treatment process is a secondary solid waste that must be disposed of in a safe and cost-effective way (Liu 2003). The increased excess sludge production is generating a real challenge in the field of environmental engineering technology (Velho et al. 2016). For example, a quasi-exponential growth of excess sludge production in the USA has been reported (Seiple et al. 2017). In the European Union (EU), some 6 million tons of sludge were generated annually in 1998. By the year 2005, within the EU, about 10 million dry tons per year were generated (Christodoulou and Stamatelatou 2016). Vietnam is also facing the challenge of trying to keep pace with the increasing environmental pollution associated with rapid urbanization and industrialization. Over the past 20 years, Vietnam has made considerable effort to develop urban sanitation policies, legislations and regulations and to invest in urban sanitation including wastewater treatment plants (WWTPs)

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(WorldBank 2013). Most of the domestic wastewater in urban areas is not centrally treated but only treated by household's septic tank and discharged directly into the environment, such as rivers, lakes and streams. Only a few big cities have centralized wastewater treatment plants (Bao et al. 2013). Currently, in Vietnam 35 urban WWTPs had been constructed in Hanoi, Ho Chi Minh City and Da Nang, Quang Ninh, Vinh, Dong Hoi, Quy Nhon, Nha Trang, Da Lat and Buon Ma Thuot cities with a total capacity of 850,000 m^3/day. Some 40 new WWTPs are in the design or construction phase with a capacity of 1,600,000 m^3/day (Nga 2017). Industrial parks in Vietnam have developed rapidly since 1986, and there are 289 industrial parks throughout the country (Thuy et al. 2016). Until now there have been about 90% of the industrial parks which have WWTPs that are operated or under constructed (Pham et al. 2016). To date, in Vietnam the wastewater treatment technology has been focused on the use of some form of activated sludge secondary treatment technology, such as conventional activated sludge (CAS), anaerobic-anoxic-aerobic (A2O), oxidation ditch (OD) and sequencing batch reactor (SBR) technologies (WorldBank 2013, 2014). During operation, a lot of sludge has been discharged. The sludge disposal has been a significant challenge and attracted great attention in both academic and engineering fields. It should be noted that the cost of the excess sludge treatment and disposal can account for 30–40% of the capital cost and about 50–60% of the operating cost of many wastewater treatment facilities (Nowak 2006; Wang et al. 2013). Moreover, the conventional disposal methods such as landfill or ocean dumping may cause secondary pollution problems and are strictly regulated in many countries (Oh et al. 2007). In Europe, sea disposal has not been used since the end of 1998. In 2000, landfill of dewatered sludge became illegal in the Netherlands. Landfilling of sludge has been totally forbidden in Europe since 2015 (Christodoulou and Stamatelatou 2016). However, in Vietnam, sludge discharged from wastewater treatment plants were mostly disposed by the conventional method such as landfill (Uan et al. 2016). Therefore, reduction of sludge volume before disposal is very important for sludge transportation and management. It should be noted that the anaerobic digestion could be used as a common method of sludge stabilization (Appels et al. 2008). Besides, anaerobic digestion is a burgeoning technology and has lately captivated much attention owing to the need for sustainable energy production (Appels et al. 2013; Browne et al. 2014; Kavitha et al. 2015a, b). Therefore biogas produced through anaerobic degradation of sludge in wastewater treatment plants has gained much more attention as it is a renewable energy resource (Appels et al. 2008; Yu et al. 2016).

1.2 Anaerobic Digestion of Waste Activated Sludge

Basically, sludge was thickened before introducing to anaerobic degradation process in order to reduce the sludge volume. Besides, hydraulic retention time is identical to solid retention time, leading to a larger volume of anaerobic digester (Wang et al. 2013). Anaerobic digesters have retention times in the range of 20–30 days, and approximately only half of the organic material fed to anaerobic digestion could be degraded and subsequently transformed to methane (Uma Rani et al. 2014). In the anaerobic degradation systems, *Proteobacteria and Bacteroidetes* were the two most predominant phyla in the digested sludge samples (Chouari et al. 2005). *Proteobacteria* are able to degrade a wide range of macromolecules (Chouari et al. 2005; Costa et al. 2017). Bacteroidetes, known to be proteolytic bacteria, are involved in protein degradation and able to ferment amino acids to acetate (Riviere et al. 2009; Świątczak et al. 2017; Yang et al. 2014). Analysis of microbial community structure suggested *Bacteroidetes* and *Firmicutes* were the dominant species when ozone pretreatment was applied (Li et al. 2017b). However, conventional anaerobic degradation process needs substantial improvements, especially for the treatment of sludge with low solids content and poor anaerobic biodegradability (Appels et al. 2008; Carrère et al. 2010; Jimenez et al. 2014; Jin et al. 2004). Besides, recently, anaerobic degradation of sludge has been enhanced by using the submerged anaerobic membrane bioreactor (Baêta et al. 2016; Yu et al. 2014, 2016). Co-digestion of wastewater sludge and food waste has been also applied in full scale anaerobic digesters for biosolids management and biogas generation (Amha et al. 2017; Fitamo et al. 2017; Nghiem et al. 2017). The anaerobic degradation process is achieved through several stages: hydrolysis, acidogenesis, and methanogenesis. For sludge degradation, the rate-limiting step is the hydrolysis. It should be noted that the initial phase of anaerobic degradation, hydrolysis, is considered to be the rate-restricting phase (Chen et al. 2013). Hence, sludge disintegration has to be done prior to the anaerobic degradation in order to increase the hydrolysis rate.

1.3 Sludge Disintegration Methods

It should be noted that in anaerobic degradation process, sludge hydrolysis leads to the rupture of cell walls and the release of extracellular polymeric substances, which provides soluble organic substrates, such as dissolved organic matter, for acidogenic microorganisms (Appels et al. 2008). It was observed that pretreatment has to be done prior to anaerobic degradation to enhance the biogas generation. The most attractive way to enhance anaerobic degradation performance is by pretreating the waste to convert insoluble organic polymers into soluble components (monomers) (Nazari et al. 2017). By doing this it is possible to enhance hydrolysis rate, which subsequently increase the biogas generation, as well as reduce the digestion time and the amount of final residuals (Banu and Kavitha 2017). There are various sludge disintegration techniques attracted attentions as promising alternatives to reduce sludge production. Sludge disintegration techniques have been reported to enhance the biodegradability of sludge. Sludge disintegration methods reported in the literature include both physical methods such as ultrasound (Han et al. 2013; Hirooka et al. 2009), ball mill, and homogenizer treatments and chemical methods such as ozone (Muz et al. 2014; Sallanko and Okkonen 2009; Vlyssides and Karlis 2004; Aquino and Pires 2016), acid (Velho et al. 2016) and alkali treatments (Do et al.

Table 1 Sludge disintegration processes

Methods	Disintegration processes			References
Mechanical	Stirred ball mill	High-pressure homogenizer	Ultrasound	Øegaard (2004), Appels et al. (2013)
Physical	Thermal treatment	Osmotic shock	High-yield pulse	Atay and Akbal (2016), Banu and Kavitha (2017)
Chemical	Acid or base hydrolysis	Oxidation with ozone	Oxidation with H_2O_2/O_2/Fentons reagent	Fang et al. (2014), Chen et al. (2013)
Biological	Enzymatic lysis	Autolysis	Cell lysis-cryptic growth	Chouari et al. (2005), Velho et al. (2016)

2009; Oh et al. 2007). Besides, thermal treatment (Higgins et al. 2017; Kim et al. 2016), cell lysis-cryptic growth (Romero et al. 2013) and enzyme treatment (Ohsaka 2005; Song et al. 2013) have also been tested. Recently, a novel and energy-efficient radio frequency pretreatment system has been developed for anaerobic digestion of municipal sludge (Barrios et al. 2017; Hosseini Koupaie et al. 2017). The aim of all pretreatments is to disintegrate the sludge flocs, disrupting the cell wall, thus releasing and solubilizing intracellular material into the liquid phase. Several researchers have reported affirmative synergistic upshots of the combined pretreatment methods on subsequent anaerobic digestibility (Feki et al. 2015; Kavitha et al. 2015b; Li et al. 2017a, b; Pilli et al. 2015; Şahinkaya et al. 2012; Tyagi and Lo 2012; Wang et al. 2016b; Zhao et al. 2017; Zhen et al. 2014). In this process, the sludge disintegration enhances transformation of particulate organic compounds into more readily biodegradable substances and subsequently accelerates the process of anaerobic methane production (Do et al. 2009; Hirooka et al. 2009; Li et al. 2017c; Liu 2003; Nowak 2006; Øegaard 2004; Oh et al. 2007; Pérez-Elvira et al. 2006; Wang et al. 2016a; Yu et al. 2016). The disintegration processes are based on mechanical, thermal, chemical or thermochemical and biological techniques shown in Table 1.

Basically, mechanical sludge disintegration methods are generally based on the disruption of microbial cell walls by shear stresses. Cells are disrupted when the external pressure exceeds the cell internal pressure (Banu and Kavitha 2017). Mechanical disruption of sludge has gained acceptance due to its various successful industrial scale applications. The high-energy levels were most probably the reason why the application of mechanical disruption methods is still limited. Heat treatment results in the breakdown of the gel structure of the sludge and the release of intracellular bound water. Thermal hydrolysis involves heating of the sludge. Increased temperature had a major positive effect on the yields of soluble COD (Atay and Akbal 2016). Apparently, the origin of the sludge is of primary importance for the final solubilization to be reached with thermal hydrolysis. In chemical and/or thermochemical hydrolysis techniques, an acid or base is added to solubilize the sludge cells (Ødegaard et al. 2002). Whereas for thermal destruction methods high temperatures are required to achieve acceptable results, the thermochemical treatments are often carried out at lower or ambient temperatures. With respect to alkaline pretreatments, variable results have been found. An additional advantage of alkali instead of acid is that it is readily compatible with subsequent biological treatment (Wang et al. 2016b). Biological hydrolysis can be considered as a partial anaerobic sludge digestion. The biochemical sludge disintegration processes are based on enzyme activity that is either produced within the system. Biological hydrolysis is an easy and inexpensive method for the in situ production of a readily degradable carbon source for nutrient removal (Velho et al. 2016). An additional advantage is that less sludge is produced, compared with a system with external carbon addition.

1.4 Brief Economic Assessment of Sludge Disintegration Methods

The strategies for sludge disintegration should be evaluated and chosen for practical application using costs analysis and assessment of environmental impact. Economic savings from the reduced costs of treatment and disposal of biomass improved operational efficiencies and reduced environmental burden with lower disposal requirements (Atay and Akbal 2016). Other economic, operational and environmental costs may be incurred and these must be considered. The environmental impact, e.g. odour problems, should be assessed (Carrère et al. 2010). The performance of some disintegration methods can be compared with each other using the specific energy, which is defined as the amount of energy that stresses a certain amount of sludge. Müller (2001) has carried out a comparison in terms of five aspects (i.e. rate of sludge degradation, degree of sludge degradation, bacteria disinfection, influence on the dewatering results and odour generation). The author found that the mechanical methods contributed an excellent role in sludge degradation rate. However, the chemical methods using ozone could give the highest degree of sludge degradation compared with other methods. Among the summarized methods, the thermal methods could have a strong effect on bacterial disinfection. It seems that the odour generation was not affected by the sludge disintegration methods. It should be due to the sludge disintegration processes that were mostly carried out in the closed reactors.

It should be noted that the mechanical disintegration has been investigated primarily on laboratory to pilot scale. Problems encountered were the heating of the cell suspension because of the high shear stresses the sludge cells are being subjected to. Moreover, mechanical disintegration often appears to require high capital equipment and is energy intensive (Han et al. 2013). On the other hand, thermal and thermochemical treatments require high temperatures and high pressures to achieve acceptable results. Not only is equipment needed to raise the temperature and the pressure, also expensive construction materials are required in order to prevent corrosion problems (Carrère et al. 2010). Furthermore, odour problems can be encountered in thermal hydrolysis techniques. Most authors

Table 2 A comparison of several sludge pretreatments

Parameters	Relatively high →			Relatively low	References
Energy demand	Lysate centrifuge	Stirred ball milling	Sonication	Ozonation	Müller et al. (2004)
Sludge degradation	Ozonation	Stirred ball milling	Sonication	Lysate centrifuge	Carrère et al. (2010)
Polymer demand for dewatering	Ozonation	Sonication	Stirred ball milling	Lysate centrifuge	Fang et al. (2014)
Soluble COD and ammonia concentrations in supernatant after dewatering	Ozonation	Stirred ball milling	Lysate centrifuge	Sonication	Yang et al. (2013)

mention that acidic or basic conditions should be applied in combination with elevated temperatures, thereby creating quite aggressive reaction conditions (Yang et al. 2013). Moreover, raising or lowering the pH requires the addition of chemicals which increase the ionic strength of the sludge (Fang et al. 2014). If the hydrolysate is used in biological applications, e.g. anaerobic digestion or nutrient removal, subsequent neutralization is required, which again implies the addition of chemicals. In addition, due to high costs caused from ozone production, it is important to decrease the amounts of ozone required for sludge reduction.

It has been reported that one of the most significant inputs, environmentally and financially, is energy. While the cost of treatment may be disposal driven, in energy terms, energy utilized should hopefully match the energy produced by increases in biogas production. The energy input depends heavily on the method and may be a function of sludge composition, operating and ambient conditions and equipment used, among others (Carrère et al. 2010). A comparison of several sludge pretreatments such as stirred ball milling, ozonation, lysate centrifugation and sonication could be classified according to the aspects below (Table 2).

The evaluation of energy balance and cost assessment should be used in the conventional data such as industrial power price of 0.23 USD/kWh, NaOH price of 345.6 USD/ton and sludge treatment and disposal cost of 441.2 USD/ton TS. Meanwhile, energy stored in the increased methane volume as a result of pretreatment reached 378.15 kWh and 751.08 kWh, respectively (Kavitha et al. 2016). An energy balance and cost assessment of the thermo-chemo-sonic disintegrated sludges were performed by Kavitha et al. (2015b). In the study, the three alkalis (NaOH, KOH and $Ca(OH)_2$) were used in the thermo-chemo-sonic disintegrated sludge process. The energy balance and cost evaluation were based on the energy content of the biogas produced from both samples. The energy utilized for the mechanical stirring and pumping of the sludge was taken into account. At the same time, the energy content of the biogas for all three samples was calculated to be 1213 kWh, 1043 kWh and 927 kWh, respectively. Moreover, to evaluate the economic viability of the disintegration process, the present study took into account

Table 3 Cost comparison of sludge disintegration methods

No.	Sludge disintegration methods	Cost	References
1	Disperser induced microwave pretreatment	A net profit of 104.8 USD/t TS was achieved for disperser induced microwave pretreated sludge	Kavitha et al. (2016)
2	Electric + alkaline	Highest methane yield of combined electrical alkali pretreatment spurred a 20.3% increase	Zhen et al. (2014)
3	Alkaline + pressure homogenization	An enhancement of 247 mL/g VS methane production and of 43.5% VS removal was achieved	Fang et al. (2014)
4	Microwave + alkaline	Nearly 83.39 USD/t TS was gained from the combined pretreatment of microwave and alkaline with a 13% increase in biodegradability and a 28% VS reduction	Yang et al. (2013)
5	Ultrasonic + alkaline	Total revenue of 21.54 USD/t TS when the VS destruction rate is increased to 46% and biogas price reached 10 USD/GJ	Park et al. (2012)

an estimation of the operational cost (including consumable chemicals) and the decreased amount of the solids to be disposed. It was noted that a positive net profit was achieved in all three samples. The net profit of all three sludges (NaOH, KOH and $Ca(OH)_2$ was calculated to be 42.6 USD, 20.6 USD and 4 USD, respectively. The presently attained net profit was found to be comparatively higher than those obtained in other studies in which Houtmeyers et al. (2014) achieved 2.48 USD and 3.02 USD as net costs for ultrasonic and microwave pretreatment. It should be noted that in low-temperature thermal treatment, it would take from several hours to a few days to achieve the maximal disintegration effect (Şahinkaya et al. 2012). This strongly implies that it was possible to achieve 20% solubilization only in this combined novel process with lesser energy consumption. Table 3 shows a cost comparison of sludge of some combined sludge disintegration methods.

Carrère et al. (2010) have analysed the energy consumption for the various sludge disintegration methods, including non-mesophilic, non-thermophilic, biological, thermal hydrolysis, sonication, ball milling and high pressure. The energy consumption in those methods was mainly electrical and thermal. Electrical requirements are mainly feed and mixing and are approximately 0.1–0.2 kWh/m^3 days. The analysis also assumes a hydraulic retention time of 20 days for mesophilic or 15 days for thermophilic. It should be noted that the heating requirements are thermal capacity plus about 10% losses in mesophilic or 20% in thermophilic (Greenfield and Batstone 2005). The authors found that the electrical consumption was varied from 0.03 to 0.04 kWh/kg VS (volatile solids) for the non-mesophilic, non-thermophilic, biological and thermal hydrolysis methods. However, it was much higher in case of sonication (0.37 kWh/kg VS), ball milling (1.04 kWh/kg VS) and high pressure (0.33 kWh/kg VS). Basically, the thermal consumed in most of the methods was in the range of 0.5–1.0 kWh/kg VS, except thermal hydrolysis (2.0 kWh/kg VS) (Carrère et al. 2010).

1.5 General Objectives and Scopes of This Work

It should be noted that among the sludge disintegration methods, chemical hydrolysis using alkaline was the most efficient for inducing cell lysis (Do et al. 2009; Banu and Kavitha 2017). The chemical-combined activated sludge processes would be more efficient for sludge disintegration. The chemical assisted sludge disintegration processes have advantages of easy control, stable performance and high operation flexibility. The minimum effective sludge disintegration index for anaerobic digestion was reported to be 25% (Gayathri et al. 2015; Zhang et al. 2008). However, achieving solubilization in excess of 18% by thermochemical pretreatment was not cost effective (Jang and Ahn 2013) but results in the loss of organics (Chiang et al. 2012). Sonication, a cavitational process, was used by legions of researchers to achieve high degree of solubilization (40–50%). However, the practical applicability of sonic pretreatment was constrained because of its high energy cost (Şahinkaya et al. 2012; Zhang et al. 2008). Thus, in order to overcome the high energy requirement, the sonic pretreatment can be combined with other pretreatments to achieve the desirable solubilization with less energy consumption. Besides, the alkaline treatment is known to be relatively cheap resulting in a significant decrease of the total treatment cost. Therefore, in this study, sludge disintegration using alkalis such as sodium hydroxide (NaOH) and calcium hydroxide ($Ca(OH)_2$) was applied to sludge taken from Yen So and Kim Lien wastewater treatment plants in Hanoi (Vietnam). In particular, the study aimed to examine (i) the sludge solubilization and sludge dewatering ability during treatment, (ii) the change of the biodegradable matter and the particle size distribution and (iii) a brief economic evaluation of the alkali sludge disintegration and (iv) to test the biodegradability of sludge after alkali digestion. The digested sludge is subsequently treated anaerobically to generate biogas, and then the biogas can give long-term economic benefits.

2 Materials and Methods

2.1 Sludge Sources and Collection

In this study, waste activated sludge was collected from Yen So and Kim Lien wastewater treatment plants in Hanoi (Vietnam) (Fig. 1). These sludge samples were taken from oxic tank, SBR tank and sludge storage tank. The Yen So wastewater treatment plant works on the principle of SBR technology with a design capacity of 200.000 m^3/day, whereas the Kim Lien wastewater treatment plants work on A^2O technology with the capacity of 3500 m^3/day. The sludge samples were collected weekly and stored in refrigerator (4 °C) until use within 24 h. General characteristics of sludge samples are presented in Table 4.

Fig. 1 Sludge samples taken from Kim Lien and Yen So wastewater treatment plants

Table 4 General characteristics of sludge used in this study

Parameters	Kim Lien (storage tank)	Kim Lien (oxic tank)	Yen So (storage tank)	Yen So (SBR tank)
pH	7.1	7.3	7.2	7.2
MLSS, mg/L	4696	1850	9660	4948
MLVSS, mg/L	2728	820	5986	3034
MLVSS/MLSS	0.58	0.44	0.62	0.61
Soluble COD, mg/L	53	41	95	32
Total COD, mg/L	4410	1730	9740	5040

2.2 Experimental Setup and Procedure

Sludge disintegration experiments will be carried out at laboratory temperature using a jar apparatus with six paddle stirrers (Model: JLT6, VELP Scientifica, Europe). Alkalis were added under stirring. Rapid mix took place for 1 min at a speed of 200 rpm, followed by slow mix for 3 h at 30 rpm. During digestion, NaOH and $Ca(OH)_2$ were added to the reactor at various dosages, ranging from 0.2 to 2 g/L (Fig. 2).

The anaerobic digestion experiments were conducted in a lab-scale anaerobic digestion system (Figs. 3 and 4).

The anaerobic reactor is a round-bottomed reactor with a working volume of 5 L. The pretreated sludge was inoculated with an active methanogenic bacterial population for the quick startup of the reactor. Sludge from an anaerobic wastewater treatment using a covered lagoon was selected as the inoculum. A slow mixer was used to keep a completely mixed anaerobic digester. Biogas generation was calculated by the liquid displacement method, in which the difference in the water level in a cylinder linked to the reactors was measured. The displaced liquid is therefore considered to have the same volume as the produced biogas. The produced gas was collected in standardized glass cylinders filled with acidified deionized water to evade losses of CO_2 due to the formation of carbonates.

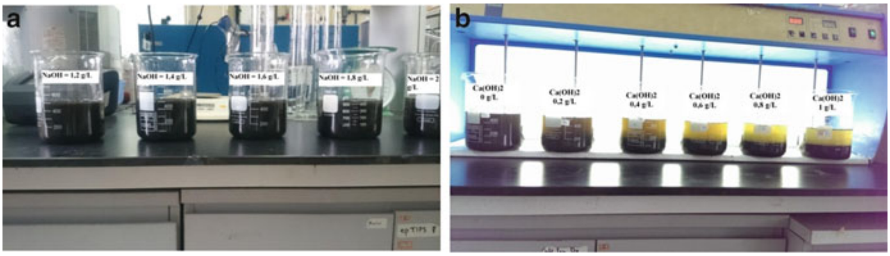

Fig. 2 Sludge digested in jar tests with different NaOH and $Ca(OH)_2$ dosages. (**a**) NaOH dosages. (**b**) $Ca(OH)_2$ dosages

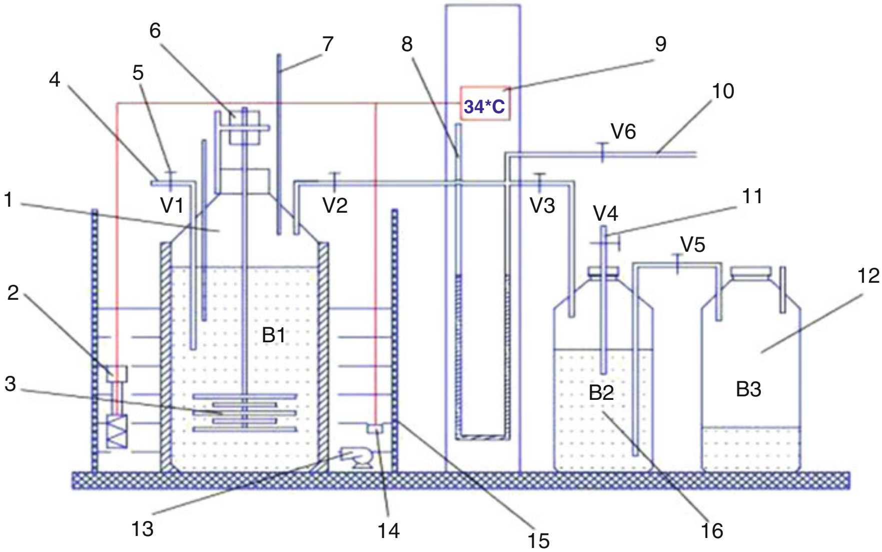

Fig. 3 Setup of a simplified test for measuring the biogas production. Notes: 1 Anaerobic reactor (V = 5 L), 2 Heating in water batch, 3 Mixer, 4 Feeding pipe, 5 Valve, 6 Motor, 7 Thermostart, 8 Pressure pipe, 9 Temperature control, 10 Biogas collection pipe, 11 Water addition pipe, 12 Gas volume bottle, 13 Mixing pump in water batch, 14 Temperature sensor, 15 Water batch cover, 16 Water bottle

2.3 Analytical Methods

Characteristics of sludge (including MLSS, MLVSS, sludge dewatering), and other parameters of the solution such as TP, soluble TP, total COD, soluble COD, BOD, TN, soluble TN, and NO_3^- were measured and analysed in accordance with the Standard Methods described in APHA (2012). Sludge disintegration efficiency was calculated as the ratio of the soluble COD (SCOD, mg/L) increase by the disintegration process to the total COD (TCOD, mg/L) of the sludge before disintegration.

Fig. 4 A photo of the anaerobic digestion system used in this study

$\alpha = \frac{SCOD - SCOD_o}{TCOD - SCOD_o}$ in which $SCOD_o$ (mg/L) is the soluble COD of the sludge before disintegration.

The biogas composition biogas was taken during experimental tests. The compositions of these samples (CH_4, CO_2, H_2S, O_2) were determined using an Optima 7 Biogas Analyzer (MRU Instruments, Inc., Germany).

3 Results and Discussion

3.1 Sludge Source Variation

Sludge used in this study were collected from the Yen So and Kim Lien wastewater treatment plants in Hanoi (Vietnam). It should be noted that the SBR technology with a design capacity of 200,000 m^3/day was used in the Yen So wastewater treatment plant, but the Kim Lien wastewater treatment plant used A^2O technology with a design capacity of 3500 m^3/day. The variation of sludge sample taken from sludge storage tank was presented in Fig. 5. As seen from the figure, the sludge concentration was very different from each other. In Yen So wastewater treatment plant, it was in a range of 4000 to about 10,000 mg/L. In case of Kim Lien wastewater treatment plants, it was around 2500 to over 6000 mg/L.

3.2 Sludge Solubilization

Alkalis added to a cell suspension react with the cell walls in several ways, including the saponification of lipids in the cell walls, which leads to solubilization of the membrane. Disruption of microbial cells then leads to leakage of intracellular material out of the cell. In this study, the collected sludge was tested with different

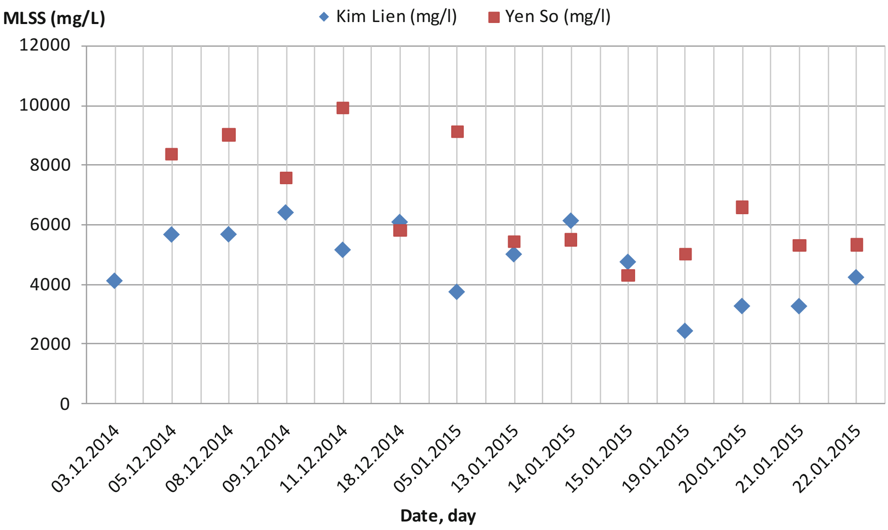

Fig. 5 Variation between MLSS of Yen So and Kim Lien wastewater treatment plants

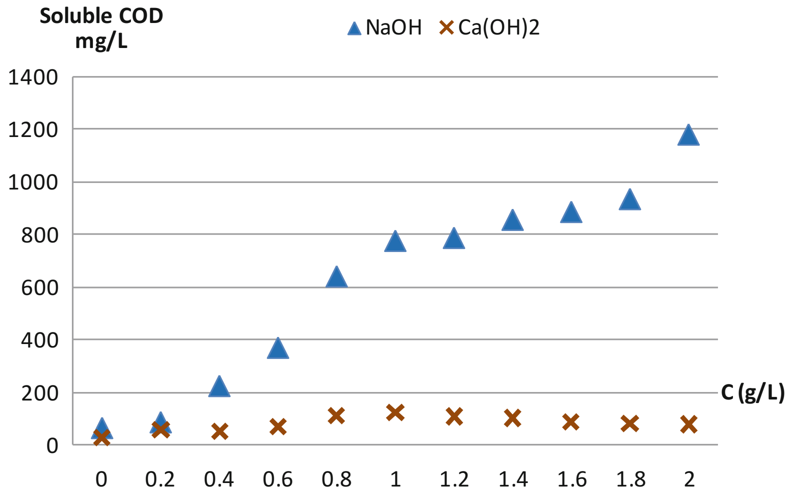

Fig. 6 Variation of soluble COD during digestion

NaOH and $Ca(OH)_2$ dosages. Each 1 L of sludge was added in six glass beakers and put in the jar test system. The first beaker was without added by any alkalis. The mining beakers were added with different NaOH and $Ca(OH)_2$ dosages, from 0.2 to 2 g/L. During digestion, soluble COD will be released out of the sludge. Figure 6 shows the variation of soluble COD during digestion.

It could be seen from Fig. 6 that the soluble COD was increased with the increase of NaOH. However, it was increased very low during adding $Ca(OH)_2$. When alkali agents were added, COD solubilization increased through various reactions such as saponification of uronic acids and acetyl esters, reactions occurring with free carboxylic groups and neutralization of various acids formed from the degradation of particular materials (Liu 2003). During digestion process using $Ca(OH)_2$, calcium ion could bind to the sludge surface through the exopolysaccharide polymer as well as assist in the bioflocculation of the sludge, resulted in reduction of the sludge solubilization (Bruus et al. 1992).

3.3 Sludge Mass Reduction

The reduction of MLSS during digestion was presented in Fig. 7. It was reduced from 6600 mg/L down to about 4800 mg/L when alkali dosage was increased from 1.0 to 1.8 g/L. Besides MLSS reduction, the soluble COD was also increased from less than 100 mg/L to over 1800 mg/L. Alkali treatment is a harsh method. At extremely high pH values of medium, the cell loses its viability, and it cannot maintain an appropriate turgor pressure and disrupts. Alkalis added to the cell suspension react with the cell walls in several ways, including the saponification of lipids in the cell walls, which leads to solubilization of membrane. The high alkali concentrations cause much degradation. Disruption of sludge cells leads to leakage of intracellular material out of the cell (Banu and Kavitha 2017). As a result, the sludge mass was decreased, and the COD was increased. Besides, a relationship between MLSS, MLVSS reduction and soluble COD increasing during digestion for different sludge samples was also measured and presented in Fig. 8. Depending on MLSS concentration, sludge reduction was varied from 10% to 18%.

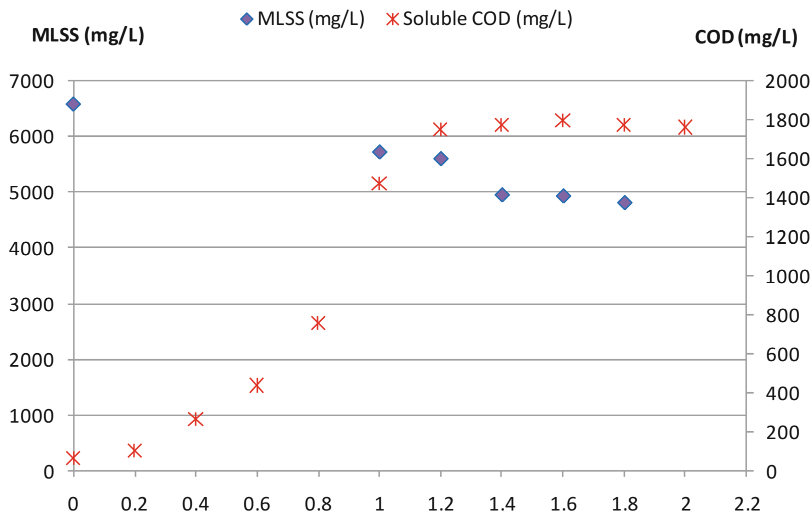

Fig. 7 Relationship between MLSS reduction and soluble COD during digestion

Fig. 8 Relationship between MLSS, MLVSS reduction and soluble COD increasing during digestion for sludge taken from storage tank of Kim Lien WWTP (**a**) and Yen So WWTP (**b**); sludge taken from oxic tank at Kim Lien WWTP (**c**) and from SBR tank in Yen So WWTP (**d**)

3.4 Enhancement of Sludge Dewatering

Sludge dewatering ability was tested by measurement of time to filter. At the same volume of digested sample, the filter time between NaOH and $Ca(OH)_2$ was much different (Fig. 9). The filter time of NaOH digested sample was much higher than $Ca(OH)_2$ digested sample, showing that $Ca(OH)_2$ could help improvement of sludge dewatering ability. One reason for the difficulty in activated sludge dewatering is the presence of extracellular polymer (ECP). ECP is present in varying quantities in sewage sludge, occurring as a highly hydrated capsule surrounding the bacterial cell wall and loose in solution as slime polymers (Houghton et al. 2001). One of the main influences on sludge dewaterability is the particle size distribution (Eriksson and Alm 1991). Flocculation changes the particle size distribution of a sludge, binding small particles together, thereby influencing the sludge dewatering characteristics. ECP can therefore be expected to have an influence on sludge dewaterability through the high level of hydration of the polymer surrounding the bacterial cell and its role in flocculation. Bruus et al. (1992) suggested that cations aid in flocculation by bridging negative sites on ECP which promotes an increase in the floc size, floc density and floc resistance to shear. Divalent cations act as a bridge between negatively charged sites on ECP resulted in improvements of sludge settling and dewatering.

Figures 10 and 11 show the settled volumes and SVI changes of various sludge samples before and after digestion. It seems that after digestion, SVI was improved slightly. The alkali disintegration alters sludge floc properties (such as size, water

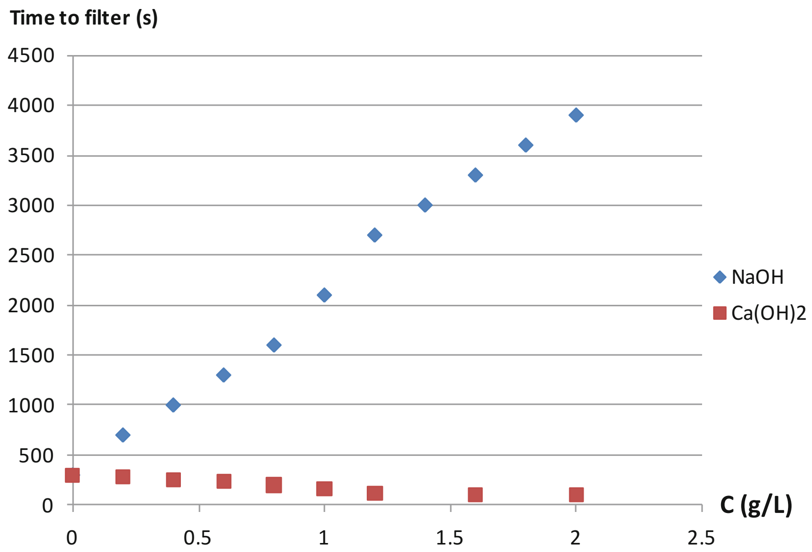

Fig. 9 Variation of time to filter during sludge digestion

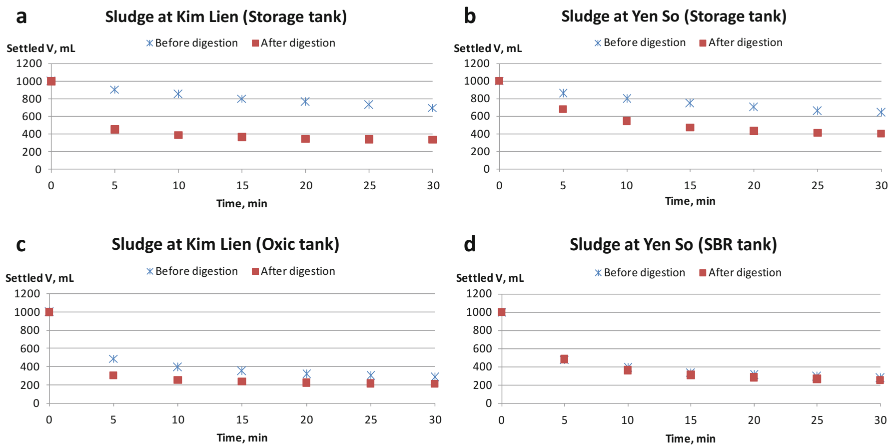

Fig. 10 Settled volumes of various sludge samples before and after digestion sludge taken from storage tank of Kim Lien WWTP (**a**) and Yen So WWTP (**b**); sludge taken from oxic tank at Kim Lien WWTP (**c**) and from SBR tank in Yen So WWTP (**d**)

content, etc.); it also modifies its settling and filtering properties. In particular, the alkali disintegration integrated in the wastewater handling units increases the settling rate and could reduce SVI. The SVI determines the settling capability of sludge. The value of SVI for alkali-disintegrated sludge was found to be in the range of 46–113 mL/g, which was comparatively higher than the others. It is known from the literature that the usage of Na^+ ions deteriorates the dewatering property of sludge, and it thereby demands the addition of sludge-conditioning aids (Banu et al. 2012).

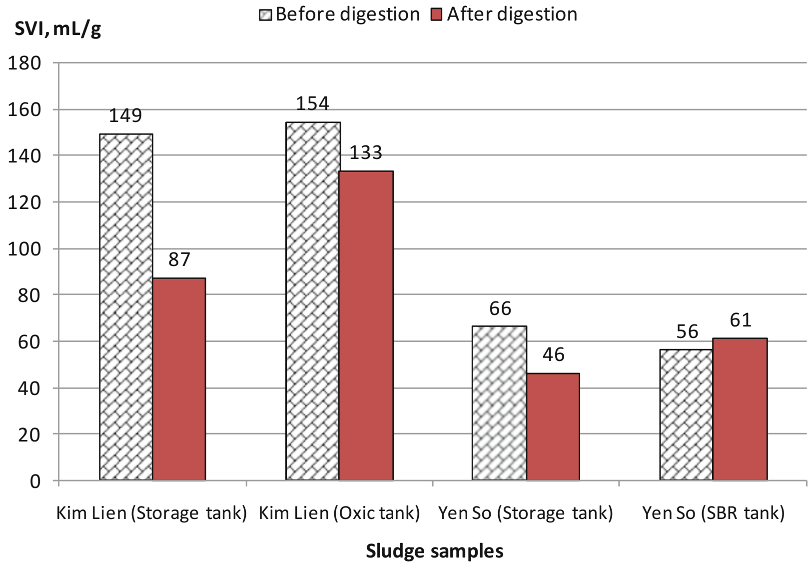

Fig. 11 SVI changes of various sludge samples before and after digestion

Table 5 Biodegradability of solubilized COD of sludge samples after digestion

Parameters	Kim Lien sludge (NaOH digestion)	Kim Lien sludge ($Ca(OH)_2$ digestion)	Yen So sludge (NaOH digestion)	Yen So sludge ($Ca(OH)_2$ digestion)
MLVSS before digestion, mg/L	2728	2728	5986	5986
MLVSS after digestion, mg/L	1684	2486	4004	4004
Soluble COD, mg/L	855	169	1534	207
BOD_5, mg/L	706	51	1228	73
BOD_5/COD (%)	83	30	80	35

3.5 Biodegradability of Solubilized COD After Alkali Digestion

The biodegradability of solubilized COD after alkali digestion was measured by BOD (Table 5). The results observed that the biodegradability of the solubilized COD by $Ca(OH)_2$ was low, around 30–35%. This unexpected phenomenon is probably due to the release of part of the sludge material by desorption or floc destructuration while no increase in its intrinsic biodegradability occurs. In case of NaOH digestion, the sludge biodegradability enhancement is linearly correlated to COD solubilization up to 80–83%. Due to its enhanced biodegradability, the supernatant of NaOH sludge digestion would be used as a carbon source to support post-

Table 6 Characteristics of sludge samples after digestion

Parameters	Kim Lien (storage tank)	Kim Lien (oxic tank)	Yen So (storage tank)	Yen So (SBR tank)
MLSS, mg/L	3824	1570	8688	4132
MLVSS, mg/L	1684	356	4004	1896
MLVSS/MLSS	0.44	0.23	0.46	0.46
Soluble COD, mg/L	855	204	1534	590
NO_3^-, mg/L	1.5	–	0.8	16.6
NH_4^+, mg/L	4.2	–	48.4	2.5
TN, mg/L	64	–	144	58
TP, mg/L	37.2	–	44.2	16.8
SVI, mL/g	87	133	46	61

denitrification in the biological nitrogen removal. During sludge disintegration, TN, NO_3^-, NH_4^+, TP and PO_4^{3-} releases were also monitored. The results were presented in Table 6. It should be noted that the structure of sludge flocs was significantly dispersed after alkali digestion. The average floc size decreased in the range of about 1–50 μm (Uan et al. 2017). Sludge flocculation capacity decreased with microorganism death.

3.6 *Anaerobic Digestion of Digested Sludge*

In anaerobic digestion process, the organic residues will be transformed by bacteria into the biogas through four sequential phases, i.e. hydrolysis, acidogenesis, acetogenesis and methanogenesis in anaerobic digestion process. The progression of anaerobic digestion has the prospective of transforming recyclable organics into biogas. On the other hand, in lack of a pretreatment, the biogas production potential of anaerobic digestion is inadequate due to the existence of refractory microbial cell walls and other organic materials present in the sludge. As a result, to enhance the biogas augmentation, pretreatment is believed to be the plausible option. Sludge disintegration was extended to boost bioenergy generation by hastening the hydrolysis pace of anaerobic digestion. To increase the biogas production, assorted pretreatment practices are applied. A comparatively current scientific progression which would probably be capable of formulating anaerobic digestion further was the expansion plus concern of preprocessing of wastes prior to anaerobic digestion to hasten the sludge degradability. Pretreatment enhances liquefaction and its rate, which consequently enhance biogas production. All pretreatments bring about the breakdown of sludge biomass, thus solubilizing and discharging substance present inside the aqueous phase and converting obstinate organic substances into recyclable nature consequently creating substances easily accessible to microbes. These entire pretreatments are exposed to enhance the biogas generation in subsequent anaerobic digestion process. Figure 12 presents the digestion curves obtained in the anaerobic

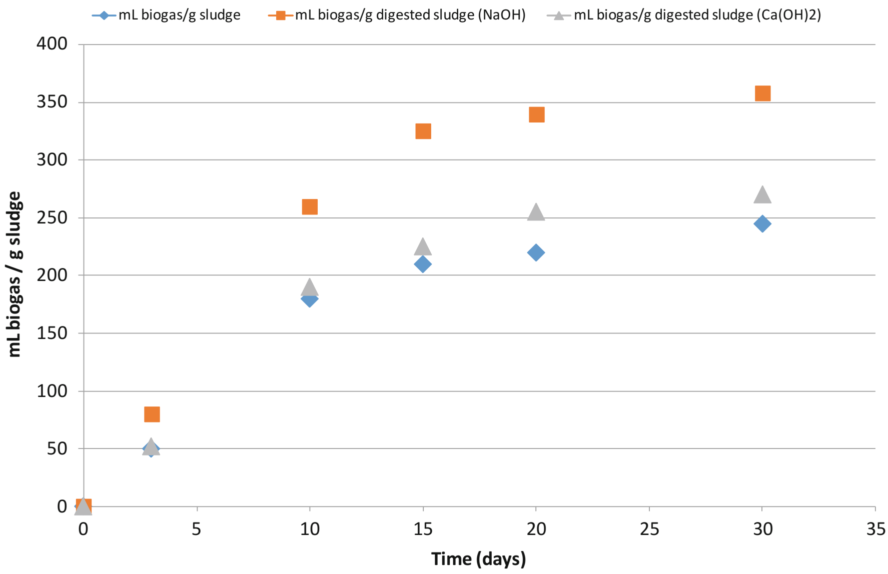

Fig. 12 Biogas production curves in anaerobic digestion tests with sludge with and without alkaline sludge disintegration

digestion tests. It can be observed that the biogas production of the sludge without the alkali disintegration was very slow. As expected, the biogas production of the disintegrated sludge was higher compared to the sludge without the alkali disintegration (358 and 245 mL biogas/g sludge added), representing a 46% increase in the biogas production in case of NaOH disintegration. However, it was only about 10% increase in the biogas production in case of $Ca(OH)_2$ disintegration. The fractioning of the disintegrated sludge showed that the contribution of the enhancement in biogas production and kinetics after the hydrolysis is due to the liquid fraction, remaining the solid fraction difficult to degrade.

The composition of biogas production during anaerobic digestion of the alkaline-disintegrated sludge was sometimes measured by using an Optima 7 Biogas Analyzer (MRU Instruments, Inc., Germany). The results show that the CH_4 composition is high at about 73%. It was tried to burn very well in the lab (Figs. 13 and 14). However, it was clearly that the H_2S concentration in the biogas is too high at about 1989 ppm. It means the biogas must be purified before using such as for burning or for electricity generation.

3.7 *Economic Analysis for Sludge Akali Disintegration*

As discussed above, the sludge can be disintegrated by mechanical, physical, chemical or biological processes. The main purpose of the disintegration processes

Fig. 13 A photo of the composition of biogas production during anaerobic digestion of the alkali-disintegrated sludge

Fig. 14 A photo of sludge after anaerobic digestion (left) and biogas generation burned (right)

is to improve the anaerobic digestion of the sludge. Therefore, a high degree of disintegration is necessary in order to realize a noticeable acceleration and enhancement of the disintegrated sludge degradation. However, it is not easy to recommend one or the other method to be applied, because of the lack of experience at full scale. A comparison between the capital costs and the operational and maintenance (O&M) costs among treatment processes is needed. Rough cost estimates are between 70 and 150 US$/t TS for capital and O&M costs (Müller 2001). It has been reported that the O&M costs vary considerably among sludge disintegration methods. The O&M costs for sludge disintegration by biological methods were lower compared with others. However, it should be noted that the biological methods required long retention time for the disintegration process, resulted in bigger volume of reactor. The high energy levels were most probably the reason why the application of mechanical disruption methods is still limited. Thermal methods also required high energy for heating. Besides, the thermal treatment time had less impact on sludge solubilization in comparison with temperature (Müller 2001). However, if temperatures are not high enough, several hours to days of heating was required (Zhen et al. 2017). The chemical methods required the chemical costs. During treatment, the corrosion issues should be considered. It is important that the sludge cells can be dissolved by acids of alkalis at low or ambient temperatures. Chemical methods could be realized at low costs if the conditions are appropriate (Uan et al. 2017).

In this study, a preliminary economic analysis was evaluated. In particular, the operating costs for alkali digestion were calculated based on the chemical consumption and TSS concentration or sludge volume. Besides, the costs of dewatering, transportation and landfill should be taken into account. In case of NaOH used for sludge digestion at 1.2 g/L (it means 0.12 kg of NaOH was used for 1 kg of TSS, and the cost of NaOH (99%) of about 0.3 US $/kg), the chemical consumption was estimated about 0.36 US $/kg TSS (about 8,100 VND/kg TSS). In Vietnam, the cost for transportation and treatment of sludge was varied from 0.20 to 0.44 US $/kg TSS (about 4500 to 10,000 VND/kg TSS), depending on the sludge characteristics and services. If the digested sludge is subsequently treated anaerobically to generate biogas, then it can be biogas can give long-term economic benefits.

It should be noted that 1 m^3 of biogas could produce hourly 2.14 kWh of electricity and 2.47 kWh of heat energy (Akbulut 2012). Therefore, an economic comparison between chemical consumed and benefits obtained from the anaerobic digestion for sludge without alkalis disintegration and the disintegrated sludge was presented in Table 7.

As seen in Table 7, the alkali disintegration of sludge shows a lot of benefits. Among the sludge disintegration processes, the chemical methods have advantages of easy control, stable performance and high operation flexibility. However, it is expected that the increased operation and capital costs due to chemical addition can be compensated from saving the cost of excess sludge posttreatment. In this sense, the chemical methods for sludge disintegration would be attractive and have great industrial potentials.

Table 7 An economic comparison for the alkali disintegration of sludge

Parameters	Sludge without disintegration	Disintegrated sludge	Benefits
Sludge mass reduction (%)	0 (6600 mg/L)	27% (4800 mg/L)	+
Sludge transportation and treatment (US$/ton TS)	320 (average)	234 (saved 27% due to sludge reduction)	+
Chemical consumption, g/L	0	1.2	−
Biogas enhancement, mL/g TS	245	358 (46%)	+
Electricity production, kWh (based on 1 ton TS)	524	766 (increase 46% due to biogas enhancement)	+

Note: + benefits, − negative

4 Conclusions

Sludge disintegration is used as a pretreatment step to enhance the sludge biodegradability before adding to the anaerobic digestion. The results obtained from the present study show that the sludge volume and mass were reduced significantly during digestion. MLSS reduced from 6600 mg/L down to about 4800 mg/L when alkali dosage was increased from 1.0 to 1.8 g/L. Soluble COD was increased from less than 100 mg/L to over 1800 mg/L. NaOH was an efficient reagent for inducing cell lysis and causes sludge solubilization. The sludge biodegradability enhancement is linearly correlated to COD solubilization. $Ca(OH)_2$ used for the sludge digestion could improve the sludge dewatering, but the sludge solubilization by $Ca(OH)_2$ was low. More importantly, sludge dewatering ability was increased much after sludge digestion showing that sludge management was benefited by sludge digestion. The anaerobic digestion tests show that the biogas production of the sludge without the alkali disintegration was very slow. The biogas production of the disintegrated sludge was higher compared to the sludge without the alkali disintegration (358 and 245 mL biogas/g sludge added), representing a 46% increase in the biogas production in case of NaOH disintegration. However, it was only about 10% increase in the biogas production in case of $Ca(OH)_2$ disintegration. The cost of chemical consumption for sludge disintegration was about 0.36 US $/kg TSS (about 8100 VND/kg TSS). Based on the preliminary economic assessment, if the sludge disintegration was carried out with a NaOH dose of 1.2 g/Lg, the increase in the biogas of 46% and 27% of sludge reduction could offset the cost of chemical. Therefore, the alkali sludge disintegration would be considered as a potential method for sludge management of wastewater treatment plants in Vietnam. Further works should be carried out to overcome including the reduction of chemicals and the prevention of corrosion issues.

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