BIOLOGICAL TREATMENT PROCESSES IN DRINKING WATER

CHAPTER 5: ANAEROBIC WASTEWATER TREATMENT PROCESS

5.1. Introduction

Anaerobic wastewater treatment is different from typical aerobic treatment. The controlled conversion of complex organic pollutions, chiefly methane and carbon dioxide, is achieved due to the absence of oxygen. Anaerobic treatment offers beneficial effects like low sludge production, removal of higher organic loading, biogas gas production, high pathogen removal, and low energy consumption. A preferable option to regular anaerobic digestion for industrial wastewaters that are released at medium to low temperature and municipal sewage, can be a psychrophilic anaerobic treatment (McCarty & Smith, 1986; Bernard et al., 2001).

Over the past few decades, the use of anaerobic wastewater purification processes has been increased. The importance of these processes is because of the favorable effects they have like methane gas production and low energy consumption, removal of higher organic loading, low sludge production and high pathogen removal (Nykova et al., 2002).

Both fundamental and applied research performed in this field is covered by the reviews of anaerobic processes. There are numerous areas into which the review is subdivided, they include biotransformation of toxic and recalcitrant compounds, microbiology, model development, toxicity, municipal solid waste treatment, reactor systems and new methods for testing of anaerobic processes (Lettinga et al., 1997; 2001). In the last decade, research activity has been expanded in the application of reactor technology to process different types of industrial wastewaters, like those from paper and pulp industry, textile industry, food processing (Rumana et al., 2000; Pantea & Romocea, 2008). It was given that high rate anaerobic systems offer low excess sludge production and production of biogas, low development cost, small land demands, low operation, and maintenance cost. Therefore, they provide sustainable and low-cost technology for industrial wastewater treatment.

Anaerobic digestion involves many biological reactions that are collective, complicated sequential and parallel. During these reactions, the substrates for the group of microorganisms are provided by the products from the previous group, as a result, the organic matter is transformed primarily into a mixture of carbon dioxide and methane. There are four phases in which anaerobic digestion occurs: hydrolysis/liquefaction, acidogenesis, acetogenesis, and methanogenesis. During the process, several biological conversion processes remain adequately coupled, it is important to avert the aggregation of any intermediates in the system to assure the balanced digestion process. *Bacteria* and *Archaea*, the microorganisms from two biological kingdoms, perform the biochemical process following rigid anaerobic conditions (Parawira, 2004; Cakir & Stenstrom, 2005).

Mainly the anaerobic reactors were used for the treatment of industrial wastewater. Researches have demonstrated that low-strength synthetic wastewater and high-strength industrial wastewater as well can be successfully treated by anaerobic systems like the Anaerobic Sequencing Batch Reactor (AnSBR), the Anaerobic filter (AN) and the Upflow Anaerobic Sludge Blanket (UASB). Anaerobic systems for municipal sewage treatment have very limited applications as yet. For maintaining high biomass (in the form of granules – suspended solids or fixed film) content in the reactor, the municipal sewage is too weak (too low BOD or COD), this is the predominant reason given for its limited applications. Nonetheless, the pilot and full scale have some successful examples. For the treatment of municipal sewage of an average BOD of 314 mgO_2/L for a hydraulic retention time of 10.3 hours, (organic loading rate 0.85 $kg/m^3 \cdot d$), Orozo (1997) examined a full scale anaerobic baffled reactor (AnBR) and attained a 70% removal efficiency. The fact that the process was carried out at a very low temperature, that is between 13 to 15°C has to be highlighted. Elmitwalli et al. (1999) conducted the treatment of two anaerobic hybrid (AnH) and domestic wastewater in UASB reactors at a temperature of 13°C. The AnH reactors removed 64% of total COD, which was greater than the removal in the UASB reactors for pre-settled wastewater treatment (Leitão et al., 2006; Hess & Bernard, 2008; Van Lier, 2008).

5.2. Environmental Benefits of Anaerobic Processes

Anaerobic treatment very adequately removes biodegradable organic compounds by itself, and leave mineralized compounds like NH^{+}, PO^{3-}, S^{2-} in the solution. Anaerobic treatment can be organized in technically plain systems, and the technique can be used at almost any place and at any scale. Furthermore, little and well-stabilized amount of excess sludge is produced, they even have a market value when there is the production of so-called granular anaerobic sludge in the bioreactor. Also, instead of high-grade energy consumption, useful energy is produced in the form of biogas (Tafdrup, 1995; González-González et al., 2013). Following the fact that anaerobic digestion hardly removes organic pollutants, sometimes it leaves serious problems, not even analogous to the start-up rate of the system. Supposing that 1 kWh of aeration energy is required for the oxidation of 1 kg COD, the fate of energy and carbon in both aerobic and anaerobic wastewater treatment (AnWT) is shown in Figure 5.1. Aerobic treatment, in comparison to anaerobic treatment, is generally characterized by high operational costs (energy), whereas the different type of waste (sludge) is produced by the transformation of a very large amount of the waste. Further treatment is required because about 50% (or more) new sludge from the converted COD, e.g. anaerobic digestion before it is reused, disposed-off or incinerated, is produced by the aerobic treatment in a typical activated sludge process. The arrangement of the related wastewater treatment system is broadly affected by the energy/carbon flow principles of anaerobic and aerobic bio-conversion. Expectedly, AnWT is considered as a competitive wastewater treatment technology until now. Anaerobic high-rate conversion processes now treat various types of organically polluted wastewaters, even those can be treated that were previously not supposed to be suitable for AnWT (Wilkie, 2005; Capson-Tojo et al., 2016). Anaerobic reactor systems are currently used, in countries like the Netherlands, to treat nearly all agro-industrial wastewaters. There is fast growth in the application potential, e.g. in the petrochemical industries. From the mid-seventies onwards, the steady increase in the number of anaerobic high-rate reactors is shown in Figure 16.2.

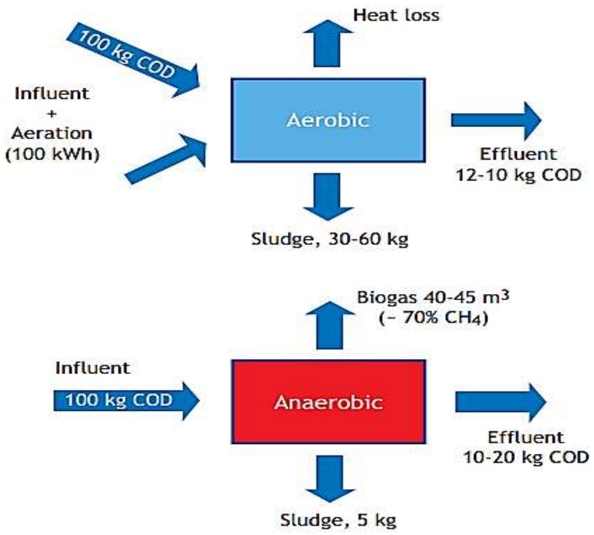

Figure 5.1. The fate of energy and carbon in aerobic (below) and anaerobic (above) wastewater treatment

[Source: https://ocw.tudelft.nl/wp-content/uploads/Chapter_16_-_Anaerobic_Wastewater_Treatment.pdf]

Famous companies like Biothane, Paques, Enviroasia, Biotim, Waterleau, ADI, Degremont, Kurita, GWE, Envirochemie, Grontmij also some other local companies, constructed a total number of 2,266 registered full-scale installations, that are currently in operation. Very small local companies or the industries themselves construct reactors that do not appear in the statistics, so an approximated number of 500 'homemade' reactors can be added to this number (Abramowicz, 1995; Nkoa, 2014).

The major advantages of AnWT over typical aerobic treatment systems are following by which the reasons for the selection for AnWT can also be analyzed.

1. Up to 90% decrement in excess sludge production.
2. When expanded sludge bed systems are used, the reduction of space requirement is also up to 90%.
3. About 20-35 kg is reached by the high suitable cod loading rates.
4. Smaller reactor volumes are required by the Cod per m^3 of reactor per day.
5. Depending on aeration efficiency, about 1 kwh/kgcod removed is saved, since there is no use of fossil fuels for treatment.

6. Production of about 13.5 mj ch_4 energy/kgcod
7. Removed, giving 1.5 kwh electricity (assuming 40% electric conversion efficiency).
8. Using granular anaerobic, there is a speedy start-up (< 1 week).
9. Seed material from sludge.
10. Hardly any use of chemicals.
11. Plain technology with high treatment efficiencies.
12. Reactors can be put in functional order during agricultural campaigns only (e.g. in the sugar industry, 4 months per year), anaerobic sludge can be stored unfed.
13. There is a market value of excess sludge.
14. water recycling in factories (towards closed loops) is facilitated by high rate systems.

Clearly, societal and local economic conditions determine the exact ranking of the given advantages. In the Netherlands, the excess sludge handling depends on the cost of operating wastewater treatment systems. The low sludge production is an instantaneous economic benefit in anaerobic reactors because for excess sewage sludge and biowastes, landfilling is no choice, however, prices for incineration can be about €500/ton wet sludge or more. A full-scale example is used to illustrate another important advantage of AnWT, which is system compactness, where an anaerobic reactor with 25 m height and a diameter of 6 m, is sufficient to treat up to 25 tons of COD daily. The produced sludge is packaged as seed sludge for fresh reactors and is not a useless product, in this example, it is less than 1 Ton dry matter per day. The industrial boundaries or sometimes also in the interiors of the factory buildings are the areas where the system becomes suitable for implementation because of this compactness. As the initial stage in treatment for reclaiming process water, the following is of specific interest for industries that intend to use anaerobic treatment and in heavily populated areas (Sandars et al., 2003).

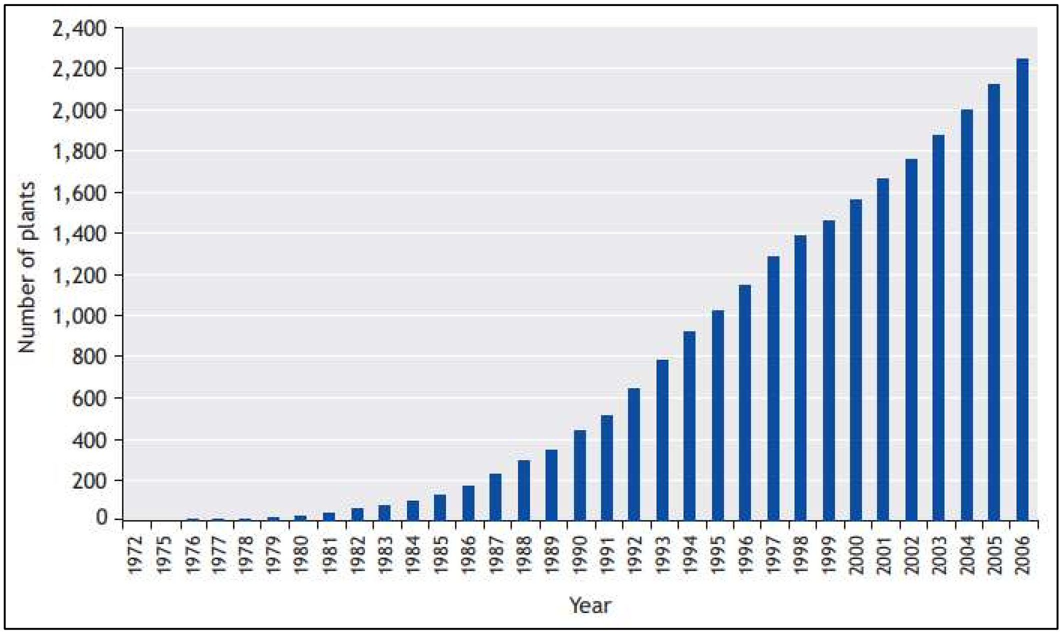

Figure 5.2. In the period of 1972-2006, increasing rate of globally installed anaerobic high-rate reactors

[Source: https://ocw.tudelft.nl/wp-content/uploads/Chapter_16_-_Anaerobic_Wastewater_Treatment.pdf]

The general interest in global warming and the ever-rising energy prices directly result in the refreshed attraction in the energy views of AnWT. With energy comparable to about 250 GJ/d, 7,000 m^3CH_4/d (assuming 80% CH_4 recovery) can result from the conversion of 25 Tons COD/d of agro-industrial waste(water). A valuable 1.2 MW electric power output can be attained with the use of modern combined heat power (CHP) gas engine, that can provide 40% efficiency. If the direct area of work or industrial fields can utilize all of the excess heat, the energy recovery as a whole could even be greater (extending up to 60%). Accepting that about 1 kWh/kgCOD taken out, or in the above case, 1 MW installed electric power, would be needed by the full aerobic treatment, over the activated sludge process, 2.2 MW is the total energy profit of using AnWT. This is equivalent to about 5,000 €/d, at an energy price of 0.1 €/kWh. Aside from the energy, generating renewable energy using AnWT can be used to get carbon credits involved by current drivers. Nearly 21 tonCO_2/d is given off by the generation of 1 MW electricity, for a typical coal-driven power plant, while it is half of this value, for a natural gas-driven plant (Collins et al,. 2003). €500/d on carbon credits (based on a coal-powered plant) can be earned by the previously discussed exampled industry, at an expected balanced price of €20/ton CO_2, however, for the treatment of the wastewater, fossil fuels are not used at all. Even though in industrialized countries, this is a very small amount, but in developing countries, a real inducement can be provided by it to secure the surrounding, by starting the treatment of wastewater using high-rate AnWT. This is why, in less developed countries, western subsidy for implementing AnWT systems is used to view the carbon credit policy (Van Lier et al., 1997; Masse et al., 2003).

5.3. Psychrophilic Anaerobic Treatment Process

For municipal wastewater, anaerobic wastewater treatment plants have not been applied, to this date, in countries with low and average temperatures, but the process is well operated in tropical countries such as Columbia, Mexico, China and India. For a one-step system, there is the requirement of long hydraulic retention time and controlled chemical oxygen demand (COD) removal to have enough hydrolysis of particulate organics at these temperatures (Vartak et al., 1997; Gasparikova, 2005). Digester failure and biogas yield decrease because of lower biochemical activity and comparatively more anaerobic bacterial populations generation time. Due to this, low-temperature results in damaging effects on anaerobic digestion (Singh et. al., 1999). In cold areas, two UASB reactors were plotted to determine the municipal wastewater start-up and treatment, they functioned at temperatures of 6, 11, 15, 20 and 32°C with many HRTs varying from 48 to 3 h (Singh and Viraraghavan, 1999). At 20°C, in approximately 281 d, the biomass aggregation (granulation) was attained. Nonetheless, in the recent past, for the range of industrial wastewater treatment, psychrophilic (< 20°C) anaerobic digestion or low temperature has been proven suitable for environmental management which shows a technological breakthrough. This is why, an appealing option to typical anaerobic digestion for wastewaters that are released at average to low temperature, is psychrophilic anaerobic treatment (Rebac et al., 1999; Smith et al., 2013).

5.3.1. Biodegradation of persistent organic compounds

As compared to mesophilic conditions, biological and chemical reactions progress very slow under psychrophilic conditions. In the biodegradation of organic matter, additional energy is needed in majority reactions to advance at low temperatures instead of temperature optimum of 37°C (mesophilic conditions) (Table 5.1). Yet less energy is demanded by some reactions, like acetate formation from hydrogen and bicarbonate, hydrogenotrophic methane production and hydrogenotrophic sulfate reduction (Table 5.1). On the highest microorganisms substrate utilization rates, a lot of researchers have observed a powerful temperature effect. In general, although the decrease in the substrate utilization rates and maximum specific growth resulted due to decreasing the operational temperature, it also has the ability to cause an increased methanogenic population or net biomass yield (g biomass g^{-1} substrate converted) of acidogenic sludge (Lettinga et. al, 2001; Christensen et al., 2009).

Fang and Zhou (1999) explored interactions of denitrifiers and methanogens, treating m-cresol (100 mg/L) and phenol (200 mg/L) containing wastewater in an up-flow anaerobic sludge blanket reactor (UASB). With hydraulic retention time (HRT) of 1 day, denitrifiers and methanogens degrade over 60 % m-cresol and 98 % phenol jointly, where the ratio of COD to NO_3-N was 5.23. Consortium seeded a continuous flow fixed-film anaerobic bioreactor and decreased the concentration of phenolic and phenol compounds by 83 and 97%, respectively, at 6 h HRT. Greater than 90% anaerobic biofilm reactors injected with combined microbial populations degraded contaminants e.g. o-cresol, phenol, debenzofuran, quinoline, phenanthrene, and acenaphthene at 3 h HRT or more (Guieysse and Mattiasson, 1999). Psychrofilic (< 20ºC) anaerobic digestion treatment or achievable low temperature of phenolic wastewater has been shown by Collins et al. (2005) Effective elimination of phenol and COD at 0.4 – 1.2 kg phenol/$m^3 \cdot d$ and 5 kgCOD/$m^3 \cdot d$ phenol and organic loading rates were noticed by them. Also, the development of the methanogenic activity under psychrophilic conditions was proved (Chaudhry & Chapalamadugu, 1991; Bajaj & Singh, 2015).

Sr. No.	Reactions	ΔG' kJ/reaction	
		37º C	10º C
1.	$4H_2 + SO_4^{2-} + H^+ \rightarrow HS^- + 4H_2O$	-148.2	-157.1
2.	$CH_3CH_2CH_2COO^- + 2.5SO_4^{2-} \rightarrow 4HCO_3^- + 2.5HS^- + 0.5H^+$	-128.3	-116.4
3.	$CH_3CH_2COO^- + 3H_2O \rightarrow CH_3COO^- + HCO_3^- + H^+ + 3H_2$	+71.8	+82.4
4.	$CH_3CH_2CH_2COO^- + 2H_2O \rightarrow 2CH_3COO^- + H^+ + 2H_2$	+44.8	+52.7
5.	$CH_3COO^- + H_2O \rightarrow CH_4 + HCO_3$	-32.5	-29.2
6.	$CH_3CH_2COO^- + 1.75SO_4^{2-} \rightarrow 3HCO_3^- + 1.75HS^- + 0.25H^+$	-88.9	-80.7
7.	$4H_2 + 2HCO_3^- + H^+ \rightarrow CH_3COO^- + 4H_2O$	-98.7	-111.8
8.	$CH_3CH_2CH_2COO^- + 0.5SO_4^{2-} \rightarrow 2CH_3COO^- + 0.5HS^- + 0.5H^+$	-29.3	-25.9
9.	$CH_3COO^- + SO_4^{2-} \rightarrow 2HCO_3^- + HS$	-49.5	-45.3
10.	$4H_2 + HCO_3^- + H^+ \rightarrow CH_4 + 3H_2O$	-131.3	-140.0
11.	$CH_3CH_2COO^- + 0.75SO_4^{2-} \rightarrow CH_3COO^- + HCO_3^- + 0.75HS^- + 0.25H^+$	-39.4	-35.4

Table 5.1. In the absence and presence of sulphate, Gibbs and Stoichiometry free-energy changes of butyrate, propionate, acetate, and hydrogen anaerobic conversion (Lettinga et. al, 2001).

While many dyes are uncontrollable to biodegradation, decolorisation of textile effluents has usually failed in the anaerobic biological process. However, the conceivable usefulness of anaerobic reductive cleavage by microbes was presented. For instance, for microbial dyes structure cleavage, anaerobic digeste sludge was applied by Donlon et al. (1997) and Cariel et al. (1995). There is a production of cancerogenic aromatic amine intermediates, under anaerobic conditions, as a result of azo-reactive dyes destruction.

For developing less injurious end products, they can degrade more under aerobic biological treatment. Tan et al. (1999) judged biodegradation of two azo dyes, Mordant Yellow-10 and 4-phenylazophenol, in batch systems whereby reacting anaerobic granular sludge to oxygen, the aerobic and anaerobic conditions were merged. Temporal accretion of aromatic amines results from the reduction of the azo dyes. The efficiency of an anaerobic-aerobic system for real textile wastewater in admixture with textile outflow has been proved by own investigations (Mrowiec and Suschka, 2006). A colour reduction of 75 % and COD reduction of 79 % was attainable only at the first anaerobic stage that contains a fixed bed filter, operating at 7.7 h HRT (Hirano et al., 2007).

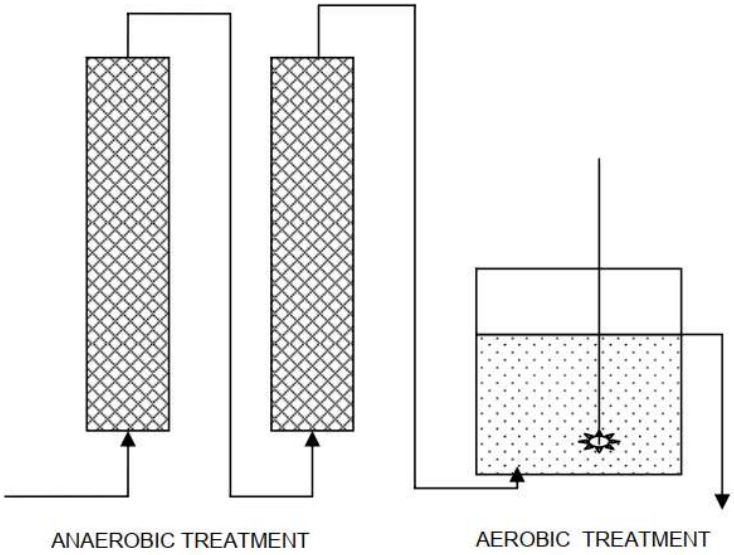

Figure 5.3. Diagrammatic portrayal of anaerobic-aerobic textile wastewater treatment.

[Source: https://www.kth.se/polopoly_fs/1.650925.1550157132!/JPSU16p13.pdf]

Figure 5.4. Anaerobic-aerobic textile wastewater treatment's laboratory system.

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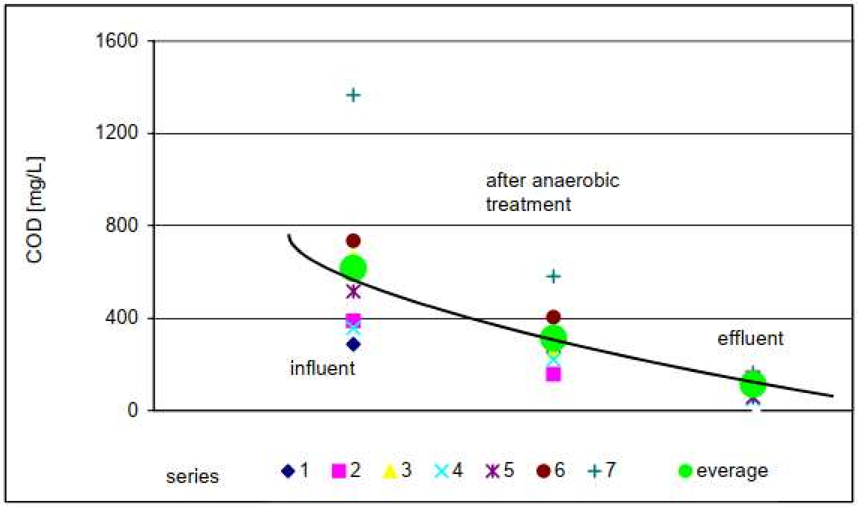

Figure 5.5. Elimination of COD in the anaerobic-aerobic textile wastewater treatment process.

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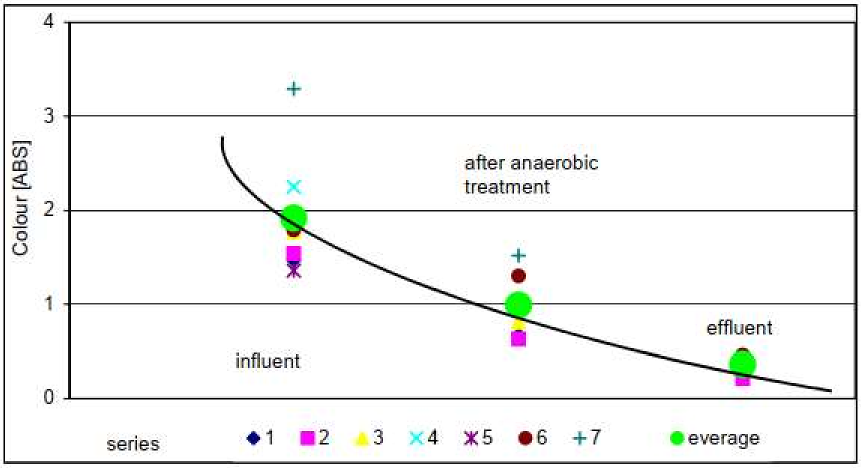

Figure 5.6. eradication of colour in the anaerobic-aerobic textile wastewater treatment process.

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Kleerebezem et al. (1999) studied the impacts of benzoate and acetate by a syntrophic methanogenic culture on anaerobic degradation of terephthalate (1,4-benzenedicarboxylate). The absolute dependence of the formation of benzoate resulting from decarboxylation of terephthalate on the concomitant fermentation of benzoate was seen by them (Pereira et al., 2005; Bandara et al., 2011).

Van Eekert et al. (1999) proved the significant mechanisms for the transformation of chlorinated ethanes, in a UASB reactor by the methanogenic consortium, to be biotic and cometabolic conversions. Navarrete et al. (1999) determined the effect on a UASB reactor fed with synthetic wastewater by 1,1,2,2-tetrachloroethane (TCE). The results proved that industrial wastewaters containing TCE can be processed by the UASB (Petala et al.,2006; Lee et al., 2013).

Vanderloop et al. (1999) used Anaerobic/anoxic fluidized-bed granular activated carbon (GAC) bioreactor, for treatment of 2,4-dinitrotoluene (DNT), in series with activated sludge reactor. In the anaerobic stage, 2,4-diaminotoluene (DAT) concludes from the conversion of DNT. 5 % of DAT was converted to 4-amino-2-nitrotoluene, 2-amino-4-nitrotoluene, about 45 % remained as DNT and other to unknown products, under denitrification conditions in the fluidized bed. Razo-Flores et al. (1999) considered the consequence of four nitroaromatic compounds (4-nitrobenzoate, 5-nitrosalicylate, nitrobenzene, 2,4-dinitrotoluene) in laboratory-scale UASB reactor. Stoichiometrically, the corresponding aromatic amines were produced from all nitroaromatic compounds.

5.3.2. Effect of low temperature on properties of wastewater

The design and operation of the treatment system can be largely affected by a fall in temperature, followed by a change of the chemical and physical properties of the wastewater. For example, with decreasing temperature below 20°C, the solubility of gaseous compounds increases (Choi et al., 1998; Yang et al., 2015). This follows that as compared to dissolved concentrations of hydrogen sulfide, methane and hydrogen from reactors operated at high temperatures, they will be higher in the discharge of reactors operated at low temperatures. A little lower reactor pH might succeed under psychrophilic conditions, this is implied by the high increase of solubility of CO_2 (Hossain et al., 2010; 2011).

The viscosity of liquids, at low temperatures, are raised as well. Due to this, specifically at low biogas production rates, sludge bed reactors are not easily mixed and more energy is needed for it. At low temperatures, due to a reduced solid-liquid separation in psychrophilic reactors particles will settle slower. Furthermore, corresponding to higher liquid viscosity, soluble compounds diffusion will decrease at fewer temperatures. In comparison to the mesophilic temperature range (30–40°C), the diffusion constant of soluble compounds at 10°C is about 50% lower (Langenhoff & Stuckey, 2000; Lettinga et. al., 2001).