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Research article

Biodegradation of bilge water: Batch test under anaerobic and aerobic conditions and performance of three pilot aerobic Moving Bed Biofilm Reactors (MBBRs) at different filling fractions

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ABSTRACT

The bilge water that is stored at the bottom of the ships is saline and greasy wastewater with a high Chemical Oxygen Demand (COD) fluctuations (2–12 g COD L^{-1}). The aim of this study was to examine at a laboratory scale the biodegradation of bilge water using first anaerobic granular sludge followed by aerobic microbial consortium (consisted of 5 strains) and vice versa and then based on this to implement a pilot scale study. Batch results showed that granular sludge and aerobic consortium can remove up to 28% of COD in 13 days and 65% of COD removal in 4 days, respectively. The post treatment of anaerobic and aerobic effluent with aerobic consortium and granular sludge resulted in further 35% and 5% COD removal, respectively. The addition of glycine betaine or nitrates to the aerobic consortium did not enhance significantly its ability to remove COD from bilge water. The aerobic microbial consortium was inoculated in 3 pilot (200 L) Moving Bed Biofilm Reactors (MBBRs) under filling fractions of 10%, 20% and 40% and treated real bilge water for 165 days under 36 h HRT. The MBBR with a filling fraction of 40% resulted in the highest COD decrease (60%) compared to the operation of the MBBRs with a filling fraction of 10% and 20%. GC-MS analysis on 165 day pointed out the main organic compounds presence in the influent and in the MBBR (10% filling fraction) effluent.

1. Introduction

Bilge waters that are stored at the bottom of the ships are greasy wastewaters that include lubricating oil, cleaning diesel oil, oily sludge, spills from the engine room, water leaks from internal pipes, and seawater filtrations (McLaughlin et al., 2014). The chemical composition of bilge water varies both between vessels and also from day to day within a vessel (Tiselius and Magnusson, 2017), while high COD concentrations (2 g L^{-1}) have been reported in the literature (Ulucan and Kurt, 2015). Additionally, the salinity of bilge water ranges between 25 and 35 g L^{-1}, hindering its biological treatment. In a recent review, Vyrides and Stuckey (2017) proposed the addition of low concentration of compatible solutes such as glycine betaine to biological system as a strategy to alleviate salinity inhibition to microorganisms. Therefore, this approach was tested at batch experiment during this study.

The discharge of oil residue to marine environments is prohibited according to the International Maritime Organization (IMO) regulations (MARPOL 73/78) and the European directive 2000/59/EC. To meet the IMO regulations, the bilge water is either treated enroute in an oil separation system before being discharged to the sea or deposited at reception facilities on land. McLaughlin et al. (2014) showed that oily water separators mounted on three container and bulk carrier's significantly reduced most substances for which there are regulated concentration limits. However, Tiselius and Magnusson (2017) pointed out that discharges of treated bilge water from oil separation system would be toxic to the zooplankton and the microbial community in the ambient water for the majority of the passenger ferries. Chemical or physical processes such as flotation, separation by centrifuge, filtration, and coagulation are the most common methods for the treatment of bilge water. Due to the fact that the major part of the oil in bilge water is emulsified, the physical methods may fail to satisfy the

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targeted treatment levels. Caplan et al. (2000) stated that emulsified oil droplets smaller than 20 μm cannot be separated by means of conventional oil/water separation systems. Noteworthy physicochemical methods contribute substantially to operational cost and therefore other alternatives are of great demand for companies dealing with bilge water treatment (McLaughlin et al., 2014).

On the other hand, few studies are so far available for the use of biological methods for real bilge water treatment. Regarding the anaerobic treatment of bilge water, only one study is available and the used bilge water was highly diluted (initial concentrations: 20–200 mg COD L^{-1}). Specifically, Emadian et al. (2015) reported 75% COD removal of low strength 0.6 g COD $L^{-1}day^{-1}$ bilge water, while no information was given for methane generation. Up to date there are no published studies regarding the anaerobic treatment of undiluted bilge water.

Regarding aerobic biological treatment methods, Sun et al. (2010) proposed a compact onboard integrated wastewater treatment system for all the wastewater streams on ships, including grey water, black water and bilge water, using aerobic biofilm-MBR technology. Both dead-end side-stream and recycle side-stream configurations of the biofilm-MBR concept were investigated, achieving a good membrane permeate quality in each process configuration, with oil concentrations <5 mg L^{-1}. However, the membrane was seriously fouled when the dead-end side-stream configuration was operated with a high filtration unit recovery (93%). The membrane fouling rate was found to be directly related to oil concentration and oily biomass characteristics in the membrane filtration unit. Mancini et al. (2012) in a pilot biofilm membrane bioreactor, inoculated with halophilic activated sludge and *Alcanivorax borkumensis*, found 72–90% COD removal from low strength bilge water (1 g COD/L) during 30 days.

The moving-bed biofilm reactor (MBBR) is a growing biofilm system which has gained much interest in the wastewater treatment sector in the last 20 years (Barwal and Chaudhary, 2014). It is based on the use of freely moving plastic carrier elements with density a little lighter than that of water in which microorganisms form biofilms (McQuarrie and Boltz, 2011). Previous study have shown the higher ability of attached biomass developed onto biocarriers to biodegrade organic synthetic compounds comparing to the activated sludge process (Mazioti et al., 2015); while MBBRs have so far applied successfully for the treatment of several types of industrial wastewater (Schneider et al., 2011; Hassani et al., 2014). Despite the effectiveness of this type of MBBR, to the authors' knowledge, no studies have investigated the treatment of real bilge water using MBBR. Apart from this, up to date, only few studies investigated the effect of filling ratios on MBBR performance (Gu et al., 2014; Zhang et al., 2016). The biofilm on carriers exist in a dynamic process whereas biofilm attachment and detachment simultaneously occur (Gu et al., 2014). The carrier filling ratio can influence this as more available carriers can lead to more available sides for bacteria to create biofilm on carriers. However, the increase in carriers could increase particle–particle collision which could cause the detachment of biofilm from the carriers (Gu et al., 2014). Gu et al., 2014 tested the effect of carrier filling ratio ranging from 20% to 60% to a MBBR (8L) that treated coking wastewater. The maximum performance was found at 50% carrier filling ratio at 20 h hydraulic retention time. In another study, Zhang et al. (2016) investigated the effect of different filling ratios (10%, 20% and 30%) on the performance of MBBR. Specifically, Zhang et al. (2016) used synthetic domestic wastewater and found no significant difference in the lab-scale MBBRs performance due to the difference filling fractions. However, no study has examined the effect of filling fractions in MBBR treating industrial wastewater at pilot scale.

Based on the above, the aim of this study was to examine the biodegradation of undiluted real bilge water using anaerobic granular sludge followed by aerobic microbial consortium and vice versa. The effect of addition of glycine betaine (GB) as well as nitrate was investigated as a strategy to increase methane generation and COD removal both on anaerobic granular sludge and to aerobic microbial consortium, respectively. Then, based on these preliminary batch results, the biological treatment of bilge water in three pilot (200 L) MBBRs, operated with different filling ratios (10%, 20% and 40%), was tested for 165 days. For the first time, GC-MS analysis was conducted in bilge water samples to identify the organic compounds prior and after MBBR treatment.

2. Material and methods

2.1. Wastewater characteristics

The bilge water was provided from a company (Ecofuel Ltd) which collects and treats this type of wastewater at Zygi, Cyprus. The characteristics of pre-settled bilge water were as follows: pH: 7.5–8.5, COD: 2.9–12.8 g COD L^{-1}, Total Solids (TS): 800–1200 mg L^{-1}, conductivity: 38.1 mS/cm, Total Phosphorus (TP): 60 mg L^{-1} and Total Nitrogen (TN): 120 mg L^{-1}.

2.2. Biochemical methane potential (BMP) test

A modified biochemical methane potential (BMP) test (Owen et al., 1979) was conducted in 250 ml glass serum flasks with a working volume of 150 mL for experiments with bilge water and for the treatment of aerobic effluent. The experiments with addition of glycine betaine and nitrate took place at 125 ml serum bottle with a working volume 60 ml.

The Na increase due to $NaNO_3$ cannot inhibit anaerobic biomass as it corresponds to 0.05 g L^{-1}. Vyrides and Stuckey 2009b was shown that Na start to inhibit anaerobic biomass after 10 g L^{-1} and that the absence of nitrate can decrease the performance of anaerobic biomass.

During batch experiments, 20% of the working volume consisted of anaerobic granular sludge (VSS 7.5 g L^{-1}, withdrawn from a full scale UASB that treats cheese whey) and the rest was filled up with undiluted bilge water. Apart from the addition of 1.3 g $NaHCO_3$ L^{-1}, no other nutrients or trace elements were added. The serum bottles were placed in a shaking incubator (Stuart SI500) at 37 °C and 100 rpm for 13 days. These experiments were contacted in duplicates or in triplicates.

2.3. Aerobic treatment of bilge water with the use of microbial consortium

The aerobic consortium consisted of *Pseudomonas aeruginosa LVD-10* (Drakou et al., 2015) *Enterobacter* sp. *SW* (Drakou et al., 2017*), Citrobacter* sp. *D2, Citrobacter* sp. *S1 and Citrobacter* sp. *S6.* The *Citrobacter* species were previously isolated and identified from soil that was contaminated by oil at the Environmental Engineering Laboratory at the Cyprus University of Technology. Subcultures of the aforementioned strains were pre-grown overnight at 30 °C in medium salt medium (MSM) supplemented with 0.5 g $L^{-1}$of phenol and 0.5 g L^{-1} glucose as a carbon source. Chemicals for MSM preparation were supplied by Sigma-Aldrich (USA). Cultures were prepared and the incubation was performed using serum bottles with 100 ml total volume (30 ml working volume) in a shaking incubator (Stuart SI500) stirred at 100 rpm. Then 2 ml of each strain (O.D. 0.4) was added to a conical flask with 190 ml of bilge water (500 ml total volume and 200 ml working volume) in a shaking incubator (Stuart SI500) which was stirred at 100 rpm at (30 °C) and the pH was adjusted to pH 7. This experiment was done in