

Biofoam formation and defoamation in global wastewater treatment systems

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Abstract

Municipal wastewater treatment is largely based on activated sludge (AS) systems due to their ability to reduce biological and chemical oxygen demand (BOD/COD). They are similarly efficient in nitrification and denitrification. However, major drawbacks such as foaming associated with the prevalence of lipids (fats, oil, grease (FOG)) and proteinaceous material arise, which reduces AS efficiency – a focus of this review. Many strategies are employed for foam reduction in AS systems, where proliferation of foam-forming microorganisms can be challenging. To understand foam formation, prevention and deterioration, including destabilisation, a multidisciplinary mitigation approach is required, in which some bioprocess aspects such as foam destabilisation kinetics should be understood and quantified. This review reports on biological foam formation and source in wastewater treatment, defoaming strategies, and biofoam destabilisation kinetics as well as factors affecting foam stability.

Key words: activated sludge systems, biodefoamers, chemical defoamers, foam formation, wastewater treatment

Highlights

- Important sources of biological foam formation.
- Comprehensive foam formation, foam prevention and destabilisation.
- Quantified foam destabilisation kinetics.
- Defoamation strategies in wastewater treatment.
- Improved activated sludge systems in wastewater treatment

INTRODUCTION

Activated sludge (AS) systems are commonly used in wastewater treatment systems. They are efficient in remove organic matter in the form of dissolved solids, nutrients and dissolved biodegradable carbon (Pal *et al.* 2014). Wastewater treatment plants (WWTPs) are common worldwide. A conventional municipal WWTP has a primary clarifier, aeration tank/basin and secondary tank/clarifier including a recycle stream that recycles sludge from the secondary clarifier to the aeration basin (Tchobanoglous *et al.* 2003). Mixed liquor suspended solids (MLSS) in the aeration basin contains suspended microbial growth responsible for aerobic nitrification and subsequent anoxic denitrification as

well as phosphorus removal (Saunders *et al.* 2015). The microorganisms in the AS aeration basin feed continuously on the nutrients in the influent.

The primary purpose of AS is to degrade organic matter and pollutants in the wastewater so that the treated wastewater meets the discharge standards (Islam *et al.* 2013). Liquid-solid separation occurs in the clarifiers. In the secondary clarifier, after the primary AS system, the solids (sludge) are returned to the aeration basin, while the clarified liquid goes to tertiary treatment systems. AS systems can be classified as primary bioengineered system designed to mimic natural processes; although, wastewater treatment is done rapidly such that the wastewater throughput is sustained for high volume treatment plant efficiency.

A major challenge in AS systems is sludge bulking and foaming (Xia *et al.* 2018). Sludge bulking is considered a challenge due its ability to render WWTPs inoperable, unlike foam formation. Foaming is generally neither studied nor monitored sufficiently and usually arise from the presence of surfactants, proteins, fats, oil and grease (FOG), as well as the presence of filamentous bacteria such as *Gordonia amarae* (*G. amarae*) (previously known as *Nocardia amarae* (*N. amarae*)), *Microthrix parvicella* (*M. parvicella*), Eikelboom types 021N, 1701, 1863, 0041/0675, 0092, 0803, 0914, 1851 and other actinomycetes with cell wall containing filaments (Wanner *et al.* 1998). These microorganisms are highly hydrophobic thus readily accumulate in organic matter-polluted wastewater, particularly in the aeration basin where they are buoyant because they easily attach to air bubbles and float to the wastewater surface, proliferating rapidly due to nutrient availability. Subsequently, they release extracellular polymeric substances (EPSs) some of which are biosurfactants, to facilitate the hydrolyzation of organic pollutants including FOG into long chain fatty acids (LCFAs), which then form their primary nutrient and carbon source (Lemmer *et al.* 2000; Xia *et al.* 2018). When the concentration of FOG is high, foam formation due to actinomycetes proliferation is inevitable and biofoam defoamation becomes crucial.

Various studies have been done outlining aspects of chemical, physical and biological biofoam control (Pitt & Jenkins 1990; Mamais *et al.* 2011; Khairnar *et al.* 2014; Shao & Kao 2014). The current review provides an insight into different approaches, including an eco-friendly and environmentally benign approach. The review's objective was to highlight biofoamation as a major hindrance for AS systems, and to consolidate information on the causes of foaming in WWTPs, including defoaming methods and strategies, and highlight the efficacy of biodefoamers.

WHAT CAUSES FOAM IN WWTPS?

Sludge bulking and foaming arise from the overgrowth of filamentous bacteria in AS systems. These are distinct and they fall under the mycolata taxon (Wanner 1994). Madoni *et al.* (2000) carried out a survey of biological bulking and foaming agents in 167 Italian WWTPs. They found abundant *M. parvicella* in sludge bulking and foaming, whereas Norcardioform actinomycetes, Eikelboom types 0041, 0092, 021N, 0675 and Thiothrix were only detected in very low concentrations. Guo & Zhang (2012) also used throughput sequencing to profile dominant microorganisms associated with biobulking and biofoaming in 14 WWTPs world-wide. In their study, *M. parvicella*, *Nostocoida limicola I* and *II*, *Mycobacterium fortuitum* and Eikelboom type 1863 were identified as dominant sludge biobulkers and biofoamers. Overall, foaming occurs in AS systems largely because of aeration in the presence of synthetic and biosurfactants from filamentous bacteria (Mulligan 2005; Nielsen *et al.* 2009; Osama Al 2014). It is also associated with the presence of proteins and FOG including foam facilitating operation conditions at WWTPs (Rossetti *et al.* 2005). Generally, the presence of biosurfactants in combination with proteins produces a stable foam, with the proteins, composed of nitrogen, carbon, hydrogen and oxygen, being used as carbon and nitrogen sources by the foam-forming filamentous bacteria (Willey *et al.* 2008). These foam-forming bacteria can also use soluble hexadecane and acetate as readily available carbon sources during their growth. Under such

conditions, the organisms will excrete biosurfactants. Microorganisms such as *M. parvicella* and *N. amarae* proliferate in FOG-containing wastewater, as they degrade the lipids and solubilise them by producing biosurfactants (Pal *et al.* 2014; Dunkel *et al.* 2018).

N. amarae is a gram-positive non-motile organism that depends on the wastewater characteristics and sparging to float to the wastewater surface. It has relatively low food requirements and depends on the effluent for a continuous supply of biodegradable FOG for growth (Tandoi *et al.* 2017). *M. parvicella*, however, is gram-positive and slow growing, and has unbranched filamentous strands that appear between flocs, disrupting floc formation. Like *N. amarae*, *M. parvicella* has a low food to microorganism ratio (f/m). It is also microaerophilic, and can grow in any AS zone, particularly dead zones. The optimum pH range for *M. parvicella* is 6.7–8 and is likely to be limited by a high dissolved oxygen (DO) concentration (Rossetti *et al.* 2005). It also has a hydrophobic cell surface containing filamentous strands that attaches to foam layers. The filamentous network ensures entrapment in bubbles, resulting in a buoyant movement to the wastewater surface. *M. parvicella* cannot use Acetyl Coenzyme A to generate LCFAs for growth but depends on the FOG in the influent or sludge (Tandoi *et al.* 2017). Table 1 outlines the different environmental conditions that promote the growth of various filamentous microorganisms associated with biofoam development.

Table 1 | Environmental conditions promoting different types of filamentous organisms associated with biofoaming (Nielsen *et al.* 2009; Liu *et al.* 2018; The Water Network 2019)

Environmental condition	Organism
High DO concentration	<i>Sphaerotilus natans</i> , Eikelboom type 021N
Low f/m ratio	Eikelboom types: 0041, 0675, 1851
Grease and oil availability	<i>Nocardia</i> sp., <i>Microthrix parvicella</i> , Eikelboom type 1863
Low pH	Fungi
Protein availability	Chloroflexi, Eikelboom types 0092 and 0041

MODERN MOLECULAR METHODS FOR IDENTIFICATION OF BIOFOAM PRODUCERS

Conventionally, filamentous bacteria were characterised based on their morphology using staining and light microscopic techniques (Eikelboom 1975). However, it was shown that filament characterisation and identification using microscopic techniques was inaccurate for the identification of filamentous bacteria. Therefore, new methods like fluorescence *in-situ* hybridisation (FISH) were developed in order to identify biofoamers (Nielsen *et al.* 2009). Polymerase chain reaction (PCR) is another molecular method that is used to amplify specific DNA sequences for microorganism identification. Generally, real-time quantitative PCR (qPCR) detects a targeted DNA sequence for a certain specie during PCR (Nittami *et al.* 2017). Other newly available techniques include 16S rRNA sequencing on the Illumina platform, which uses a reversible dye terminator method (Dunkel *et al.* 2018). Metagenomics of *in-situ* mixed cultures and the MiDAS 2.1 database were also designed to profile AS bacteria (Speirs *et al.* 2019).

CONTROL METHODS FOR EXCESSIVE FOAM IN WWTPS

Various methods have been used to control excessive foaming in AS systems but it persists. Some foam control methods were successful in laboratory studies but, when scaled up, were costly, and after a time, reduced AS efficiency (Mamais *et al.* 2011). Foam reduction or complete removal is complex and a sustainable solution requires more in-depth research. Methods reported in previous studies (Table 2) included the adjustment of AS operating conditions, water spraying, steaming, use

Table 2 | Biofoamers and general AS control strategies in various countries (by country)

Country	Biofoamers	Control strategies	References
Argentina	<i>S. natans</i> , <i>M. parvicella</i> , <i>Nocardia</i> sp. <i>Thiothrix</i> I, Eikelboom Types: 021N, 1701, 1863, 0041, 0092	Chlorination, oxidant or coagulant addition	Di Marzio (2002)
Australia	<i>M. parvicella</i> , Eikelboom Types 0041/0675, 0092 and <i>H. hydrossis</i>	Chlorination, aeration and f/m ratio 0.4 manipulation and reducing sludge age	Seviour <i>et al.</i> (1989), Seviour <i>et al.</i> (1994), Seviour <i>et al.</i> (1997)
Czech Republic	<i>M. parvicella</i> , Eikelboom Type 0092	FEX-120 supplementation, reducing sludge, water sprays, mechanical skimming, use of contact zone selectors, chlorination	Krhotkova <i>et al.</i> (2002), Kragelund <i>et al.</i> (2010)
Denmark	<i>M. parvicella</i> , Eikelboom Types: 0041/0675, 021N, 0092, 0803, 0914, 1851 and <i>N. limicola</i>	FEX-120, chlorination, addition of PAX-14, use of selectors	Kristensen <i>et al.</i> (1994), Kragelund <i>et al.</i> (2010)
France	<i>M. parvicella</i> , Nocardioform Eikelboom Types: 0041/0675, 021N, Type 0092 and <i>M. limicola</i>	Chlorination, contact zone loading rate manipulation	Pujol (1994)
Germany	<i>M. parvicella</i> , Eikelboom Types: 0041/0675, 021N, 0092 and 1701	Coagulants (FeCl ₃ , Ca(OH) ₂), oxidants (H ₂ O ₂ , Cl ₂), antifoam and coagulant use; contact zones use, reduction of sludge age, DO reduction and mechanical skimming	Kunst & Reins (1994)
Italy	<i>M. parvicella</i> , NALO, Eikelboom Types: 0041/0675 and 0092	Sludge retention time (SRT) reduction, maintaining DO at >2 mg/L. Influent delipidation, chlorine and PAX-14 usage.	Rossetti <i>et al.</i> (1994), Madoni <i>et al.</i> (2000), Rossetti <i>et al.</i> (2005)
Japan	Eikelboom Types: 021N, 1701, 0041/0675 <i>S. natans</i> and <i>Thiothrix</i> sp.	DO reduction, use of coagulants and feast-fast technique	Mino (1995), Chua <i>et al.</i> (2000)
Netherlands	<i>M. parvicella</i> , <i>H. hydrossis</i> , Eikelboom Types: 021N, 0041/0675, 0092 and 701	Decreasing f/m ratio and SRT. Delipidation and polymers addition with DO at >2 mg/L	Eikelboom (1977), Eikelboom (1994), Kruit <i>et al.</i> (2002), Mino (1995)
South Africa	<i>M. parvicella</i> , Eikelboom Types: 1851, 0041/0675 and 0914	Chlorine, O ₃ , H ₂ O ₂ and polymer addition. Use of a defoamer, DO adjustment, sludge age reduction, physical skimming	Blackbeard <i>et al.</i> (1986), Blackbeard <i>et al.</i> (1988), Mangundu (2017), Pitman (1996)
United Kingdom	<i>M. parvicella</i> , Eikelboom Type 021N, <i>N. limicola</i> , and NALO	Manipulation of DO and f/m, and reduced MLSS at 2.5 g/L	Foot (1992), Foot & Robinson (2003)
United States of America	<i>M. parvicella</i> , NALO, Eikelboom Types: 1701, 021N, 0092, 0041/0675 and NALO	Addition of coagulant	Strom & Jenkins (1984), Switzenbaum <i>et al.</i> (1992), Jenkins <i>et al.</i> (1993)

of 'feast-fast' reactors in series and the use of chemical defoamers (Hoyle *et al.* 2006; Tsang *et al.* 2008; Shao & Kao 2014).

Non-specific foam control strategies

Adjustment of AS operating conditions

The operating conditions used to control foam formation include the mean biomass (cell) residence time (MCRT) within the AS system; that is, the average residence time (days) of the microorganisms within the AS system. Pitt & Jenkins (1990) reported that lowering the MCRT reduced *N. amarae*

growth in 6 days. On this basis, reducing MCRT to less than 8 days was deemed as an appropriate strategy for managing *M. parvicella* proliferation (Pal *et al.* 2014). However, it was inefficient for removing excessive foam in industrial-scale operations, whereby nitrification is also required. Generally, many other foam producers are present in excess in AS systems and can survive MCRT reduction. Richard (1989) reduced air in the aeration basin so that the bubbles in it were significantly minimised, thus reducing excessive foaming. This led to incomplete nitrification as well as an excess of suspended solids in the secondary clarifier.

Application of water sprays to control excessive foam

Water spraying has also been used to control excessive foaming in AS systems. Water is sprayed on to the foam surface so that the bubbles collapse rapidly. The sprays can be suspended in the AS system. Using sprays does not minimise the foam former proliferation but the filamentous bacteria are returned to the AS system and integrated into the MLSS. This method tends to assist in the reintegration of filamentous organisms, some of which will be embedded in the AS solids and recycle back via the secondary clarifier, promoting bulking or the generation of more foam, which will require more spraying, subsequently increasing the plant's operating costs (Saby *et al.* 2002).

Application of steam to reduce filamentous bacteria

Hoyle *et al.* (2006) conducted a study whereby steam was applied to reduce foam-causing filamentous bacteria. The steam was delivered to a reactor containing a 0.5 ℓ foam sample. Various pressure and time combinations were tested from 207 to 483 kPa and 10 to 60 minutes, respectively. *Nocardia* sp. filament growth decreased gradually at 483 kPa for 60 minutes' exposure but this did not affect *M. parvicella* growth. As a result, foam forming potential was unaffected although foam stability was reduced. Although the study demonstrated a reduction of *Nocardia* sp. filaments, the system is energy intensive and other microbial structures might evolve to perform differently, with temperature sensitive microorganism cells lysing.

'Feast-fast' operation

Tsang *et al.* (2008); Chua *et al.* (2000) used a 'feast-fast' process, whereby floc formers were separated from foam formers by harnessing their growth conditions in separate reactors prior to the passing of wastewater containing LCFAs and FOG. The two reactors, operated in series, were named feast (reactor 1) and fast (reactor 2). The feast reactor influent favoured floc former growth because it contained a high f/m ratio of 0.75 BOD/gMLSS/d, while the fast flask contained a low f/m ratio that favoured the growth of filamentous foam formers (Tsang *et al.* 2008). Since floc and foam formers were harnessed in different growth reactors, there was minimal competition for the nutrients. This technique reduced the sludge volume index from 300 to 80 mL/g, as well as foam stability in the system. Some 95% of BOD was removed and *N. amarae* growth decreased, while the system's stability improved. Although the results were good at laboratory scale, operation at full-scale operation is impractical.

Chemical dosing to reduce excessive foaming in WWTPs

Chemicals – for example, chlorine and chemical polymers or coagulants, have also been used to reduce excessive foaming. They are a short-term solution because consistent dosing is required for foam reduction. Dosing chlorine into the aeration tank can also be inefficient as it causes floc disintegration and can lead to sludge bulking, which renders the AS system inefficient

(Eckenfelder & Grau 1998). Chlorine also breaks microorganism cell walls and affects their metabolism. Other than that, when chlorine is added to organic matter, it may form chlorinated oxidation by-products like trihalomethanes (Pasinetti *et al.* 2005) that might cause further damage to the microbial population in AS systems (Liu 2003; Dai *et al.* 2013) while raising the soluble COD concentration in the treated wastewater (Cotruvo & Amato 2019).

Mamais *et al.* (2011) investigated the addition of coagulants such as ferric chloride, ferrous chloride, polyaluminium chloride, hydrated aluminium sulphate, and cationic polymers. In the study, polyaluminium chloride with a cationic polymer was determined as the most effective foam reduction combination. Adding polyaluminium chloride at 31.5 mg/L improved sludge settleability while good foam reduction was achieved with 0.6 mg/L of the cationic polymer. Microscopic analyses showed that polyaluminium chloride improves floc density and enhances floc formation. The study also showed *M. parvicella* and *G. amarae* (*N. amarae*) filaments embedded within the flocs, and thus deprived of a nutrient-rich environment (Nielsen *et al.* 2005). At this polyaluminium chloride dose, biofoam reduction was between 75 and 100% (Pal *et al.* 2014).

Although these coagulants achieved significant results, they may not be suitable for use in full-scale WWTPs. Most wastewater discharge regulations and/or standard guidelines specify that the treated water discharged must contain 0.25 mg/L or less total chloride, so using chemicals with a significant chloride content – for example, chloride-based coagulants – will not be ideal.

Inhibition of filamentous bacterial growth by deselection mechanism

Biofoam formation and sludge bulking can be inhibited by a deselection mechanism, whereby conditions are made unfavourable for filamentous growth but favour floc formers. Several filamentous foam forming bacteria occur in AS including alphaproteobacteria, bacterioidetes, and so on (Nielsen *et al.* 2009), with alphaproteobacteria taking up most of the nutrients and subsequently storing polyhydroxyalkanoate, which facilitates sludge bulking. These types of organisms can be deselected by adding a dehydrogenation treatment stage or creating a selective reactor configuration for their removal (Kragelund *et al.* 2005). Apart from *M. parvicella*, bacterioidetes such as *Curvibacter* and TM7 cause neither bulking nor foaming (Kragelund *et al.* 2008). Additionally, *M. parvicella* can be controlled by removing FOG to degrade surface active lipases. All these filamentous, foam-forming bacteria can be controlled selectively using a deselection mechanism whereby anoxic and anaerobic selectors can control their growth, assisted by chemical dosing.

Specific foam control methods: future perspectives

Application of microbial cells as defoamers

Chemical and physical methods are short-term solutions to excessive foaming in AS systems, and novel approaches are required to overcome their limitations (de los Reyes 2010). The novel biological approaches studied include the use of bacteriophages to reduce foam formers. Phages are specific to individual microorganism types/species. Withey *et al.* (2005) showed that bacteriophages could reduce mycolata cell numbers sufficiently for foam formation to be reduced significantly. Petrovski *et al.* (2011a) used multi-spectrum DNA phages, i.e. GTE2 and GTE7, isolated from an AS system. They were screened for their ability to lyse 65 different mycolata species. In their study, GTE7 lysed all the 65 mycolata species while GTE2 was lytic against only five (Petrovski *et al.* 2011a, 2011b).

Khairnar *et al.* (2014) isolated and characterised three bacteriophages NOC1, NOC2 and NOC3 to lyse *Nocardia* sp., which they did efficiently under laboratory conditions (at 30 °C for 2 days). Pajdak-Stós *et al.* (2017) conducted further, full-scale studies for reducing foam-forming microorganism using

rotifers, which ingest and minimise filaments in sludge, thus reducing excessive foam. Three rotifers – isolates *Lecane tenuiseta*, *Lecane inermis*, and *Lecane pyriformis* – were used successfully on *M. parvicella* and Eikelboom type 0092 filaments within two weeks. In general, the rotifers reduced some actinomycetes populations significantly. *Lecane inermis* was then introduced into a full-scale WWTP, which was monitored for a year. It was further discovered that the introduction of rotifers reduced *M. parvicella* and other actinomycetes.

Antimicrobial characteristics of biodefoamers (microbial defoamers)

Antimicrobial defoamers can either lyse or inhibit the growth of the filamentous bacteria present in AS systems. If they can lyse bacterial cell walls, they are called bactericidal, but if they inhibit foam formers growth they are bacteriostatic. [Khairnar et al. \(2014\)](#) used bactericidal phages to eliminate the growth of filamentous bacteria. Microbial cells can also be used as antibacterial agents to lyse other cells or to inhibit the growth of other microorganisms. The extra- and intra- cellular structures of compatible microorganisms can be used as defoamers to inhibit or lyse filamentous bacteria known for foam formation. These isolates can be taken from the AS and cultured further using a microbial culture/inoculum development programme for subsequent reintroduction into the wastewater to be treated.

CHARACTERISATION OF FOAM STABILITY AND DESTABILISATION

Parameters associated with foam stability and instability

Foam stability and instability science is complex. Foam can be characterised by various parameters – for example, the dynamic foam test, where foam height is tested against time, surface rheology, surface tension, conductivity, bulk viscosity and/or foam drainage, which can be measured by foam dispersion imaging procedures ([Sakker et al. 1997](#)). The parameters used varies with the distribution of gas molecules in the liquid. In the foam decay test, foam is produced mechanically by air sparging, agitation, and pouring ([Iglesias et al. 1995](#)). The Ross-Miles simple method is conducted by pouring a foaming solution at a definite height through a minute aperture with a known diameter while measuring the foam's height over time. The amount of gas introduced into the solution; that is, the gas dispersion rate, cannot be controlled ([Ross & Miles 1941](#)). Pneumatic methods that measure the amount of gas distributed through the solution also exist, but are time-consuming and unusable as standard procedures ([Pinazo et al. 2001](#)). [Lunkenheimer et al. \(2010\)](#) measured foam's rheological characteristics under defined gas dispersion, and applied deviation and transition time simultaneously by measuring the changes in foam volume and foam drainage. Deviation and transition time characterize different stages of foam decay, of which there are three; the initial stage when the lamellae are still unruptured is described by Equation (1):

$$\frac{\Delta h^F}{\Delta h^S} = 1 \quad \text{and/or} \quad \Delta h^F - \Delta h^S = 0 \quad (1)$$

whereby any level changes of the foam/air (Δh^F) or the solution/foam (Δh^S) boundaries can be quantified simultaneously as a function of time. When the foam drainage and lamellae rupture, transition stage occurs concurrently (Equation (2)) the ratio between the changes in foam/or air and solution/foam must exceed unity.

$$\frac{\Delta h^F}{\Delta h^S} > 1 \quad \text{and/or} \quad \Delta h^F - \Delta h^S > 0 \quad (2)$$

In the final stage, lamellae rupture is dominant and negligible, the liquid is drained from the foam and Equation (3) applies.

$$\frac{\Delta h^F}{\Delta h^S} \geq 1 \quad \text{and/or} \quad \Delta h^F - \Delta h^S \geq 0 \quad (3)$$

The initial deviation time (t_{dev}) occurs towards the end of the initial stage, whereas the time transition (t_{tr}) occurs during the transition and final stages. For unstable foams t_{tr} is <10 s, for stable foams it exceeds 10 s or 100 s for very stable foams (Lunkeinheimer *et al.* 2010). Stable foam crust can also occur in the presence of biofoamers; for example, *M. parvicella* and *N. amarae*, where $t_{tr} >100$ s. The Ross-Miles foam height test can be applied with minor modifications and is still used as a standard operating procedure despite its drawbacks.

Factors affecting foam stability and destabilisation

Foam stabilisation and destabilisation can be controlled by many factors including liquid surface tension, surface viscosity, lamellae rupture, surface elasticity, disjoining pressure, critical micelle concentration and foam drainage. Increases in viscous film thickness, surface viscosity, and/or critical micelle concentration, like decreases in intact lamellae or disjoining pressure, lead to bubble deflocculation and minute thin liquid film drainage, and stabilise the foam. Table 3 list the major factors that affects foam stability and destabilisation.

Table 3 | Factors affecting foam stability and destabilisation

Factors	Foam stability	Foam destabilisation	References
Surface tension	Decreases due to hydrogen bond disruption, resulting in increased interfacial area and energy.	Increases due to the undisrupted hydrogen bonds, resulting in decreased interfacial area and energy.	Pradhan & Batthacharrya (2017)
Surface viscosity	As viscous modulus increases the surface viscosity also increases.	Viscous modulus and surface viscosity decreases.	Wang <i>et al.</i> (2019)
Lamellae film	Electrical double layer is formed, repulsive forces are produced to stabilise the lamellae.	Lamellae rupture, resulting in bubble drainage.	Keal <i>et al.</i> (2016)
Surface elasticity	Increase in elasticity elongates foam life	Decreased elasticity reduces foam life.	Wang <i>et al.</i> (2016)
Disjoining pressure	Strong attractive interactions between the liquid and gas phases strengthen the interfacial film so that it remains intact.	Weak repulsive disjoining pressure causes rupture of the gas-liquid film.	Schramm (1994)
Critical micelle concentration (CMC)	Increased beyond CMC.	Falls below CMC.	Malysa & Lunkenheimer. (2008)
Foam drainage	Bubble deflocculation leads to decreased rate of bubble collision. This lowers the pressure exerted on the thin liquid film and minute liquid drainage.	Bubble flocculation causes the thin liquid film to break. The pressure in the lamellae is lower than that inside the bubble, causing the film to rupture, which increases liquid drainage.	Narsimhan & Xiang (2018)

FACTORS ENHANCING FOAM STABILITY IN WASTEWATER

Influences of wastewater characteristics on foam formation and stability

Foam is formed by gas dispersion and aqueous surfactants (Osei-Bonsu *et al.* 2015) and/or biosurfactants (Santos *et al.* 2016). The surfactants prevent bubbles from coalescing and reduce the wastewater's surface tension (Malysa & Lunkenheimer 2008). Foam stability is determined by assessing foam height or volume with generation time (Osama Al 2014). Foam stability and foamability are correlated because a stable foam means that the solution has high foamability. Surfactant concentration up to the critical micelle concentration (CMC) triggers foam formation under reduced surface tension conditions (Malysa & Lunkenheimer 2008). Foam stability also depends on the foam film's electric double layer, bubble drainage by gravity, surface elasticity and bulk viscosity (Osama Al 2014).

The disjoining pressure difference is the total pressure variation between the gaseous and aqueous phase within a film and is highly dependent on film thickness, which in turn is influenced by the surface tension (Schramm 1994). The difference arises from repulsive electrostatic and attractive van der Waals forces (Derjaguin & Landau 1993). The van der Waals forces arise from the induction of dipole-dipole interactions that dominate in the absence of surfactants, with the disjoining pressure within the film being reduced (Derjaguin & Landau 1993). Any surfactants present penetrate the gas-liquid film and create an electrical double layer. Subsequently, repulsive forces are generated and stabilise the lamellae, which means that the wastewater's ionic strength is crucial for foam stability (Almajid & Kovsky 2016).

The presence of proteins contributes to uniform distribution of bubbles and they stabilise foam at low concentration; that is, 0.1% (Halling 1981). Soluble proteins diffuse to the air-water interface, and reduce surface tension (Narsimhan & Xiang 2018). Their polar groups attach to the aqueous phase while their hydrophobic groups attach to the non-aqueous phase (Fameau & Salonen 2014). They are adsorbed at the lamellae and stabilise foam formation by reducing interfacial tension, increasing the liquid phase's viscosity and elasticity, and strengthening the film; albeit this depends on the temperature, pH and protein concentration (Zayas 1997). Proteins are surface active, yielding wet foam and reducing foam drainage in the presence of other surface-active agents. The quaternary structure of the proteins present can lead to foam formation in reactors, so it is important that its constituents are known. The quaternary structure of proteins contains more than one polypeptide chain based on their amino acid sequence. A protein structure is stabilised by hydrogen, ionic, and disulfide bonds (Willey *et al.* 2008), with the disulfide bonds stabilising the foam at the interface if not disrupted (Zayas 1997).

The effects of oil on foam stability are also complex and influenced by many factors such as the lightness or thickness of the FOG. Tang *et al.* (2018) states that light oil in wastewater disrupts foam whereas thick FOG stabilises foam in the presence of surfactants. Some studies showed that emulsified FOG lengthens foam life and that a pseudo-emulsion film (the film between air and oil droplets) can be used to determine the foam stability thermodynamics (Tang *et al.* 2018). The length of the FOG-hydrocarbon chains in the wastewater also affects foam stability. Shorter chain hydrocarbons adsorb into the foam's liquid-gas interface, disrupting the film, which in turn disrupts the lamellae, leading to foam coalescence (Jones *et al.* 2016). However, long chain hydrocarbons stabilise the foam. At high surfactant concentrations, long half-life foam films are stable because the disjoining pressure decreases, leading to slower gas-liquid film disruption.

Surfactants' influence in foam stability

Surfactants, which can be ionic, cationic and non-ionic, have been applied widely; for example, in the pharmaceutical, food and petroleum industries (Osei-Bonsu *et al.* 2017). They have been used for

flocculation, foaming and de-emulsification (Mulligan 2005), and can increase detergent power, wetting ability and foaming strength (Mulligan & Gibbs 1993), as well as to solubilise FOG, enhance the solubility of polar substances, reduce surface tension, lower the CMC and reduce interfacial tension (De Almeida *et al.* 2018). The critical CMC is influenced by the surfactant's strength (hydrophile-lipophile balance (HLB) number), pH, temperature and ionic strength (Rosen & Kunjappu 2013). Surfactants may generate unstable foam in the presence of FOG but it has been shown that stable foam can be generated in such circumstances when a surfactant with long hydrocarbon chains is used (Yekeen *et al.* 2016). The liquid-film thinning that is caused by FOG influences spreading on the gas-liquid interface (Derjaguin & Landau 1993), thus affecting foam stability mechanisms. Foam stability is affected by FOG because it enters and spreads across the gas-liquid interface and an unstable bridge can be formed that could cause lamellar rupture. FOG spreading across the foam film can also result in pseudo film instability. Foam stability in the presence of FOG can be described by Equations (4)–(6):

$$E = \sigma_{gw} + \sigma_{ow} - \sigma_{og} \quad (4)$$

$$S = \sigma_{gw} - \sigma_{ow} - \sigma_{og} \quad (5)$$

$$B = \sigma_{gw}^2 + \sigma_{ow}^2 - \sigma_{og}^2 \quad (6)$$

where σ_{gw} is the surface tension between the gas and liquid film, σ_{ow} is the surface tension between the FOG and liquid, while σ_{og} is the interfacial tension between the FOG and gas phase (Simjoo *et al.* 2013). If E is positive, the FOG can enter the liquid-gas interface. If S is positive, the FOG droplet stretches and spreads in the liquid gas layer and cause the thinning and rupture of the foam film. If S is negative, the FOG will form a lens and the liquid gas interface will not rupture, although an unstable bridge (B) that can destabilise foam in some circumstances can be created.

Impact of biofoamers in stabilising excessive biofoam

Biofoamers affect AS systems globally and many attempts have been made to determine the predominant biofoamers so that control strategies can be developed. FOG is the major contributor to biofoam stability because it can be biodegraded and assimilated by the biofoamers (Almajid & Kovscek 2016; Osei-bonsu *et al.* 2017; Cisterna-Osorio & Arancibia-Avila 2019). Foam stability depends not only on the hydrophobicity of the cells of biofoam-producing organisms, but also on the presence of solids; the filaments attach to suspended solids and float on the wastewater surface (Fryer *et al.* 2011). In gram-negative organisms, cell lipopolysaccharides and phospholipids are the main contributors of cell hydrophobicity, whereas in gram-positive organisms, polypeptides enhance hydrophobicity, which influences filament formation and nutrient uptake. The nutrients present in the wastewater are also major contributors to foam formation. During starvation, the polysaccharide slime capsules of the filaments are lost due to carbon source limitations. Hence, in the presence of poor hydrophobic cells and low uronic acid concentrations, the foam surface can become unstable. The presence of uronic acid is an indicator of phosphorus deficiency in gram-positive microorganisms and a substitute for polyol phosphate chains of the teichoic acids, required for filament formation (Miłobędzka *et al.* 2015).

Effect of biosurfactants in excessive biofoam stability

Biosurfactants are surface-active biomolecules that are produced by microorganisms during their growth from various nutrient sources; for example, sugars, oils, alkanes, etc (Lin 1996). Biosurfactant CMC varies from 1 to 200 mg/L, and it is relatively better than that of synthetic surfactants because biosurfactants are biodegradable (Osei-Bonsu *et al.* 2017). Microorganisms excrete biosurfactants during nutrient reduction in wastewater – for example, during the reduction of nitrogen availability,

with an example being *Pseudomonas aeruginosa*, which produces rhamnolipids under nutrient-limited conditions. Other microorganisms produce biosurfactants to emulsify and take up the lipids and fatty acids in FOG that are readily available in the wastewater. Actinomycetes have been found to produce surface-active lipid molecules in AS that lower surface tension (Zhang *et al.* 2018). They also produce EPS or byproducts such as biosurfactants that enable them to solubilise, biodegrade, and emulsify the hydrophobic material as their carbon and nitrogen source. Actinomycetes such as *G. amarae* in conventional AS produce sufficient biosurfactants to form foam. The *G. amarae* cells and biosurfactants contribute to foam stabilisation in AS systems (Pagilla *et al.* 2002).

FACTORS THAT INFLUENCE FOAM DESTABILISATION IN WASTEWATER

Conditions associated with foam destabilisation

Foams are thermodynamically unstable and rupture easily if bubbles coalesce (Pajdak-Stós *et al.* 2017), with rupturing occurring as the gas-liquid film thins. Many techniques have been employed to stabilise foam but they are not suitable for AS systems, in which foam is an unwanted nuisance (Fryer *et al.* 2011). The presence of dissolved solids in the wastewater may stabilise or destabilise foam. Soluble solids in the micelle may make foam stable because of their agglomeration in the gas-liquid interphase, leading to a need for the presence of hydrophilic substances to destabilise the foam (Miller 2008; Garrett 2014). The lamellae in the foam can also rupture due to bridging (Rafati *et al.* 2016).

Foam decay behaviour, gravity drainage, bubble coalescence and the lamellae number, should all be considered for foam destabilisation (Pajdak-Stós *et al.* 2017). Foam can be destabilised by silicone and oils that contain short alkane chains, which solubilise in the micelle further reducing micelle volume and the repulsive forces between the micelle, diminishing stratification in the foam film leading to faster foam destabilisation (Hill & Eastoe 2017). Spreading of the oil in the gas-liquid plateau (lamellae connection points) increases lamellae thinning and leads to foam destabilisation, bubble film deformation, which ultimately leads to foam instability and disjoining pressure differentiation. The pressure differentiation would increase foam decay as the lamellae thins. Svarz (1990) invented a good antifoam/defoamer consisting of a polyether surfactant and polyhydric alcohol fatty acid ester. However, its constituents, which are toxic to microorganisms, could damage the environment and reduce sludge efficacy (Conley & Kabara 1973).

Defoamer impact in foam destabilisation

Antifoamers are used to prevent foam generation, whereas defoamers reduce foam that already exists. These terms are sometime used interchangeably, for example, Denkov *et al.* (2014) describes defoamers as chemicals used to collapse foam that incorporate various short chain alkane oils and hydrophobic molecules, while antifoamers prevents unwanted foam from forming (Gastrock & Heid 1938). Prevention of foam dates back to the 1930s when mechanical devices, for example, skimmers, and so on, that consume a large amount of energy were used. In the late 1940s, chemical methods were introduced, including the use of essential oils, kerosene and alcohols. For this reason, post 1970, oils and hydrophobic particles have been used as defoamers (Ross & McBain 1944; Gunderson & Denman 1948; Jacoby & Bischmann 1948).

Although a wide variety of biodefoamers have been commercialised, they are not suitable for all systems and their composition is not always clearly outlined, which makes it difficult to determine a suitable defoamer for any particular system. Even when this has been done, the defoamer might deteriorate in the system, especially when environmental conditions vary; a trait of open-air biological

processes, for example, AS, systems (Mangundu 2017). Therefore, a defoamer's mechanism must be determined; that is, whether it imparts defoamation through bridging and/or stretching. Under normal circumstances, a defoamer enters the FOG-water and gas-liquid interface, and the FOG droplet is spread across the interface, forming a bridge stretching in all directions and causing a rupture at its centre (Figure 1). Since the FOG droplet is hydrophobic when it enters the foam interface, the interface dehydrates. A higher defoamer concentration is then required for it to be efficient. Defoamers can also reduce AS efficiency, reducing the ability of microorganisms to degrade foam and use it as a carbon source. Few microorganisms can degrade FOG constituents in sludge containing LCFAs (Kougias *et al.* 2015).

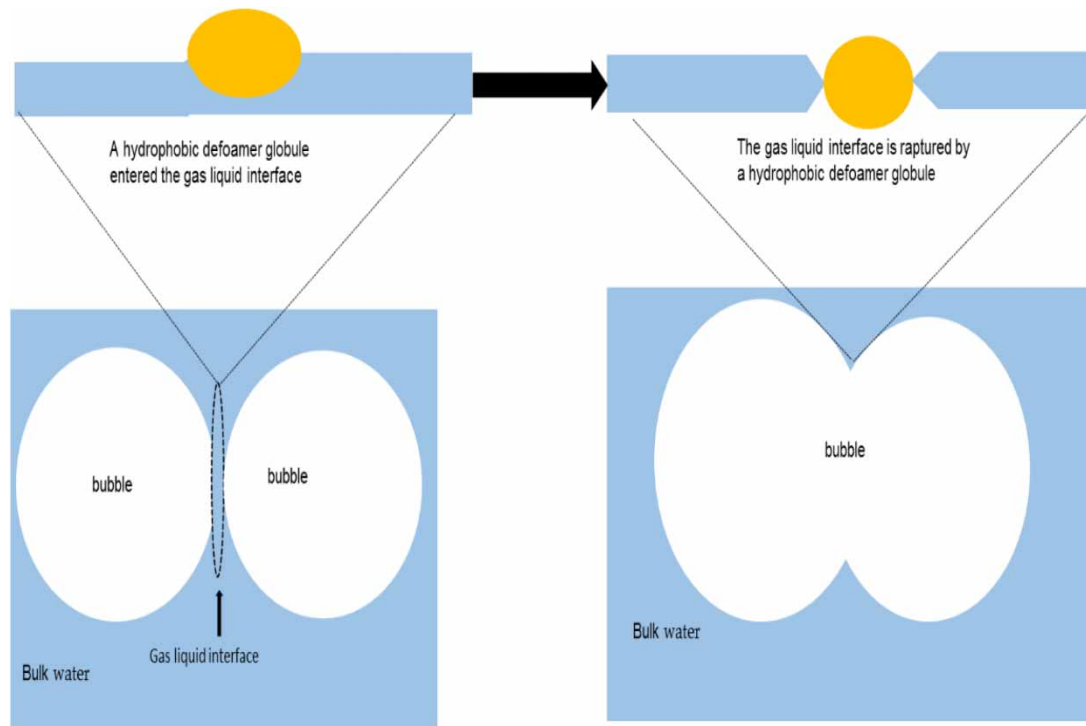


Figure 1 | Rupture mechanism of a gas-liquid interface by a hydrophobic defoamer.

Chemical-based defoamers are added to surfactants to reduce detergency; that is, foam formation, but yield by-products that cannot be biodegraded and are toxic (Cook *et al.* 1997; Brown *et al.* 2005). Brown *et al.* (2005) invented an efficient defoamer containing halogens, epoxy oxygen and an alkyne group. However, halogens are highly reactive, corrosive, and toxic to aquatic life (Pourmoghaddas & Stevens 1995). Surfactants containing a non-silicone type anti-foaming agent such as polyether might also be unsuitable for large-scale use due to their toxicity and/or relatively low levels of solubility at low temperatures (Jin *et al.* 1997). The use of synthetic defoamers such as polyhydric alcohol fatty acid esters destabilise AS systems (Denkov *et al.* 2014). Many defoamers/antifoamers may be efficient, but Pitt & Jenkins (1990) confirmed that microorganism-produced foams are resistant to chemical antifoamers and biodefoamers may be required.

Foam destabilisation by biodefoamer application

Biodefoamers are environmentally benign and biodegradable, when compared to silicone- or oil-based defoamers. They are based on agricultural oils or EPS (microbial polymers secreted by the cell) constituents of microbial origin which contain no fats. They are dioxin precursor compliant

and non-toxic (Bajpai 2017). Their major advantages include improved bubble drainage and reductions in chemical and energy use, and they can be added directly to the AS aeration basin.

Microorganisms can also produce polymers intra- and/or extra-cellularly that can play a definite role in biodefoamer production. In oily wastewaters, such microorganisms can be isolated to produce EPS independent of the AS system, with their bioproducts being concentrated for use as defoaming agents. If a microorganism from an oily wastewater is used to treat the same wastewater, that ecosystem will usually be harnessed and become undisturbed.

Quantifying biofoam destabilisation kinetics

Mathematical models of foam destabilisation are required, because foaming mechanisms are complex, and such models will provide a better understanding and optimisation of the foam destabilisation process and the opportunity to determine foam destabilisation efficiency. Studies of foam destabilisation kinetics will improve understanding of the processes causing foam decay, liquid drainage disproportionation, and foam collapse when a biodefoamer is used. Liquid foam destabilisation occurs when the lamellae becomes thin, thus causing the bubbles to disintegrate and leading to liquid drainage. Gas entrapment in a bubble can also be disproportionately higher than atmospheric pressure, potentially causing bubble bursts. Sceni & Wagner (2007) noted that some mechanisms, which can be described using defoamation kinetic models, occur concurrently because of the breakage of the interfacial film, which results in the foam collapsing. They studied the destabilisation kinetics of sodium caseinate dispersion in foam using multiple-scattering, quick-scan, optical microscopy and traditional volumetric analysis. A kinetic model for estimating liquid drainage and foam collapse was then formulated (Equation (7)),

$$\alpha(t) = \alpha_{max} \frac{t^n}{t^n + t_{1/2}^n} \quad (7)$$

where: $\alpha(t)$ is the volume of liquid drainage as function of time, α_{max} is the maximum volume of drained liquid, n is the sigmoidal character of the curve, and $t_{1/2}$ is the half-life of drainage or collapse (the time needed to drain half of the maximum volume) (Sceni & Wagner 2007; Delgado-Sánchez *et al.* 2017). This method can be used for foam collapse kinetics with the area $\alpha(t)$ of collapse as the function of time and α_{max} the maximum area of the collapse both of which can be estimated. Equation (8) can also be fitted to kinetic profiles generated using a multiple-scattering quick-scan and data obtained using optical microscopy under different defoamer concentrations (Jin *et al.* 1997).

$$BS_{(t)} = a + b(-t/k) \quad (8)$$

where $BS_{(t)}$ is back scattering as a function of time t . $a + b$ is the proximity of $BS_{(max)}$, and k is inversely proportional to the $BS_{(t)}$ fall rate.

Only chemical- and FOG-based defoamer destabilisation kinetics have been studied to date, with no biodefoamer studies reported. To elucidate defoamer globule influences on foam destabilisation kinetics, pneumatic defoamer movement must also be understood, as bubbles under turbulence float defoamer globules pneumatically. Foam bubbles aggregate in the presence of defoamer globules, which is what determines bubble longevity. Defoamer efficiency can thus be estimated by quantifying the ratio (F) of the foam volume (V_1/V_2) generated in the presence of the defoamer (V_2) in comparison with foam volume (V_1) generated in the absence of a defoamer – Equation (9). The parameters required for Equation (9) are bubble size distribution [$P(rb)$] and the probability of bubble persistence ($qiCiY(rb)Zi$) for a bubble with radius r_b that is bound to antifoam droplets ($CiY(rb)Zi$). Similarly, the concentration of the defoamer droplets (Ci) of specie i , and the function of bubble size dissemination,

$Y(rb)$, including the average length of mono bubble (L) during mixing (Z), are all required. Defoamer globule dispersion during mixing is quantified according to size, which can also be estimated as $(\frac{4}{3})\pi\rho\int_0^\infty \chi r_i^3 dr_i$, where r is the size of a defoamer aggregate of specie i , and P is the defoamer's average density.

$$F = \frac{V_2}{V_1} = \frac{\int_0^\infty P(r_b)r_{bq_i}^3 C_i Y(r_b)Z dr_b}{\int_0^\infty P(r_b)r_b^3 dr_b} \quad (9)$$

Usually, some of this model's functions are unknown; for example, Z_i , q_i , $P(rb)$, $Y(rb)$ but it can be used to understand the experimental data (Karakashev & Grozdanova 2012).

CONCLUDING REMARKS

Foam can be removed or reduced in various ways; that is, using physical, chemical and/or biological methods in AS systems. All, however, have limitations. Physical methods often require additional and costly WWTP equipment, while chemical dosing can reduce microbial (AS) system efficiency over time. Therefore, biocontrol methods for foam formation might be appropriate as they are generally environmentally benign and can be applied in full-scale WWTPs. Bacterial cells and their intra- and extra-cellular polymeric bioproducts can be used as bio-controllers to reduce foam generation. Such microorganisms can be isolated from the wastewater to be treated so that, once harnessed to produce defoamers, there are no negative effects on the wastewater treatment system. This, in turn, will improve WWTP operability while reducing the abundance of the microbial filaments that provide the buoyancy of the foam producing microbial populations in WWTPs. Future biodefoamer application in different wastewater treatment apparatus could lead to advances in WWTP designs.

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DATA AVAILABILITY STATEMENT

All relevant data are included in the paper or its Supplementary Information.

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