Journal of Environmental Management 157 (2015) 139–145

Contents lists available at ScienceDirect

Journal of Environmental Management

journal homepage: www.elsevier.com/locate/jenvman

Evaluation of granular sludge for secondary treatment of saline municipal sewage

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ARTICLE INFO

Article history:
Received 5 January 2015
Received in revised form
10 April 2015
Accepted 15 April 2015
Available online

Keywords:
Granular sludge
Salinity
COD
Nitrous oxide
Dissolved oxygen

ABSTRACT

This study examined the impact of chemical oxygen demand (COD) loading and dissolved oxygen (DO) concentration on the stability and performance of granular sludge treating high saline municipal sewage. Under high DO concentrations of 4.0–7.0 mg/L, and COD loading rates of 0.98 and 1.55 kg/m^3/d, rapid settling granules were established within four weeks of start-up. Under the highest COD load, a reduction in DO lead to the rapid deterioration of the sludge volume index (SVI) and washout of granules due to prolific growth of the filament *Thiothrix* Type 021N. Conversely, when operated under a lower COD load, a reduction in DO concentration had no adverse impact on the stability of SVI and granules. A decrease in DO also improved nitrogen removal performance, where simultaneous removal of ammonium (98%), total nitrogen (86%) and BOD_5 (98%) were achieved when median DO concentrations were between 1.0 and 1.5 mg/L. Phosphate removal was lower than expected, however the level of biological phosphate removal activity observed appeared sufficient to maintain granule stability, even under low DO concentrations. Nitrous oxide emissions were also characterised, which ranged between 2.3 and 6.8% of the total nitrogen load. Our results confirmed that granular sludge is a viable option for the treatment of saline sewage.

1. Introduction

In recent years, new technologies have been developed to improve the settling properties of activated sludge and the use of granular sludge in sequencing batch reactors (SBRs) is a novel solution that has the potential to become industry standard for secondary treatment. Extensive research has shown that the operation of SBRs can be modified to convert slow settling activated sludge flocs into dense microbial granules, which have superior settling properties (Beun et al., 1999; de Bruin et al., 2004; Etterer and Wilderer, 2001). In particular, granular sludge is achieved by employing a short settling phase to select for fast settling biomass, coupled with an anaerobic feed that encourages the development of slow-growing organisms, such as polyphosphate accumulating organisms (PAOs). PAOs convert easily degradable substrates, such as acetate, into microbial storage polymers, thereby gaining a competitive advantage over floc-forming organisms (Bassin et al., 2012; Beun, 2001; Beun et al., 1999; De Kreuk et al., 2005).

The potential benefits of a granular sludge system are numerous. Granules have excellent settleability, resulting in a highly clarified effluent and, like biofilms, have an oxygen gradient which facilitates efficient and simultaneous nitrification-denitrification and biological phosphorus removal (Bassin et al., 2012; Pronk et al., 2013). The rapid settling phase enables shorter cycle times, which increase the hydraulic capacity or reduce physical footprint, and the fast settling granules allows the reactor to retain more active biomass than a conventional SBR of the same size (Pronk et al., 2013).

To date, granular sludge has been incorporated into several full-scale industrial and municipal WWTPs in Europe, South America and South Africa, however this technology has not yet been utilised at full-scale in Australia. For Australia, granular sludge may offer the

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http://dx.doi.org/10.1016/j.jenvman.2015.04.027
0301-4797/

potential to enhance the performance and capacity of existing SBRs; particularly those that service regional and coastal areas, which operate under challenging conditions as they are subject to high and variable inflows, and high salinity caused by substantial ground water intrusion into ageing sewer networks. This in turn has adverse effects on treatment performance leading to carryover of suspended solids from bulking sludge and loss of nitrification. Modifying the operating parameters of existing SBRs to select for granular sludge may be a cost effective solution to improve effluent quality and increase hydraulic capacity to cater for high flows, and may be achieved without increasing reactor volume.

Whilst granular sludge has recently made entry into secondary wastewater treatment, little is known about the impacts of high saline sewage on granule formation and stability given that salt is known to inhibit biological processes such as nitrification, denitrification and phosphate removal (Pronk et al., 2013). To date, published data on the impacts of salt on granular sludge formation and operation is limited to a few papers and in some cases, the findings are conflicting. Typically, high salt has been shown to inhibit biological phosphorous removal, which is considered central to granule stability, particularly when required to operate under low DO concentrations (de Kreuk et al., 2005; Welles et al., 2014). Inhibition of PAO activity can be directly caused by salt or by the presence of inhibitory nitrite that accumulates as a result of salt inhibition of nitrite oxidising microorganisms (Welles et al., 2014; Zheng et al., 2013). Figueroa et al. (2008) reported that high saline wastewater (9 g Cl^-/L) from a fish canning factory did not have a detrimental effect on the operation of a granular sludge reactor, however granule formation was delayed and nitrogen removal was poor owing to the accumulation of nitrite. Similarly, in a study where influent salt concentration was incrementally increased from 0.2 to 20 g Cl^-/L, Pronk et al. (2013) showed that the structure of granules remained stable, despite an increase in nitrite and reduction in PAO activity up to 20 g Cl^-/L, at which point granule size decreased and high effluent turbidity was observed. Conversely, Taheri et al. (2012) showed that salinity concentrations greater than 5 g Na Cl^-/L produced light brown and ‘fluffy’ granules, which contributed to severe biomass washout.

To date, most of the published data about salinity impacts on granular sludge were largely derived from trials conducted using bench-scale reactors fed with readily biodegradable synthetic wastewater. Without complementary field based research we can only assume that these observations reflect the performance of granular sludge systems fed with real sewage. Sewage however is a far more complex matrix, where COD is comprised of numerous elements with varying rates of biodegradability and these differences have been shown to influence granular sludge morphology and performance (De Kreuk, 2006; Moy et al., 2002). Accordingly, the aim of this study was to examine the application of granular sludge technology for secondary treatment of high salinity municipal sewage in South Australia. Granular sludge pilot-plant trials were conducted at a high saline municipal wastewater treatment plant (WWTP) in parallel to a full-scale conventional floc-based SBR. The pilot was seeded with activated sludge floc from the neighbouring full-scale SBR, which was already adapted to high saline conditions. The impact of high saline sewage on granular sludge was investigated in combination with changes in COD loading and DO concentration, which are crucial regulators of granular sludge stability and treatment performance (Zhou et al., 2014). This work represents a significant step in developing a better understanding of critical operating conditions needed to enable stable operation of granular sludge for the treatment of high saline conditions.

2. Materials and methods

2.1. Study site

A pilot-scale granular sludge SBR was operated at the Bolivar High Salinity (HS) WWTP in South Australia in parallel to a full-scale SBR. Bolivar HS WWTP treats municipal sewage that is high in salinity due to groundwater infiltration from various seaside sewer catchments northwest of Adelaide. Typical influent characteristics are described in Table 1. Sewage salinity at this site typically ranges between 5.8 and 7.0 g/L TDS, which is mostly in the form of $NaCl^-$ (*ca* 5.3–6.1 g/L). For the duration of the trial, salinity was stable, ranging between 5.5 and 6.0 g/L TDS.

2.2. Granular sludge SBR pilot-plant description

Following initial screening through 3 mm contra shear drum screens and grit removal at an off-site pumping station, sewage was screened once more though a 0.95 mm mesh screen before entering the pilot-plant. The key design components of the pilot-plant are presented in Fig. 1. SBR cycle times were controlled using a Siemens programmable logic controller (PLC) and touch screen which allowed versatility in setting cycle times needed for granular sludge development (Table 2). The pilot SBR consisted of a 60 L (1 m high $\times$ 0.3 m diameter) Perspex reactor, a peristaltic feed pump (Verderflex, Dura 10) and air compressor (Hailea, ACO-009) for aeration. Air was supplied through a 5 inch diameter, Perma-Cap fine bubble air diffuser, placed at a height of 10 cm from the bottom of the reactor, at a flow rate which was varied between 300 and 900 L/h, depending on the desired DO concentration. DO was monitored using an optical DO sensor (Endress + Hauser Australia, Oxymax COS61D) and a 300 Watt aquarium heater (Ocean Free) was used to maintain bulk liquid temperatures above 25 °C during winter. Effluent was discharged under gravity at 50 cm from the bottom of the SBR using a decant pipe positioned inside the reactor. The discharge of effluent was controlled using an actuated valve (Belimo, NMQU24). The reactor was fed to reach a height of 90 cm and the final decant height could be varied between 50 and 89 cm to achieve the desired volume exchange ratio (VER), COD loading, and hydraulic retention time (HRT). Reactor water levels during feeding and decant were controlled by the PLC which was based on pressure measurements received from a transducer (BD Sensors, DMP331) that was positioned at the bottom of the SBR.

Start-up was performed using the operational parameters presented in Table 2. Operational parameters of the full-scale SBR are also included for comparison. The experiments consisted of two phases. In phase I, the mean COD loading was 1.55 kg/m^3/d, while in phase II the load was reduced to 0.98 kg/m^3/d. Changes in COD loading were achieved by varying the VER. Before each phase, the reactor was initially seeded with flocculated activated sludge from full-scale SBR to give a final biomass concentration of 2 g TSS/L when completely filled. The pilot SBR was filled anaerobically from the bottom, in a plug-flow fashion in accordance with De Kreuk et al. (2007). DO was maintained between 4 and 8 mg/L for the first 15 days (phase I) and 50 days (phase II) and was reduced to 1.0–2.5 mg/L thereafter to enhance denitrification. In phase I the settling phase was maintained between 7 and 10 min. In phase II, the settling was initially set at 20 min for the first 3 days and was reduced to 15 min between days 14–52; 10 min between days 52–61; and 6–8 min thereafter. Settling time was only reduced when biomass settleability (as defined by SVI) had shown signs of improvement.

2.2.1. Full-scale SBR description

Bolivar HS WWTP is comprised of six reactors. Each reactor