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Full scale performance of the aerobic granular sludge process for sewage treatment

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ABSTRACT

Recently, aerobic granular sludge technology has been scaled-up and implemented for industrial and municipal wastewater treatment under the trade name Nereda®. With full-scale references for industrial treatment application since 2006 and domestic sewage since 2009 only limited operating data have been presented in scientific literature so far. In this study performance, granulation and design considerations of an aerobic granular sludge plant on domestic wastewater at the WWTP Garmerwolde, the Netherlands were analysed. After a start-up period of approximately 5 months, a robust and stable granule bed (>8 g L^{-1}) was formed and could be maintained thereafter, with a sludge volume index after 5 min settling of 45 mL g^{-1}. The granular sludge consisted for more than 80% of granules larger than 0.2 mm and more than 60% larger than 1 mm. Effluent requirements (7 mg N L^{-1} and 1 mg P L^{-1}) were easily met during summer and winter. Maximum volumetric conversion rates for nitrogen and phosphorus were respectively 0.17 and 0.24 kg $(m^3\ d)^{-1}$. The energy usage was 13.9 kWh $(PE_{150}\cdot year)^{-1}$ which is 58–63 % lower than the average conventional activated sludge treatment plant in the Netherlands. Finally, this study demonstrated that aerobic granular sludge technology can effectively be implemented for the treatment of domestic wastewater.

1. Introduction

Aerobic granular sludge (AGS) technology is an upcoming technology for the treatment of domestic and industrial wastewater (Heijnen and Van Loosdrecht, 1998; Morgenroth et al., 1997; de Bruin et al., 2004; de Kreuk et al., 2007; Coma et al., 2012; Show et al., 2012; Morales et al., 2013). AGS technology for combined carbon, nitrogen and phosphorous removal is based on a repeated fed batch process and relies on microorganisms selected to grow in granules rather than flocs. As a result of the high settling rate of the sludge granules, separate settling tanks are not needed and an 80% reduction in area use is possible (de Bruin et al., 2004).

Aerobic granules are characterised by a compact structure, without the need for carrier material, resulting in high settling velocities and a low sludge volume index (SVI). A good indication of granulation is the limited difference between SVI after 5 and 30 min (Etterer and Wilderer, 2001). Aerobic granules are also characterised by their layered structure. The presence of an aerobic outer layer and an anaerobic or anoxic core, facilitates co-existence of nitrifying organisms in the outer layers of the granules and denitrifying phosphate accumulating organisms (dPAO), as well as (facultative) anaerobic organisms towards the centre of the granules (Gieseke et al., 2001; Winkler et al., 2012; Pronk et al., 2015). Due to this structure, aerobic granular sludge can simultaneously remove phosphorus, nitrogen and COD (chemical oxygen demand) from the liquid (de Kreuk et al., 2005; Gonzalez-Gil and Holliger, 2011).

Aerobic granular sludge technology was developed during the last decade at laboratory scale (Morgenroth et al., 1997; Beun et al., 1999; Tay et al., 2002; Zeng et al., 2003), as well as pilot scale (Morales et al., 2013; Liu et al., 2010; Isanta et al., 2012; Wei et al.,

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2012; Li et al., 2014). Aerobic granular sludge is applied by Royal-HaskoningDHV in the Nereda® technology which was first adapted for industrial applications and then further scaled-up for domestic sewage treatment. Valuable scale-up experience gained from full-scale demonstration plants in Gansbaai, South Africa and Frielas, Portugal were used for optimization of the process design and construction in 2010 of the first full-scale AGS wastewater treatment plant in Epe, The Netherlands (van der Roest et al., 2011; Giesen et al., 2013). The AGS technology used relies on a sequencing fed-batch process with a constant working volume. This is possible due to simultaneous feeding and effluent discharge, that relies on a plug-flow pattern for displacement of effluent from the reactor (de Kreuk et al., 2005). In addition, as a result of an oxygen gradient within the granular sludge particle during aeration, extensive biological phosphate removal and simultaneous nitrogen removal can be achieved during one aeration step. The absence of conventional recycle pumps, sludge return pumps and mixers provides a significant reduction in electricity consumption compared to standard nutrient removal plants.

In July 2013, a full scale installation based on the aerobic granular sludge process was taken into operation in Garmerwolde, the Netherlands.

Few papers have been published so far describing full scale operation of the AGS process on domestic wastewater. Li et al. (2014) showed the performance of an full-scale AGS plant fed with 30% domestic and 70% industrial wastewater (BOD/COD = 0.23). The full-scale installation in Epe, the Netherlands briefly described by Giesen et al. (2013) also treats wastewater that is derived for a large part (35%) from industry (mainly slaughterhouses). Moreover, data provided in these studies are very general. More detailed descriptions of the process, conversions, energy usage and design considerations when treating domestic wastewater are lacking.

After start-up and more than one year of operation, this paper reflects not only on the performance, but also on granulation, COD, nitrogen and phosphorus conversions and especially design considerations. The measured energy requirement of the AGS process is compared to conventional activated sludge systems. Furthermore, differences between the full scale granular sludge process, conventional activated sludge and laboratory reactors are discussed in detail.

2. Materials and methods

2.1. Description of plant

In 2012 water board Noorderzijlvest decided that the treatment plant in Garmerwolde would be upgraded with the aerobic granular sludge process. The extension of the existing activated sludge based sewage treatment process (STP) was necessary to meet the effluent requirements. The STP in Garmerwolde treats approximately 27 million m^3; of wastewater per year and 0.8 MW of electricity is generated by the use of the biogas formed during sludge digestion. The existing WWTP consists of a two-stage activated sludge plant (so-called AB-process), with chemical phosphorus removal and either glycerol or methanol for denitrification (Böhnke, 1978) and an SHARON reactor to treat side-streams from the plant's digester and sludge thickeners (Hellinga et al., 1998).

The AGS plant was designed by Royal HaskoningDHV and is operated in parallel with the existing AB-plant. The AGS plant treats 41% (28,600 $m^3\ d^{-1}$) of the total influent received at Garmerwolde WWTP during dry weather flow with a maximum of 4,200 $m^3\ h^{-1}$. The average flow received by AGS and the AB-plant totals approximately 70,000 $m^3\ d^{-1}$ with a peak flow of 11,600 $m^3\ h^{-1}$.

Wastewater characteristics are given in Table 1. The designed sludge loading rate was 0.10 kg COD (kg TSS d)$^{-1}$ at an expected sludge concentration of 8 kg m^{-3}. The sludge-loading rate is calculated by dividing the treated kg COD per day by the total biomass present in the reactor (Table 2). The volumetric loading rate of the AGS reactors is 1.5 m^3 (m^3 d)$^{-1}$. Wastewater enters the plant by a pressure main. After screening by 6 mm screens, the wastewater goes to a grit removal plant and an influent buffer (4,000 m^3) (Fig. 1). From the influent buffer, the wastewater is fed to two AGS reactors (height 7.5 m, volume 9,600 m^3 each) that are equipped with an internal recirculation system (top to bottom of reactor) with a capacity of 2,500 $m^3\ h^{-1}$ for each reactor. Treated effluent is directly discharged from the reactors to the surface water via static fixed overflow weirs.

Biological phosphate removal in the AGS process can be supplemented by metal salt addition directly in the bulk if necessary. Surplus sludge is stored in a sludge buffer tank (400 m^3). To prevent anaerobic phosphorus release and to ensure continuous discharge towards the mechanical belt thickeners, the retention time in the surplus sludge buffer is kept to a minimum.

The AGS plant is operated as a sequencing fed batch process, consisting of a simultaneous feeding and effluent withdrawal period, a reaction period, and a settling/sludge withdrawal/idle period. Nitrogen removal is predominantly established by simultaneous nitrification and denitrification, but for maximisation of nitrogen removal (non-mixed) anoxic periods with a recycle from top to bottom can be provided. The cycle can be adjusted to the influent characteristics (rain or dry weather conditions), the actual sludge conversion rates, the desired effluent conditions and the granular sludge selection pressure. During aeration periods, the dissolved oxygen concentration was maintained between 1.8 and 2.5 mg L^{-1}. The total operational cycle time of the reactors is 6.5 h at dry weather conditions. During rainy weather, the cycle time is shortened to 3 h by decreasing the aeration and increasing the feeding time in order to treat the increased influent flow (Fig. 2). The reactor was seeded with surplus sludge from an existing full-scale AGS plant in Epe, the Netherlands, treating wastewater that consist out of a large industrial part (slaughterhouses) to a concentration of 1 g L^{-1}. In this surplus sludge, no granules were present (SVI_{30} 140 mL g^{-1}).

2.2. Online measurements

Each reactor is equipped with measurements for dissolved oxygen concentration, redox potential, temperature, water level, dry matter and turbidity. Ammonium and phosphate are semi continuously measured (5–10 min interval) during the cycle by an automatic sampling and analysis device (Hach Lange; Filtrax, AMTAX and PHOSPHAX). Sampling points for ammonium and phosphate are located on 0.5 m under the water surface in the reactors. This means that during feeding, when the reactor is not mixed the concentrations of ammonium and phosphate in the effluent can be followed. During the feeding, the liquid an S:CAN spectro:lyser[tm] probe from Interline was used to continuously monitor nitrate concentrations at 0.5 m under the water surface. This means that during feeding, when the reactor is not mixed the concentrations measured are the effluent concentrations.

2.3. Grab samples

Periodic sampling for the determination of the concentration of the biomass present in the reactor is performed by a grab sampler. Samples can be acquired pressureless at various depths to account for possible segregation over the depth of the reactors due to granulation. Normally, samples are only taken when the reactor is well mixed during the reaction period.

Table 1
Influent and effluent of the aerobic granular sludge plant in Garmerwolde, Netherlands.

	Influent				Effluent	
	Min.	Max.	Average	Load	Effluent requirements	March–December 2014 average
Parameter	[mg L^{-1}]	[mg L^{-1}]	[mg L^{-1}]	[kg d^{-1}]	[mg L^{-1}]	[mg L^{-1}]
Suspended Solids	101	465	236	8,000	30	20
COD	146	715	506	14,636	125	64
BOD_5	60	420	224	5,495	20	9.7
Total Phosphorus	1.9	9.7	6.7	212	1	0.9
PO_4^{3-}–P dissolved	1.5	6.8	4.4		–	0.4
Total Nitrogen	14	81	49.4	1,387	7	6.9
NH_4^+-N dissolved	13.4	56.5	39.0		–	1.10

Table 2
Operational parameters of the Nereda® granular sludge plant in Garmerwolde, Netherlands.

Data obtained between March and December 2014					
Parameter	Unit	Value	Parameter	Unit	Value
Solid retention time	d^{-1}	20–38	Max Recycle ratio	–	0.3
Dry weight	kg m^{-3}	6.5–8.5	Fe(III)/P ratio[d]	–	0.18
Ash content	%	25	Volumetric phosphate uptake rate[b]	kg P (m^3 d)$^{-1}$	0.011
Sludge loading total[a]	kg COD (kg TSS d)$^{-1}$	0.10	Max. volumetric phosphate uptake rate[c]	kg P (m^3 d)$^{-1}$	0.24
Sludge loading biological[b]	kg COD (kg TSS d)$^{-1}$	0.12	Volumetric ammonium uptake rate[b]	kg N (m^3 d)$^{-1}$	0.058
Sludge production	kg d^{-1}	3900	Max. volumetric ammonium uptake rate[c]	kg N (m^3 d)$^{-1}$	0.17
Hydraulic retention time	h^{-1}	17	Energy	kWh (m^3 year)$^{-1}$	0.17
Volumetric load	m^3 (m^3 d)$^{-1}$	1.45	Energy	kWh (kg N)$^{-1}$	3.6
Nitrogen Load	Kg N_{tot} (kg TSS d)$^{-1}$	0.011	Energy[e]	KWh ($PE_{150,\ rem.}$ year)$^{-1}$	13.9

[a] Based on kg COD received per day over the total biomass present in the reactor.
[b] Based on the aerated time.
[c] Based on actual rates measured inside the cycle at 20 °C.
[d] Only during heavy rain weather events. Molar ratio is given.
[e] P.E.: Population equivalent or 1 i.e. equals 150 mg TOD per day.

2.4. Physical characteristics of granules

Approximately 1,500 mL of reactor volume was washed over a 212 and 600 μm sieve. The sieved granules and 100 mL of un-sieved sample were dried at 105 °C until no weight change was measured. The dried fractions were then used to calculate the percentage of granules and the total biomass concentration in the reactor. Sludge volume index was measured by pouring 1,000 mL of sample in a graduated measuring cylinder. Volume of the biomass was consequently recorded after 5 and 30 min. Only undiluted samples were used in determining the SVI.

2.5. Fluorescent in situ hybridisation

Sample handling, fixing and staining was performed as described in Bassin et al. (Bassin et al., 2011a). Probes used to stain all bacteria consisted out of a mixture of EUB338, EUB338II and EUB338III (Amann et al., 1990; Daims et al., 1999). To stain the PAOs present, a mixture of PAO462, PAO651 and PAO846 described by Crocetti et al. (2000) was used. Probes GAOQ431 and GAOQ989 were used to stain the glycogen accumulating organisms (GAO) population (Crocetti R et al., 2002). The samples were examined with a Zeiss Axioplan 2 epifluorescence microscope equipped with filter set 26 (bp 575–625/FT645/bp 660–710), 20 (bp 546/12/FT560/bp 575–640), 17 (bp 485/20/FT 510/bp 5515–565) for Cy5, Cy3 and fluos respectively.

3. Results

3.1. Description of start up

The first year of operation of the treatment plant can be divided into two periods, based on the extent of granulation and conversion efficiencies established. The first period was characterised by a gradual increase of the volumetric loading rate to the design loading rate and the development of granules from the flocculent sludge (September 2013–February 2014). Reaching design loading rate took about 3 months with fluctuating conversion efficiencies. In period two reactor operation and process efficiencies were optimized (March–December 2014).

Preceding period one, technical testing was performed in July and August 2013. Both AGS plant and the existing conventional activated sludge treatment plant were expected to meet nitrogen and phosphorus effluent standards of 15 and 1 mg L^{-1} respectively. To safeguard these effluent quality requirements, the loading rate towards the AGS reactors was increased only as far as the effluent quality allowed. This meant that on some occasions during the start-up period, the loading rate had to be temporarily reduced. Within 3 months the design flow rate was reached and 41% of the wastewater received at the WWTP was treated by the aerobic granular sludge plant (Fig. 3a). Nitrogen, phosphorus and COD removal were already stable during a part of period 1, but were not optimized yet (Fig. 5). Sludge volume indexes, decreased significantly during start-up (Fig. 3b). Initially, the SVI_5 and SVI_{30} were 145 and 90 mL g^{-1}, but they decreased during the start-up period to approximately 70 and 50 mL g^{-1} respectively. The difference in SVI_5 between the samples from the reactor and those from the surplus sludge continued in the second period. SVI values kept improving until stable values were reached of 45 and 35 mL g^{-1} for SVI_5 and SVI_{30} respectively. Biomass concentration increased from 3 to 6.5 kg m^{-3} in the first period (Fig. 4). At the end of the second period it reached a concentration of >8 kg m^{-3}. The percentage of sludge present as granules increased from 30 to >80% over this period.

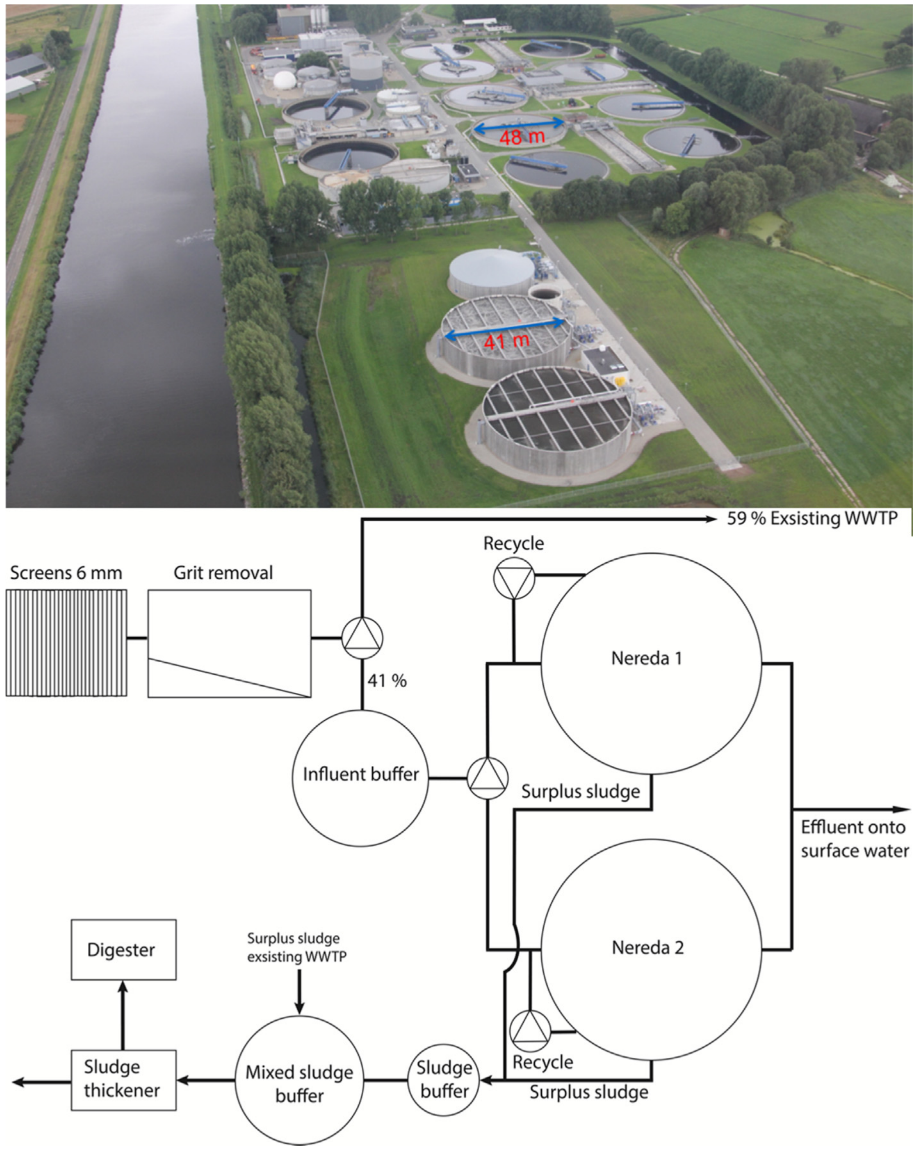

Fig. 1. Garmerwolde WWTP in the Netherlands. The photograph displays the existing AB- treatment system in the background with the aerobic granular sludge plant extension in front. The figure below shows the process scheme of the aerobic granular sludge plant.

3.2. *Performance results during period 2*

February 2014 was considered as the end of period 1. Nutrient removal as well as a significant granule bed was present from that time onward. Performance results and operational parameters obtained between March and December 2014 are summarized in Tables 1 and 2 Monthly average nutrient concentrations are given in Fig. 5. Effluent guidelines were met for all parameters, with an average N_{tot} and P_{tot} of 6.9 and 0.9 mg L^{-1}, respectively, in the period March–December 2014. Elevated nitrogen and phosphorus concentrations in the effluent always coincided with heavy rain weather events. Due to presence of a large sewer system, heavy rain events provoke a flow rate that can be 2–5 times as high (Fig. 3a) with nutrient concentrations as occurring in dry weather flow (so-called first flush). The combination of increased flow and loading rates leads to shorter cycle times and resulted in elevated effluent concentrations. At these times, in accordance with the design set-up targeting an optimized installed reactor volume, metal salts are used for the removal of phosphorus at the end of the cycle. The Fe(III)/P molar ratio during the operational period was 0.18 with a total P-removal efficiency of 90%. During normal rain and dry weather flow the removal of phosphorus is completely biological.

3.2.1. *Granulation*

As the dry weight increased, so did the percentage of granules in the total biomass. In period 2 more than 80% of the TSS consisted of granules larger than 0.2 mm with more than 60% above 1 mm. Since selection pressure on settling behaviour is needed for the development of well settling granular sludge during start-up (Beun et al., 1999; Qin et al., 2004), the smaller and less settling sludge should be removed with the surplus sludge, while larger granules should be maintained in the reactor. Barely any granules larger than 0.2 mm were detected in the surplus sludge and consisted mostly out of relatively slow settling flocs, which indicates good separation between slow and fast settling biomass (Fig. 7a, b, c). Protozoa, mainly ciliated, are present on the outside of the granules. Sliced granules examined under a stereo zoom microscope revealed a layered structure with micro colonies present inside (Fig. 7d).

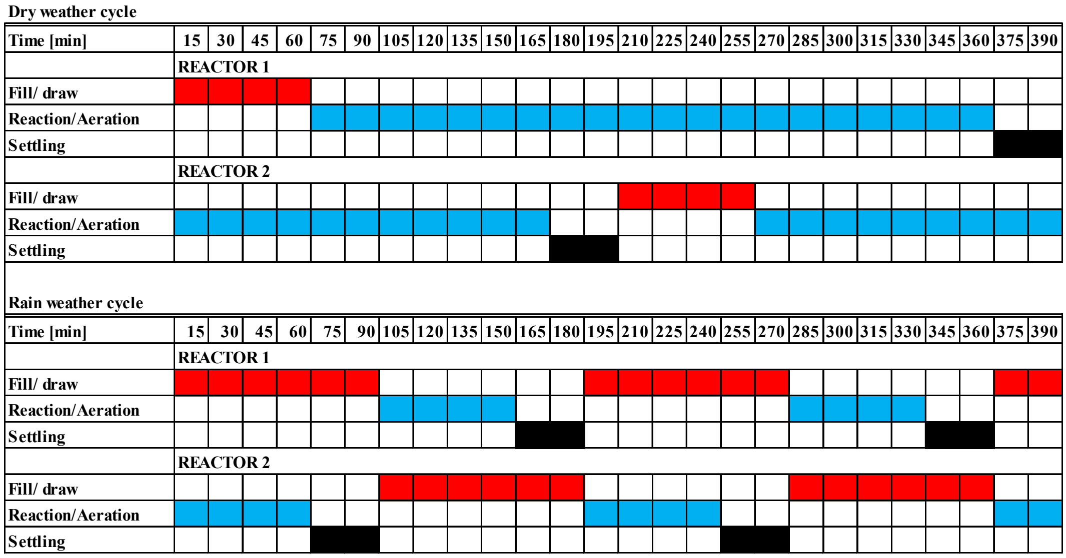

Fig. 2. Typical cycle timing during dry and rain weather operation of the aerobic granular sludge plant.

3.2.2. Fish

Fluorescent in situ hybridisation performed on pottered granules showed the presence of a significant PAO population (Fig. 7e, f). Most of the bacteria stained with the EUB probe (blue) (in the web version), also stained with the PAO probe (red) resulting in the pink colour. In fact, hardly any GAOs could be detected (<5%). Other heterotrophs that are typically stained with the EUB probe were present, but were not further investigated.

3.3. Conversions within a typical Nereda® cycle

The cycle begins with simultaneous influent feeding and effluent discharge (Fig. 2 and Fig. 6). Influent is pumped into the reactor from the bottom with an upward velocity of 3–3.3 m h^{-1} without further mixing or aeration. Simultaneously, effluent is pushed over the overflow weirs at the top of the reactor. In these anaerobic feeding conditions, easy biodegradable organic carbon will be converted into storage polymers by glycogen (GAO) and phosphorus (PAO) accumulating organisms in the granule bed. The uptake of easy biodegradable substrates by PAOs (Fig. 7h) is accompanied by the release of ortho-phosphate into the bulk liquid as can be seen from the relative high phosphate peak at the beginning of the aeration period. The volume exchange ratio in each cycle is limited by dispersion of the influent and thus the imperfection in the plug flow pattern during feeding. Since the sampling points for nitrogen and phosphorus are located 0.5 m under the liquid surface (just below the effluent weirs), the influent feeding can and needs to be stopped as soon as the phosphate concentration increases at this sampling point, since this indicates that a small fraction of the influent is reaching the effluent weir. This occurs when the effective volume exchange ratio is 65%, while during dry weather flow in accordance with the design this effective volume exchange ratio is typically between 30 and 40 %. Ammonium present in wastewater is partially adsorbed by the biomass during the feeding period (Bassin et al., 2011b). Adsorbed ammonium will, however, slowly desorb from the biomass during the aerobic period and consequently be converted. After the feeding period, the reactor is aerated and thus mixed, while the dissolved oxygen concentration is controlled at 1.9 mg L^{-1}. At the start of the aeration period ammonium and phosphate concentration peak due to the mixing of the clean water in the top (where the measurement system is) and the influent in the lower part of the reactor. The concentrations after mixing, combined with the amount of wastewater fed, can be used as an indication of the loading rate of the cycle, but need to be corrected for phosphorus release and ammonium adsorption.

In the aerated period ammonium and phosphorus uptake occurred, while nitrate was produced. Once ammonium reached its desired set-point the oxygen concentration in the bulk was lowered to allow for maximum denitrification potential. Denitrification is taking place inside the granule. Since the diffusion of oxygen into the granule is slower than the conversion rate, an anoxic volume is created inside the granules (de Kreuk et al., 2005; Osman et al., 2001). Storage polymers of the GAOs and PAOs are delivering the reducing power for the conversion of nitrate to dinitrogen gas. Once both nitrogen and phosphorus have reached the required effluent quality the reactor is ready to receive new influent.

4. Discussion

4.1. Start-up

Generally, technological start-up times for nutrient removing conventional activated sludge (CAS) plants range anywhere between 30 and 40 days, assuming the biomass inoculated is adapted to the wastewater and adequate amounts are available. Otherwise longer start-up times are needed to grow and adapt the biomass. Start-up times with respect to effluent quality required for the AGS plant were similar. In this case, the start-up period was a bit longer due to the requirement to achieve full effluent quality standards at all times, even during start-up conditions. This resulted in a 3 months stepwise increase in redirected influent flow from the AB-plant towards the aerobic granular sludge plant.

This study demonstrated that a good effluent quality can be achieved long before granulation is complete. The required flow and treatment capacity was reached in only 3 months, but the intended biomass concentration of 8 g L^{-1} was only achieved after 9 months. The studied reactors were started with surplus sludge of the full scale Nereda in Epe, The Netherlands. Although this surplus sludge was well settling (SVI_{30} = 140 mL g^{-1}), it hardly contained any granular sludge. Development of granular sludge is expected to be shortened drastically by starting with granular sludge from

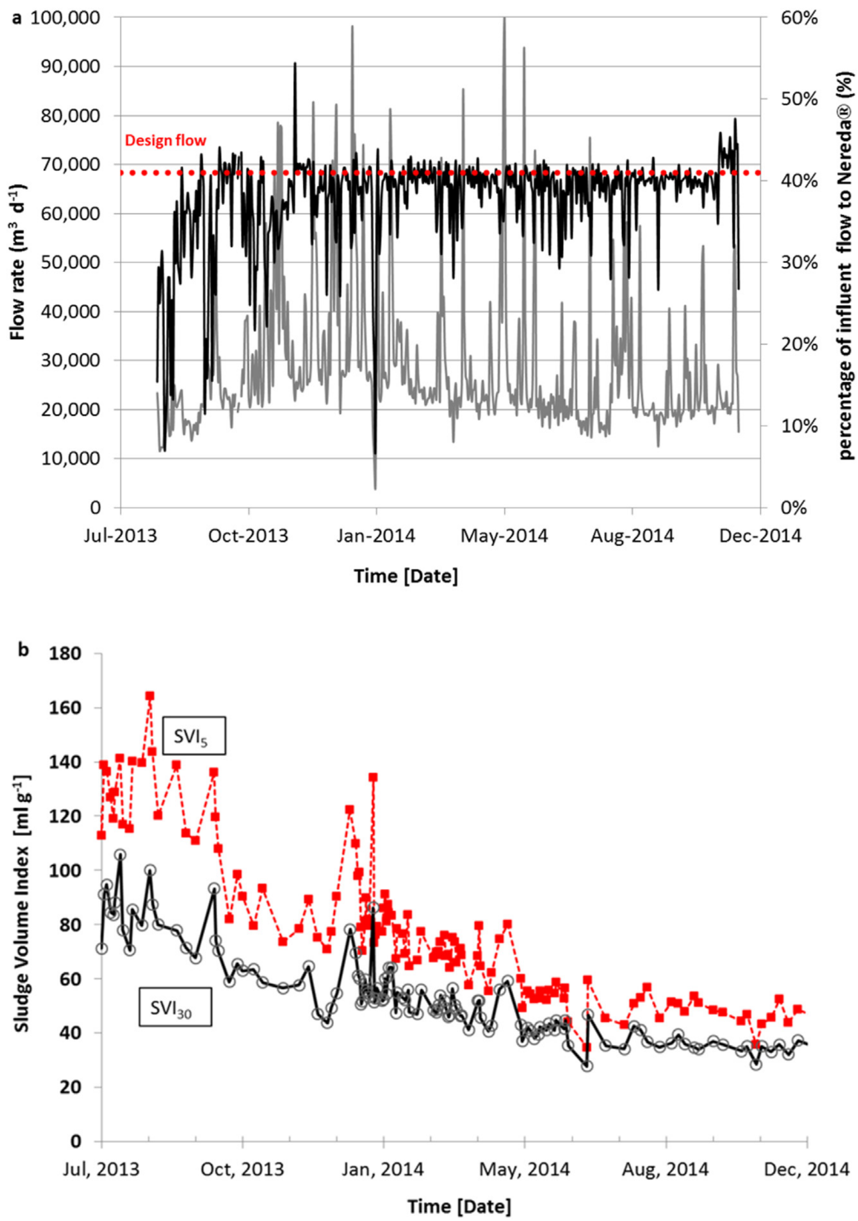

Fig. 3. Flow and biomass characteristics during start-up of the aerobic granular sludge reactor: (a) percentage of the influent flow towards the aerobic granular sludge plant (black line; five day moving average), total influent flow received at the Garmerwolde plant (light grey line) and the design influent percentage towards the aerobic granular sludge plant (dotted line); (b) sludge volume index after 5 (filled cubes) and 30 (open circles) minutes of settling.

other plants (Pijuan et al., 2011). Also, when the effluent criterium is less critical during start-up, the time to reach the design sludge mass can be significantly reduced by increasing the COD load. It should be noted that a treatment plant is designed for the highest load at the coldest temperature. For this reason the required treatment capacity can be achieved during most of the year, even if full granulation is not yet reached.

4.2. Granulation

Successful formation of AGS in a full scale plant treating domestic wastewater was demonstrated in this study. A high percentage (>80%) of the sludge consisted of aerobic granules larger than 0.2 mm. Aerobic granulation on domestic wastewater was shown before in lab-and pilot scale studies, but the full granulation of biomass on full scale was not (Coma et al., 2012; Giesen et al., 2013; de Kreuk and van Loosdrecht, 2006; Wagner and Da Costa, 2013).

As proposed previously in lab-scale and pilot-scale studies, granule formation could also be monitored satisfactorily with the SVI_5/SVI_{30} ratio for this full-scale reactor start-up (de Kreuk et al., 2007; Etterer and Wilderer, 2001). More complicated tests as granular size distribution measurements give more insight, but are not required for full scale operations. In full scale plants there will always be a fraction of flocculent sludge resulting in an SVI_5/SVI_{30} ratio higher than 1 (Fig. 7 a, b). This flocculent material is resulting from influent particulates, detached pieces of granules and growth on polymeric substrates (De Kreuk et al., 2010).

Many factors have been mentioned as critical for the formation of granular sludge, such as high shear, very short settling times and high DO concentrations (Beun et al., 2000a; Liu et al., 2005; Lee et al., 2010). Most of these are related to experiments with

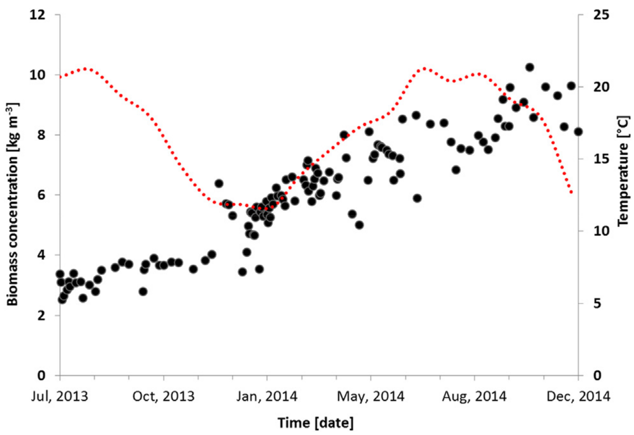

Fig. 4. Evolution of biomass concentration with time (filled circles) and temperature of the bulk liquid (dotted line) in the aerobic granular sludge installation.

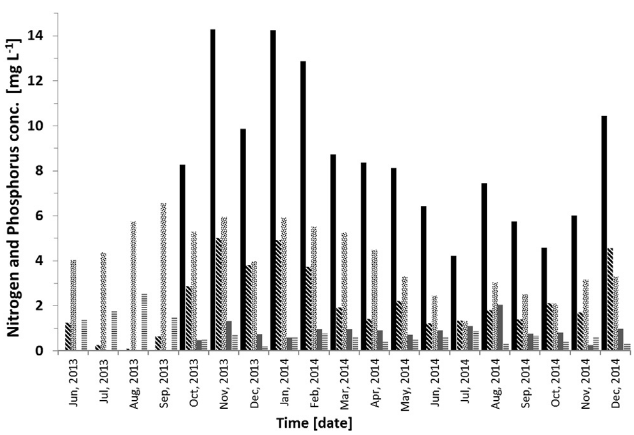

Fig. 5. Monthly nutrient averages in the effluent of the aerobic granular sludge system with (■) total nitrogen, (▧) ammonium, (▒) nitrate, (■) P-total, (▤) ortho-phosphate. Total nitrogen and phosphorus measurements for the aerobic granular sludge plant separately were only available from October 2013.

aerobic or completely mixed feeding. In the case of anaerobic plug flow feeding through the granule bed, most of these factors (like high settling pressure, shear, high DO etc.) become less relevant, although not irrelevant, during start-up (de Kreuk et al., 2005). Anaerobic feeding selects for relatively slow growing bacteria such as PAOs, that convert the easy degradable COD under anaerobic conditions into storage polymers, resulting in smooth and compact biofilms (Van Loosdrecht et al., 1997). The plug flow feeding furthermore ensures high substrate concentrations near the granule surface allowing substrate to diffuse through the entire granule, further contributing to the formation of stable, compact and dense granular biomass.

4.3. Suspended solids in the effluent

Suspended solids in the effluent can originate from the influent, as well as from detached biomass. In the present case the influent contained on average 236 mg SS L^{-1} while in the effluent on average only 20 mg SS L^{-1} was present. The exact removal mechanism of suspended solids in aerobic granular sludge systems is not fully understood yet, however a few relevant observations were made. Suspended material flocculates and can be retained in the AGS plant despite the up flow velocity (3–3.3 m h^{-1}), after which it is removed with the excess sludge. Cellulose fibers, which are present in domestic wastewater (Ruiken et al., 2013), are predominately removed with the excess sludge (Fig. 7b), and do not end up in the effluent. Particles can also be entrapped in the granules themselves (Fig. 7h), where they are degraded or finally end up in the excess sludge as well. Finer material and colloidal particles that are not entrapped can be further removed by protozoa (Lemaire et al., 2008; Priya et al., 2007; Li et al., 2013). In granular sludge, stalked protozoa such as *Vorticella* and *Carchesium*-like organisms attach to the granules (Fig. 7c). Protozoa have also been related to particulate starch removal in laboratory experiments (De Kreuk et al., 2010; Lemaire et al., 2008; Schwarzenbeck et al., 2004) and in biofilm plants fed with suspended bacteria (Lee, 1996), however, their exact contribution to the removal of suspended solids from municipal wastewater has not been quantified yet. Although these protozoa slightly hinder settling, they do not

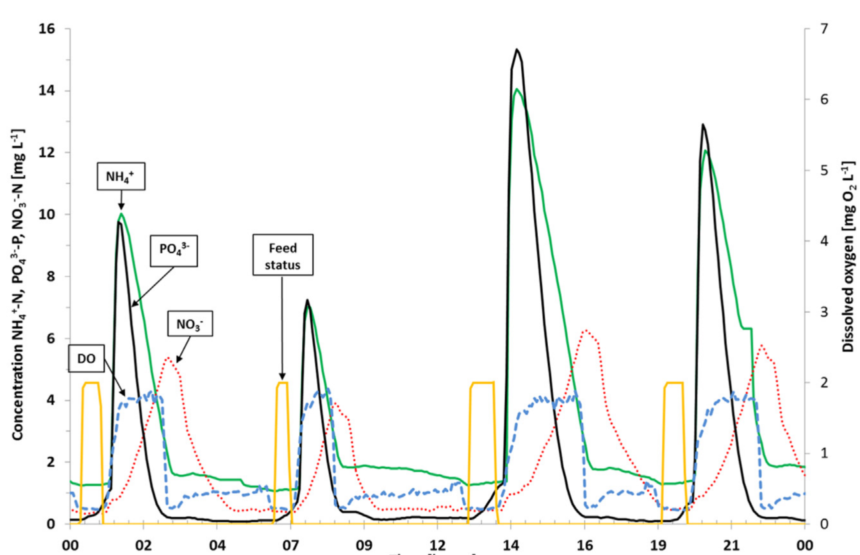

Fig. 6. Typical operational cycle in the aerobic granular sludge plant with concentration of ammonium (green solid line), phosphate (black solid line), nitrate (red dotted line), oxygen (blue dashed line) and feed (yellow solid line). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

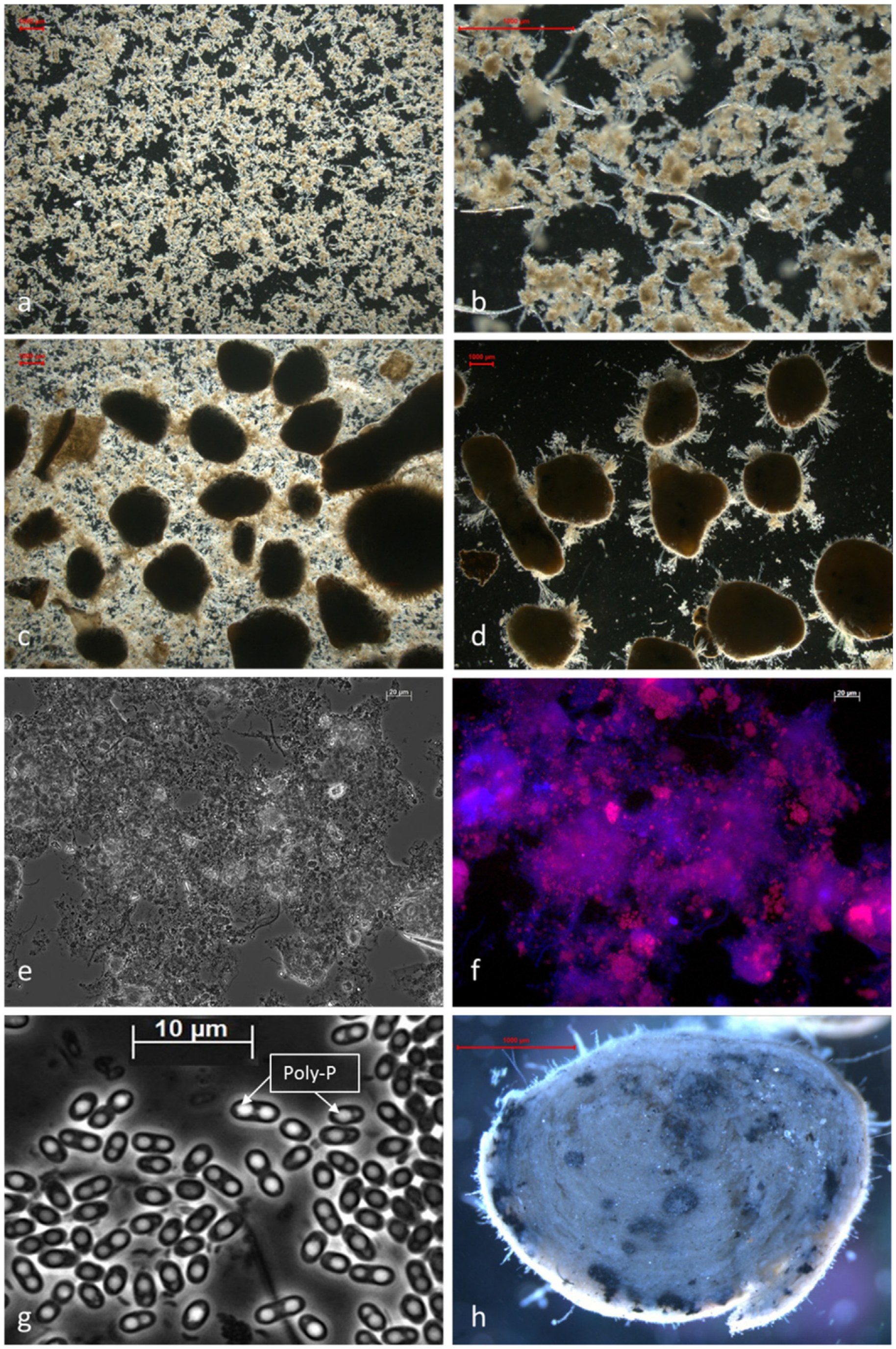

Fig. 7. Aerobic granular sludge: (a, b) biomass on the top of the granular sludge bed, (c) mixed sample of the bulk liquid, (d) granules sieved and washed with tap water, (e) phase contrast and (f) fluorescent in situ hybridisation with blue EUB (Cy5) and red PAOmix (Cy3) with pottered granules. Overlay of blue and red gives the pink colour, (g) 1000× magnification of phosphate accumulating organisms found in pottered granules and (h) sliced granule. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

disturb the plant and should be seen as a sign of a healthy microbial ecosystem in the treatment of domestic wastewater.

4.4. Nutrient removal

In sewage treatment, the nitrogen removal efficiency depends on the incorporation of nitrogen in the growing biomass, and the nitrification and denitrification capacity of the microbial community. The effluent nitrogen concentration of conventional activated sludge plants with pre-denitrification is furthermore limited by the recycle flow from the aerobic to the anoxic tank. Through post denitrification very low nitrogen concentrations can be established, but this requires the supply of external carbon. External carbon dosage is costly and can lead to elevated BOD concentrations in the effluent when overdosed (Jobbágy et al., 2008). One of the main advantages of the aerobic granular sludge plants is that the biological nitrogen removal processes are not separated in different anoxic and aerobic tank volumes, but can be optimised by a flexible aeration and denitrification period. During aeration, ammonium will be converted to nitrate and due to the inherent oxygen gradients present in the granules, a part of the nitrogen will be simultaneously denitrified. The ratio of nitrification to denitrification can therefore be regulated by changing the DO set-point. When the nitrate concentration at the end of the nitrification process is still too high for effluent discharge, a recycle from the bulk liquid at the top of the reactor towards the settled granules bed can be introduced for enhanced nitrate removal. As an alternative, an intermittent aeration strategy can be applied. The effective recycle flow rate can be much smaller than is usually used in activated sludge plants and in the present study, the average was only 0.3 of the influent flow rate to the reactors during the operational period. Compared to the AGS plant (0.3), a conventional predenitrification system would require a recycle ratio of roughly 10–15 times higher to achieve the same total nitrogen removal of over 90% (excluding nitrogen in SS effluent). In the aerobic granules, internally stored polymers of the PAO and the GAO serve as electron donor in the denitrification process. Simultaneously, the PAOs store phosphate as poly-P, hereby fulfilling a double function (denitrification and phosphate removal). The phosphate uptake is therefore less sensitive to aeration than nitrification, making the process control more flexible. As long as anoxic conditions prevail, the PAOs do not release phosphate in the bulk liquid, which results in very low nitrogen and phosphorus concentrations in the effluent.

Due to the high amount of phosphorus stored inside the sludge as poly-P, a large amount of phosphorus is released during the anaerobic periods. This is especially true during heavy rain events, when the increased flow rate is combined with concentrations that initially remain as high as during dry weather flow (so-called first flush). The resulting increased COD loading rate triggers extra phosphate release. The high hydraulic loading rate, however, enforces a short hydraulic residence time with a shorter cycle time as a consequence (Fig. 2). This reduces the time for aerobic P-uptake significantly. In the design of the Garmerwolde WWTP, this decreased biological phosphorous removal capacity under extreme rain events is tackled by iron dosing, rather by tuning the reactor size. This enables further reduction of the installed reactor volume. During normal rain – and dry weather - events metal dosing was not needed as phosphate was always removed biologically.

4.5. Volumetric conversion rates

As a result of the higher biomass concentrations (>8 g L^{-1}), the batch wise operation and the parallel occurrence of conversions that can be reached with granular sludge plants, volumetric conversion rates are higher than in conventional activated sludge plants and thus volumetric loading rates can therefore be increased accordingly. The simultaneous nitrification and denitrification in granular sludge plants, can even lead to a more than proportional increase of volumetric loading rate. The volumetric nitrogen conversion rate (based on the aerated time) of this plant was 0.058 kg N $(m^3\ d)^{-1}$. The maximum volumetric ammonium conversion rate was 0.17 kg N $(m^3\ d)^{-1}$ at an average temperature of 20 °C (Table 2). This is faster than generally found in conventional activated sludge plants. The obtained conversion rates underestimate the actual rate, since they are based on on-line ammonium measurements, while a fraction of the influent ammonium is initially adsorbed or incorporated in struvite (Bassin et al., 2011b; Lin et al., 2012). The granular sludge allows for a compact treatment process, which is reflected in the volumes of the existing old conventional plant, treating 59% of the sewage in 28,400 m^3 of activated sludge tanks plus 24,800 m^3 of clarifiers, while the AGS reactors treats 41% of the sewage in 19,200 m^3 tanks plus 4,000 m^3 of rainwater buffer. This amounts to 1.2 $m^3\ d^{-1}$ of wastewater treated per m^3 volume needed for the AGS plant compared to 0.8 $m^3\ d^{-1}$ for the existing treatment plant. Leading to a volume reduction of 33% for the AGS plant compared to the existing plant.

Effluent nutrient concentrations were also higher for the conventional AB-plant (9.9 mg N L^{-1} and 0.9 mg P L^{-1}) in the same period. It should be noted that there is no biological phosphorus removal in the AB-plant; hence most of the phosphorus is removed chemically (metal dosing).

Generally, the ammonium and phosphorus removal rates are such that both compounds can be easily removed well before the end of the cycle, thus creating room for polishing the effluent quality if desired (Fig. 6). However, in light of saving energy, effluent concentrations are steered towards the demands instead of the lowest concentration obtainable.

The oxygen gradients inside the granules, due to diffusion limitation, and the batch wise operation of the plant allow for enhanced process control via DO (Lochmatter et al., 2013; Isanta et al., 2013; Kagawa et al., 2015). An increase of the DO will lead to a larger aerobic volume fraction of the biofilm and thus an increase in volumetric nitrification rate (Aravinthan et al., 1998; Wijffels and Tramper, 1995; Beun et al., 2000b). Likewise, lowering the DO creates more anoxic volume leading to higher denitrification rates. This opens up process control opportunities that are difficult to achieve with activated sludge plants. A proper DO control strategy will therefore enable accurate steering of effluent parameters by manipulating the oxygen gradients inside the granules.

Finally, transport characteristics are less temperature dependent than bacterial growth rates (Wijffels et al., 1995). This implies that biofilm systems are less sensitive to temperature changes than suspended and flocculated systems (Leenen et al., 1997). So the effect of decreasing temperature (summer - winter) in AGS systems on the conversion rates is expected to be less than that of activated sludge systems.

4.6. Energy

We have estimated the electricity consumption of the AGS plant in the total energy balance of the Garmerwolde WWTP. The calculation includes the fraction of energy used in the influent pumping station for the AGS plant influent, but dewatering of the biomass and the energy generation by sludge digestion is not taken into account. The reason for this is that the digester (onsite) receives sludge from the AGS as well as the AB-plant plus external sludge from neighbouring plants. Determining the exact contribution of the waste aerobic granular sludge to the total biogas production is therefore impossible. The energy consumption of the full-scale AGS plant is compared to the average energy use of all Dutch WWTP's (Table 3). These are partitioned in plants with a good effluent quality (N ≈ 10 mg N L^{-1}; P ≈ 1 mg L^{-1}) and plants with an excellent effluent quality due to post treatment (N = 3–10 mg N L^{-1} and P = 0.1–0.5 mg L^{-1}) (STOWA, 2013). These data for Dutch WWTP's include energy generation by sludge digestion, which is present at most of the Dutch WWTPs. The AGS process should be compared to the higher effluent quality WWTP's. The AGS process shows very low energy consumption. Due to the lack of mixers, conventional recycle pumps, settlers and sludge return pumps, energy usage of sequencing batch systems will be lower than that of conventional activated sludge plants. Moreover, energy usage for aeration is affected by the water depth of the tanks (Fernández-Álvarez et al., 2014). The AGS tank studied had a height of 7.5 m, while activated sludge tanks are often limited to 6 m. This potentially adds 5–10 % extra to the aeration energy saving. Overall, the studied AGS plant consumed 58–63 % less energy compared to conventional activated sludge plants.

Table 3
Comparison between the average energy demand between various WWTP's in the Netherlands.

Type	Specific energy demand [kWh ($PE_{150,\ removed}$ year)$^{-1}$]
WWTP with polishing (Total N effluent < 10 mg N L^{-1})	37.5
WWTP without polishing (Total N effluent $\geq$ 10 mg N L^{-1})	33.4
WWTP Nereda® Garmerwolde	13.9

When comparing the energy usage of the existing conventional AB-plant (0.33 kWh m^{-3}) with the AGS plant (0.17 kWh m^{-3}) at WWTP Garmerwolde over the same period per m^3 of treated influent, the AB-plant consumed 51% more energy. To make a proper comparison between the two plants the energy required for the sludge treatment and the SHARON plant were omitted in both energy calculations.

Due to the low energy use of the AGS plant, the concept contributes to the development of energy neutral or energy producing WWTP's. In addition, preliminary results showed that the total biomethane potential of aerobic granular sludge during digestion is similar to the digestion of activated sludge (data not shown), but more detailed studies on the digestion and biogas production of aerobic granules grown on sewage are needed to be able to give a detailed energy balance over the plant (Val Del Río et al., 2014).

5. Conclusion

The present study described the start-up and operation of one of the currently largest full scale aerobic granular sludge plants treating domestic sewage. The results illustrate that the technology has grown in a decade from a promising innovation to a mature technology capable of competing with established wastewater treatment technologies. The operational data show that in Dutch climate conditions the effluent requirements (7 mg N L^{-1} and 1 mg P L^{-1}) were easily reached, while maintaining a high granular biomass concentration (>8 g L^{-1}) with SVI_5 values of 45 mL g^{-1} and process stability for both summer and winter periods. Because of the high biomass concentration, the volumetric loading rate could be increased accordingly. Energy usage of the aerobic granular sludge system was 13.9 kWh ($PE_{150}\cdot$year)$^{-1}$. This is 58–63 % lower than the average conventional activated sludge plant in the Netherlands, with an effluent quality which is comparable or better. The specific volume that was needed for the aerobic granular sludge based system was 33% lower than the volume needed for the existing conventional activated sludge plant. Finally, the results show that aerobic granular sludge technology can be effectively implemented for the treatment domestic wastewater. Further enhancement, mainly based on improved process control, is still possible. The full operational window with respect to influent variations has still to be established.

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