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11

Aerobic granular sludge

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11.1 INTRODUCTION

The aerobic granular sludge process is a wastewater treatment process developed to allow higher sludge concentrations and higher energy efficiency than conventional activated sludge (CAS) processes (Figure 11.1). Conventional wastewater treatment based on the activated sludge process requires large volumes due to the relatively low concentrations of sludge that can be maintained (3-5 g/l). This concentration is limited by the clarification process, which relies on gravity to separate the flocculent sludge from the treated wastewater. Conventional wastewater treatment plants therefore need large secondary clarifiers; the limited sludge loading rates that can be applied result in the large surface area of the clarifiers (Chapter 12).

Several options for designing more compact wastewater treatment plants have emerged over time. These rely mostly on immobilising the biomass on carriers as biofilms (Chapter 18) or by keeping sludge in the reactor by membrane separation in membrane bioreactors (see Chapter 13). Biofilm processes are usually limited by the amount of biofilm mass transfer area and need post-treatment to remove suspended solids, making the reduction in footprint limited. Membrane bioreactors require a relatively high level of investment and extra energy for the membrane separation process. An alternative option is to create sludge with a low sludge volume index (20-70 ml/g) that has a granular morphology, without the need for a carrier. Although this was developed in the early 1970s for anaerobic wastewater treatment (Chapter 16), granular sludge processes have only become available in the early 2000s for aerobic biological nutrient removal processes.

In aerobic granular sludge (AGS), the morphology of the biomass allows for a high settling velocity of the sludge. Flocs are in general small and have a high drag coefficient, whereas granules generally have a larger radius and lower drag coefficient (Figure 11.2). The resulting higher bed settling velocities of aerobic granules (4-10 m/h) compared to activated sludge flocs (0.8-1.4 m/h) allow for the integration of the settler in the treatment reactor and thus a compact reactor design. This is further enhanced by the discrete settling properties of a granular sludge bed. Biomass concentrations of more than 10 g/l are easily achieved,

however in practice lower values suffice since the design of wastewater treatment plants is often limited by other constraints of the plant (Chapter 5).

Figure 11.1 Nereda® wastewater treatment plants in the Netherlands in (A) Garmerwolde (140,000 P.E.) and (B) Utrecht (430,000 P.E.). In Utrecht the new 6-tank Nereda® facility (in operation since 2019) is visible on the left with part of the old abandoned plant on the right.

Since the emergence of anaerobic granular sludge in the 1970s, granulation has mostly been considered to be related to the specific microbiology of methanogenic processes. It was thought that the complex community structures required to convert substrates anaerobically into methane, and the crucial role of interspecies hydrogen transport, form the evolutionary driver for anaerobic organisms to grow in compact aggregates or granules. However, trials to form aerobic granular sludge failed and it was assumed this was because aerobic conversion of substrates in general do not depend on syntrophic interactions.

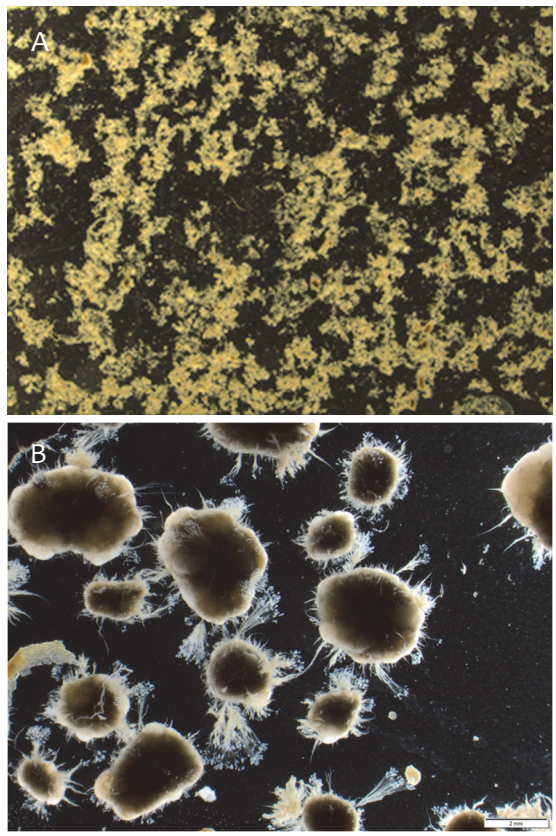

Figure 11.2 (A) Activated sludge from WWTP Harnaschpolder, the Netherlands, and (B) washed aerobic granules with stalked ciliates from the Nereda® plant in Utrecht, the Netherlands.

Due to the introduction of taxes on discharging wastewater in 1970 in the Netherlands, industry started to find compact solutions for treating their wastewater. This led to the Upflow Anaerobic Sludge Bed Reactor (UASB) (Lettinga *et al,* 1975), which was quickly introduced especially in the food industry. During the 1980s the company Gist-Brocades also started to develop a compact wastewater treatment system (Heijnen, 1984; Heijnen *et al.*, 1990). For the design of compact reactors,

without the need to rely on granular sludge formation, they developed anaerobic and aerobic treatment technology using biofilms on small suspended carriers (sand grains). This led to well-settling particles with a high specific biofilm surface area. This anaerobic technology rapidly developed into the now widely used expanded granular sludge bed (EGSB) and internal circulation (IC) reactors (Chapter 16). Both rely on granular sludge biofilm processes. For aerobic processes the formation of granules proved more complex and the CIRCOX® reactor technology became the standard compact aerobic technology for industrial wastewater (Heijnen *et al.,* 1993). However, a translation of these technologies for use in municipal wastewater treatment did not occur since the reactor technologies were not suited to handle the large hydraulic variations occurring in municipal wastewater flows.

Research in the 1990s into the morphogenesis of biofilms and flocs (especially for the CIRCOX® reactor development) resulted in the postulation of a general hypothesis. This hypothesis stated that the ratio between biofilm surface loading (or the rate at which new biomass is produced) and shear rate determines the biofilm structure. When shear forces are relatively high, only a patchy biofilm will develop, whereas at low shear rates the biofilm becomes highly heterogeneous with many pores and protuberances. With a correct balance between biofilm surface loading and shear rate, smooth and stable biofilms can be obtained (Van Loosdrecht *et al.,* 1995). This was further evaluated by mathematical models (Van Loosdrecht *et al.,* 2002; Picioreanu *et al.,* 1998). A similar approach could be used in the context of predicting the Sludge Volume Index (SVI) (Martins *et al.,* 2004). From this research it was concluded that slower growing bacteria (methanogens, anammox and nitrifying bacteria) will form granules more naturally than fast-growing bacteria such as aerobic heterotrophs and fermentative bacteria.

The first proof of principle for aerobic granulation was reported by Heijnen and Van Loosdrecht (1998) and a patent was applied for. The granules were grown aerobically on molasses in a sequencing batch reactor (SBR). By applying a critical settling rate of 35 m/h, granulation could be obtained. Fast-settling granules remained in the system while the slow-settling flocs were washed out. However, although granules were formed initially, stable long-term operation was found to be problematic (Morgenroth *et al.,* 1997). The reason for this was not clear at the time. Nevertheless, the follow-up research led to the development of full-scale aerobic granular sludge technology. The specifics are further discussed in this chapter.

The translation from lab-scale observations to commercial full-scale AGS reactors took approximately 12 years. A public-private consortium formed by Delft University of Technology, several Dutch water authorities, the consultancy company Royal HaskoningDHV, supported by national and international innovation programs, was responsible for the development of aerobic granular sludge technology in the Netherlands. This new technology was registered as a trademark under the name Nereda® by Royal HaskoningDHV. The technology began to be applied for industrial wastewater treatment in 2005, and the first demonstration plant for municipal wastewater was designed and constructed in Gansbaai, South Africa, between 2006 and 2008. In 2010 the first commercially implemented full-scale aerobic granular sludge reactor treating domestic (65%) and industrial (35%) (slaughterhouse) wastewater was constructed in Epe, the Netherlands. Another important milestone in the technology development was in 2013 at the Garmerwolde WWTP, the Netherlands, when Nereda® reactors were installed that ranked among the world's largest SBR reactors, establishing it as a fully scaled-up technology. The Garmerwolde Nereda® reactor design and performance is described in detail in Pronk *et al.,* 2015). This technology is currently spreading throughout the world and by 2020 had been adopted by more than 80 plants ranging from 5,000 P.E. to 2,400,000 P.E. (Figure 11.3).

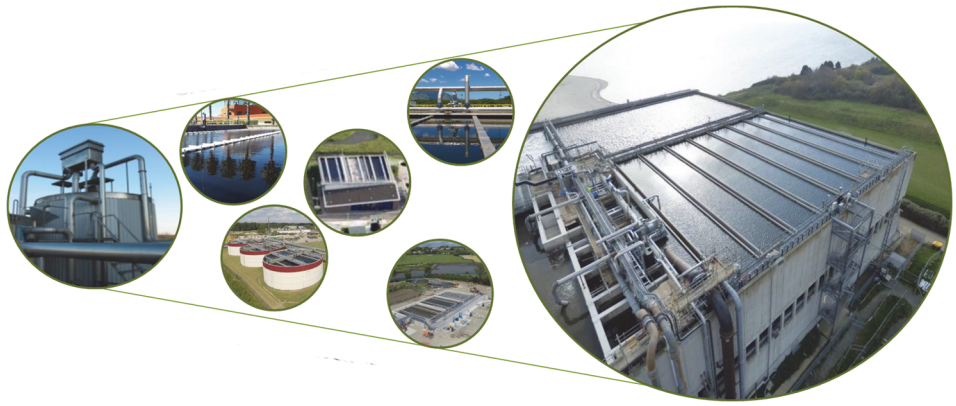

Figure 11.3 Nereda® plants; from Vika, the Netherlands (5,000 P.E.) to Ringsend in Dublin, Ireland (2,400,000 P.E.).

11.2 IMPORTANT CONSIDERATIONS FOR SELECTING AEROBIC GRANULAR SLUDGE

11.2.1 Gradients

Substrate gradients are an important parameter that controls the morphology of flocs and biofilms (Chapter 10). Due to simultaneous substrate consumption and diffusion, different concentrations will occur as a function of the depth in the floc or biofilm. To support large granules, the whole interior needs to receive substrate. If the substrate does not diffuse throughout the interior, the structure will eventually deteriorate due to decay causing unstable granulation. If the substrate uptake rate is lower than the diffusion of the substrate into the granule, it will lead to more uniform bacterial growth throughout. When the substrate uptake rate is higher than the diffusion rate, the substrate only penetrates the outer part of the granule or even only the tips of the protrusions. This relation can be elegantly expressed with the dimensionless factor *G* (growth), which was developed by Picioreanu *et al.* (1998) for planar biofilms. It shows the ratio between maximum biomass growth rate and the maximum substrate transport rate through diffusion.

$$G = L_Y^2 \cdot \frac{\mu_m \cdot C_{x,m}}{D_s \cdot C_{s,o}} \qquad (11.1)$$

where:

$C_{s,o}$	concentration of the substrate in the bulk (g/m^3)
D_s	diffusion coefficient of that substrate (m^2/d)
$C_{x,m}$	maximum density of the biomass in the biofilm (g/m^3)
μ_m	maximum specific growth rate (1/d)
L_y	biofilm thickness (m)

High G values will lead to finger-type outgrowths, rougher biofilms and thus slower settling rates. It will also cause unstable biofilm or granule formation. In the case of flocculent sludge it leads to bulking sludge. On the other hand, low G values will result in stable, dense biofilms or granules.

One of the reasons for the difficulty in obtaining aerobic granular sludge is the low solubility of oxygen. Oxygen diffusion into the biofilm is usually rate-limiting for most conversions. For example, at an oxygen concentration of 2 mgO_2/l, oxygen is rate-limiting for nitrification unless the ammonium concentration is less than 0.44 mgN/l (Chapter 17). Typically, at these concentrations the penetration depth of oxygen is not more than 20 to 40 µm.

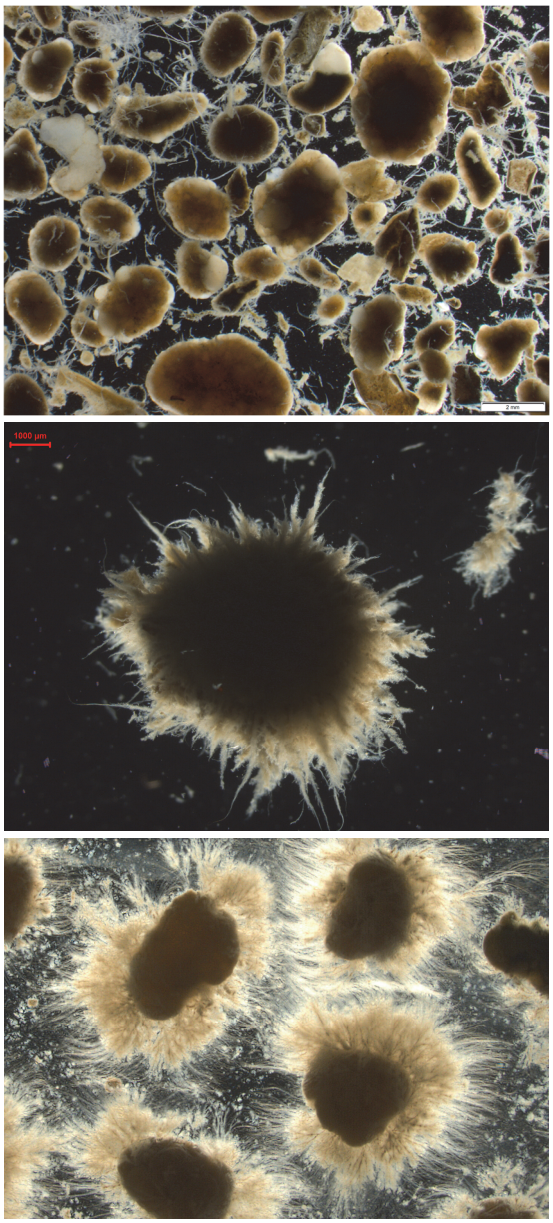

Figure 11.4 Aerobic granules with (top) smooth morphology, (middle) with finger-type outgrowth and (below) filamentous outgrowth.

Low concentrations of readily degradable COD (*i.e.* acetate, glucose etc.), *e.g.* when substrate is supplied to a completely mixed reactor, will lead to finger-type outgrowth on the outside of granules or biofilms. In granular sludge systems such conditions will even lead to filamentous outgrowth when also the oxygen transport rate to the biofilm is limiting (Figure 11.4). This dual oxygen and COD limitation has a negative impact on the settling rate, stability and process performance. This problem has been overcome in Nereda® technology by applying an anaerobic feed of sewage allowing full uptake of readily degradable COD before aeration starts.

Note the G factor (Eq. 11.1) assumes zero-order processes while some conversions will occur at concentrations below the cellular substrate affinity coefficients (usually around 10-100 µg/l). Nevertheless, the G factor gives valuable insight into the granulation process.

11.2.2 Microbial selection

Cultivation and stability of aerobic granular sludge under low oxygen concentrations is vital for the full-scale application of aerobic granular sludge. A low dissolved-oxygen concentration is needed for minimising aeration energy as well as for efficient nitrogen removal by allowing simultaneous nitrification and denitrification. Incorporating an anaerobic feed in the process leads to readily biodegradable COD (RBCOD) to be converted into storage polymers (mainly polyhydroxyalkanoates (PHA)). This is then followed by an aerobic period where growth on the storage polymers occurs, effectively bypassing the mechanism expressed by the G factor via separation of substrate uptake and biomass growth. This makes the granule formation less dependent on the oxygen concentration. In the anaerobic period phosphate-accumulating (PAO) or glycogen-accumulating organisms (GAO) will store the RBCOD as PHA. Since substrate concentrations will be much higher under anaerobic conditions than those in the presence of oxygen the RBCOD is stored over a significant depth. In the aerated phase the inner layers either grow on nitrate (denitrification) or get oxygen later in the process when the storage pools of the organisms in the outer layers are depleted. There is an extra advantage in the conversion of RBCOD

into storage polymers (PHA). Bacteria growing on storage polymers have a lower growth rate (μ_{max}) than heterotrophic bacteria that grow directly on RBCOD in the presence of oxygen (Van Loosdrecht *et al.,* 1997). In general, slower-growing bacteria grow in more dense structures than fast-growing bacteria. This growth in dense structures is an extra reason why slow-growing bacteria (methanogens, anammox, nitrifiers, PAO) generate smooth and compact granular sludge. The distribution of substrate combined with slow growth was also found to be the key mechanism for stable, smooth biofilm formation (Picioreanu *et al.,* 1998).

11.2.3 Physical selection

In practice sewage is a complex mixture of substrates. It consists of both dissolved and particulate matter (Chapter 3). The soluble compounds can diffuse into the granular sludge, whereas the particulate matter first has to be converted through hydrolysis. The non-biodegradable particulate matter fraction will accumulate in the reactor. For aerobic granular sludge one could classify the COD into granular, sludge-forming COD, and non-granular, sludge-forming COD. The non-granular sludge-forming COD consists of inert material and COD that is not convertible into PHA anaerobically and eventually leads to floc formation. As a result, granular sludge reactors always contain a fraction of flocculent material consisting of inert particulate COD, biomass grown on the remaining COD after the anaerobic period, and eroded material from the granular sludge. This poor settling fraction needs to be removed to form and maintain the granular fraction in the AGS plant. To remove the slow-settling sludge and at the same time retain granular sludge, selection is applied. To do this the top of the sludge bed is removed after a short settling period. The short settling phase allows fast-settling granules to be retained in the reactor while flocculent material residing at the top of the sludge blanket is spilled. This results in a solid retention time of 0.5-5 days for the flocculent fraction and around 30 days for the granular fraction. More in-depth discussion on settling and its associated parameters can be found in Section 11.3.4.

11.2.4 Shear

Shear can counterbalance the tendency of fast-growing bacteria to form filamentous outgrowth. For instance, for the CIRCOX® reactor mentioned in the introduction, shear is a dominant factor in obtaining smooth biofilms in a reactor which is fed continuously under aerobic conditions (Kwok *et al.,* 1998). When slow-growing bacteria (*i.e.* all the RBCOD is converted anaerobically into PHA) are properly selected, shear plays no significant role in the granule formation process. Also, in practice applying shear would mean extra energy dissipation and costs and is therefore neither desirable nor a generally required parameter for AGS.

11.2.5 Plug-flow feeding

All these aspects described above have led to the operational strategy for granular biomass formation with sewage using the Nereda® process (Figure 11.11). To minimize the substrate diffusion limitation and ensure the best uptake of substrate over the depth of the granules, the influent should not be diluted before contact with the granules. In AGS reactors, this is achieved by feeding the influent through the settled granular sludge bed from the bottom (Figure 11.5) in the absence of aeration.

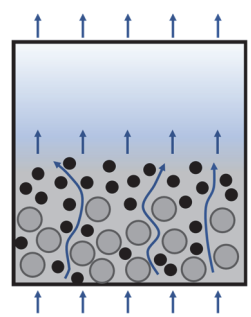

Figure 11.5 Graphical presentation of anaerobic plug-flow feeding through a bed of settled granules in an aerobic granular sludge reactor.

The plug-flow regime ensures a relatively high substrate concentration through the settled sludge bed. Combined with a sufficiently long anaerobic feeding

period, the substrate can penetrate deep into the granule. Therefore, growth can take place over the depth of the granular biomass and not primarily at the granule surface. This mode of operation has a self-stabilising effect: larger granules will tend to be at the bottom of the sludge bed and get thereby most of the RBCOD, while none or very limited RBCOD is left for the floc fraction at the top of the settled sludge bed.

11.2.6 Effect of substrate and feeding regime on granule morphology

The applied feeding regime of AGS reactors with different substrates will have a significant impact on the granule morphology and thus on the stability of the system in general. In practice sewage is of course a complex mixture of substrates. Figure 11.6 (A-D) demonstrates this complexity and the impact of various feeding strategies in SBR systems on the formation, morphology and stability of aerobic granular sludge.

(A) Readily biodegradable soluble substrates when fed anaerobically are taken up by PAO or GAO types of bacteria and converted into storage polymers. In a subsequent aerobic period these storage polymers are used for growth at a relatively slow rate. This results in compact granular sludge. Compounds that are not converted anaerobically (*e.g.* butanol, propanol) but are adsorbed in the granular sludge matrix will be converted in the aerobic phase. This also leads to growth throughout the granule and thereby to stable granule formation. This anaerobic feeding strategy not only ensures stable granulation, but also ensures optimal phosphate and nitrogen removal which is important for the treatment of domestic sewage.

(B) Readily biodegradable soluble substrates remaining in the liquid after the anaerobic phase or dosed as a pulse in a fully aerobic granular sludge reactor will lead to substrate uptake under oxygen diffusion limitation. The substrate is used for simultaneous growth and formation of storage polymers and is mainly limited to outside areas of the granule, while the inner regions are deprived of oxygen (Beun *et al.,* 2002). The fast consumption of easy biodegradable substrates in the presence of oxygen on the outside fraction of the granule will lead to formation of finger-type or filamentous outgrowth. This will depend on the oxygen concentration in the bulk and increased shear is needed to ensure smooth and stable granulation (Beun *et al.,* 1999). Granules formed under this regime are more prone to breaking under shear stress since the inner regions are inactive and will eventually decay and weaken the granule (Beun *et al.,* 2002). This leads to unstable granulation and poor settling characteristics combined with higher suspended solids (flocs and loose cells) in the liquid after fast settling. Besides this, the nitrification and phosphorus removal potential are also decreased. The slow-growing nitrifying bacteria will be overgrown and pushed down to oxygen limited layers by the faster-growing heterotrophs (Elenter *et al.,* 2007; Gonenc and Harremoes, 1990). Some substrates also lead to good granulation even if they are converted aerobically. Ammonium and methanol are such substrates. Both these substrates are converted with oxygen by relatively slow-growing bacteria, which leads to a denser biofilm formation (Mosquera-Corral *et al.,* 2003; Villaseñor *et al.,* 2000). In AGS systems, substrates that are converted aerobically by slow-growing bacteria are therefore generally expected to lead to stable granulation.

(C) Particulate substrates (*e.g.* starch, proteins) present another challenge, because of the need for hydrolysis. Particulate substrates are mainly hydrolysed at the surface of the granules (De Kreuk *et al.,* 2010). The hydrolysis products (VFA) will thereafter become available for conversion into storage polymers by PAO and GAO-like organisms. This results in stable granulation. Depending on the anaerobic hydrolysis rate, some aerobic hydrolysis will also occur. Under aerobic conditions the hydrolysis products will be directly used for growth by the organisms present at the granule's surface. This creates steep substrate diffusion limitation gradients, which will induce finger-type outgrowth, less stable granule formation and higher levels of suspended solids in the liquid phase.

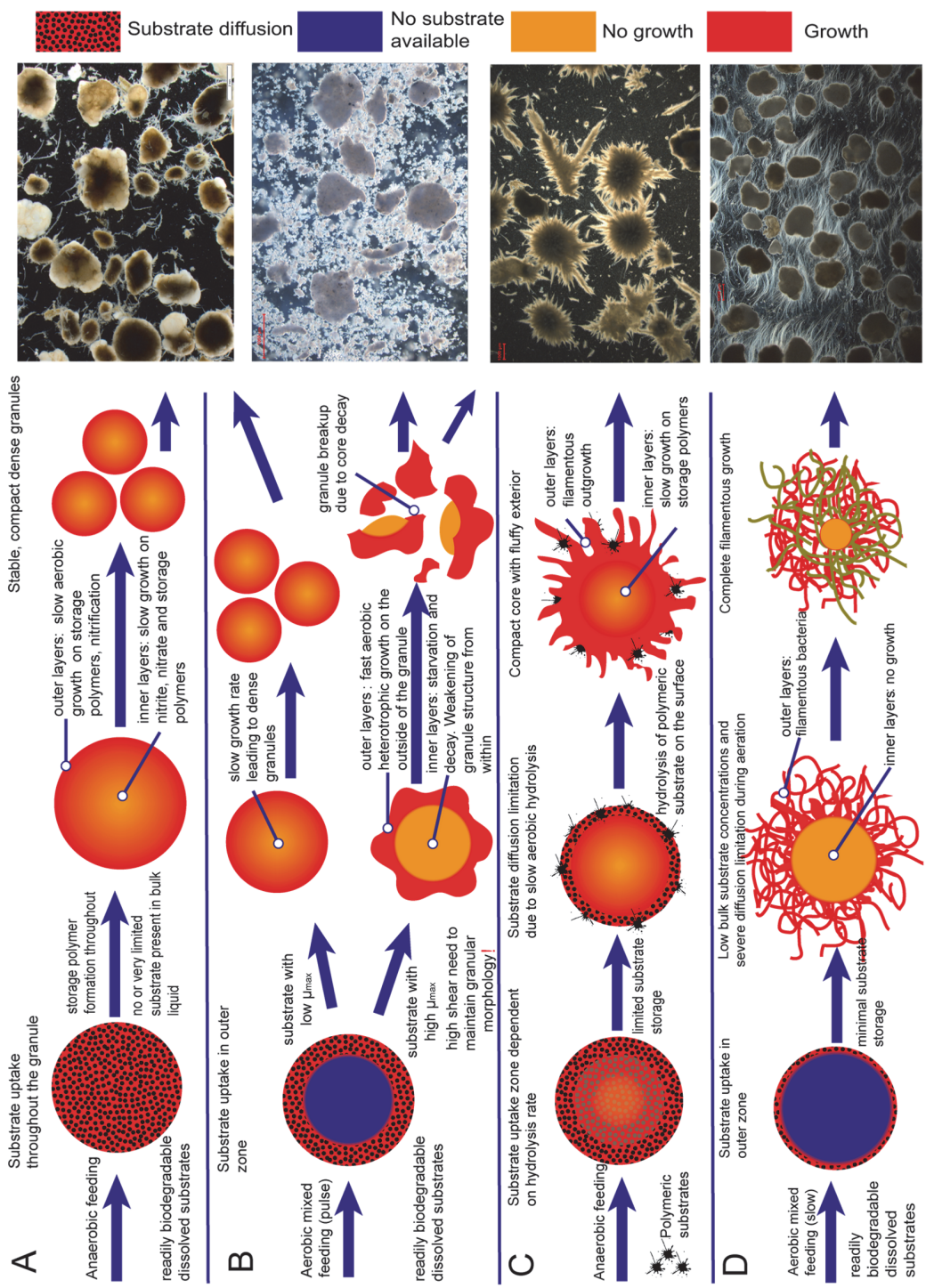

Figure 11.6 Effect on aerobic granule formation with different carbon sources and feeding regimes in sequencing batch reactors selecting for aerobic granular sludge; (A) Readily biodegradable dissolved substrates fed anaerobically, (B) aerobic feeding of readily biodegradable dissolved substrates into a mixed reactor, (C) anaerobic feeding of polymeric substrates through a settled granular bed, (D) slow aerobic feeding of readily biodegradable dissolved substrates in a mixed reactor.

(D) Readily-biodegradable substrates fed slowly in a mixed aerobic environment will lead to severe substrate diffusion limitation gradients. The substrate concentration in the bulk is very low or most likely zero. This provides very good conditions for the proliferation of filamentous organisms as they have the advantage of growth direction and available surface area (Martins *et al.,* 2003; Martins *et al.,* 2011). Aerobic granular sludge fed under these conditions will therefore quickly deteriorate. Breakage of the granules will occur as the inside will not receive any substrate and will die. Filamentous growth will have a detrimental effect on the settling properties of the granules and thus on the effluent quality. Granulation formation is thus unlikely. High shear will help control the outgrowth, but this is not always an option.

11.3 KINETICS OF AEROBIC GRANULAR SLUDGE

11.3.1 Carbon removal

In AGS carbon removal is not much different from normal biological phosphate-removal plants. Overall the same bacteria are active in activated sludge as in granular sludge. A major difference is the distribution of SRT in granular sludge plants. In activated sludge plants the sludge flocs all have the same SRT. Although the average SRT in granular sludge plants is similar to that of activated sludge plants, the SRT of granules is larger than the SRT of the flocculent sludge fraction as shown by Ali *et al*. (2019). Large granules consist of PAO and several other bacterial groups growing on soluble and readily hydrolysable substrates and can have an SRT larger than 30 days. However, the flocculent sludge fraction also contains inert particulate compounds, eroded granular sludge material and slowly-hydrolysable particulates, and has an SRT of 0.5-5.0 days, which is much lower than the SRT of the granular fraction. The composition of the flocculent fraction is comparable to sludge of highly-loaded activated sludge plants. It is not mineralised and therefore gives a relatively high methane yield.

11.3.2 Nitrogen removal

Nitrification is performed by nitrifiers which are present in both the granules and flocs. Due to the difference in SRT between granules and flocs, nitrification can easily be maintained even at cold temperatures. In activated sludge plants the nitrifying activity of sludge decreases due to two factors: the intrinsic effect of temperature on the activity of microorganisms (halving the rate for every 8-10 °C temperature decrease) as well as an accumulation of more suspended COD in the sludge due to decreased hydrolytic rates. However, for granular sludge the effect of temperature is much less as the reduced microbial activity is compensated for by a larger penetration depth of oxygen into the granular sludge. Furthermore due to the preferential removal of suspended COD in an AGS plant (by the selective sludge wasting of the flocculent fraction), the nitrifying fraction in the sludge is not diluted by it. Therefore overall, the sludge-specific nitrification rates for granular sludge are much less affected by temperature (Figure 11.7).

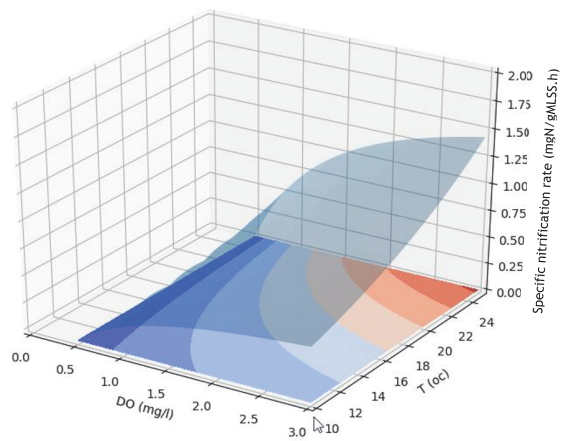

Figure 11.7 General relation of temperature and dissolved oxygen concentration on the volumetric nitrification rate in full-scale aerobic granular sludge reactors.

In sufficiently large granules the oxygen concentration in the inner parts of the granule becomes zero, while the oxygen concentration near the outer layer of the granule is high enough for nitrification. Because of this, nitrification and denitrification can occur simultaneously within the granule**.** The dissolved oxygen concentration needed for optimal simultaneous nitrification and denitrification is directly related to the size and activity of the granules. Larger granules have more anoxic volume than smaller granules at the same dissolved oxygen concentration in the bulk liquid (Figure 11.8). At lower temperatures, because of lower activity, the penetration depth of oxygen increases. This decreases the volume for denitrification, effectively making it more difficult to maintain simultaneous nitrification-denitrification.

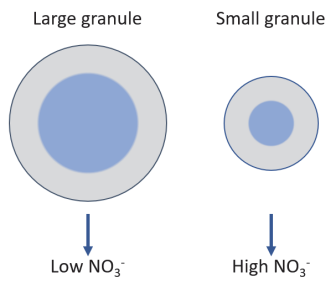

Figure 11.8 Effect of oxygen concentration in the bulk liquid on the available denitrification volume inside granular sludge. The grey zone indicates the oxygen penetration depth. The blue is the anoxic zone.

The efficiency of simultaneous nitrification and denitrification also depends on the amount of storage polymers stored in the deeper parts of the granules. In wastewater with relatively low RBCOD concentrations, denitrifiers will depend more on the slowly-degradable COD or storage polymers present in the outer layer of the granular sludge. In these cases, denitrification can be enhanced by alternating aerobic/anoxic conditions *i.e.* by switching aeration on and off.

11.3.3 Biological phosphorus removal

Biological phosphorus removal in AGS is performed by the same organisms found in conventional activated sludge (CAS) processes. In the AGS plant the growth conditions for PAOs are intrinsically optimal, since stable granulation depends on the effective and full consumption of COD under anaerobic conditions as well.

The phosphorus present in the influent is removed by biomass growth and via storage of polyphosphate in the PAOs. Under anaerobic conditions the PAOs take up VFAs forming PHA while hydrolysing polyphosphate for energy production (Chapter 6). This leads to phosphate release into the bulk liquid. The duration of the feeding phase is defined by the hydraulic need to take in wastewater and typically lasts 0.5-1 hour. In sewer systems where there is significant hydrolysis and fermentation occurring, for example in systems with long pressure pipelines, this time is long enough for complete uptake of the substrates. In sewer systems with less hydrolysis and fermentation in the sewer, for example in a gravity flow system with short pipelines, a prolonged anaerobic period might be needed to convert hydrolysable COD into VFAs to allow growth of PAOs.

In the aerobic phase following the anaerobic feeding phase, the reactor is aerated to allow for growth of the bacteria. Oxygen diffuses from the bulk liquid into the granules, allowing the PAOs and GAOs to start consuming the anaerobically stored PHA for growth. The PAOs will replenish the polyphosphate pool removing the phosphate from the bulk liquid. In the deeper layers of the granules, the PAO will be active using nitrate or nitrite instead of oxygen as the electron acceptor. This makes the phosphate uptake rate less susceptible to changes in the oxygen concentration than nitrification.

11.3.4 Granular sludge properties

The terminal settling velocity of an aerobic sludge granule is mainly dependent on the diameter and can have values up to 100 m/h (Figure 11.9).

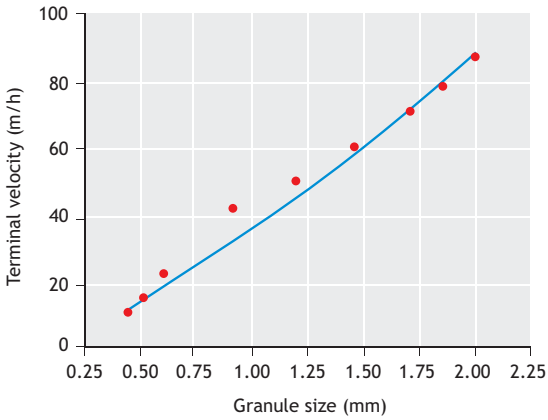

Figure 11.9 Terminal settling velocity of individual granules from the full-scale Nereda® plant in Garmerwolde, the Netherlands (the line with $R^2 = 0.98$ is calculated by a model).

A standard 30-minute measurement of the sludge volume index (SVI) therefore will give limited information about the settling behaviour of granular sludge, because in 30 minutes the granules will already have finished settling for a long time. It is therefore common practice for granular sludge to be measured after 5 minutes (SVI_5) and after 30 minutes (SVI_{30}) of settling. If the SVI_5 approximates the SVI_{30}, this is an indication of a high degree of granulation. The SVI of a mature granular bed typically is in the range between 20 and 40 ml/g. One needs to keep in mind that the SVI measured is a maximum value dictated by the worst-settling fraction present. A small fraction of slower-settling flocs in a traditional SVI test largely determines the SVI. Subsequently the SVI is not a very useful parameter to monitor quality of granular sludge. However, the SVI is important to monitor the amount of biomass that can be maintained without *e.g.* during feeding expanding towards the effluent decant. Another metric for the characterization of aerobic granular sludge is the granule size distribution. A sludge sample is sieved in a series of sieves (between 200 and 2,000 µm). Then the biomass concentration of the sieve fractions is measured. Since a granule is defined as an aggregate larger than 200 µm, all sludge smaller than 200 µm is considered non-granular. Note that this threshold is relatively arbitrary and the 'non-granular' fraction next to flocculent sludge also contains smaller 'baby granules'. A normal value for the flocculent fraction is between 1 and 2 g/l in a full-scale AGS reactor. At higher MLSS concentrations the contribution of the fraction larger than 1 mm increases and in some cases this fraction can compose up to 90% of the total biomass concentration in the reactor.

11.3.5 Reactor operation aspects

Current aerobic granular sludge technologies depend on a sequencing batch operation. This has several advantages. The main aspect is the promotion of stable granulation by direct contact of granular sludge and raw wastewater at high concentrations. Also, granules are preferentially fed over flocs and RBCOD is fully converted under the anaerobic conditions.

This batch-wise operation also allows for lower effluent concentrations. The effluent concentrations in continuous processes are equal to the concentrations in the aeration tank. For low effluent requirements this can lead to decreased conversion rates at low operating concentrations. In a sequencing batch operation the aeration phase starts with high concentrations reaching high conversion rates. This means that substrate concentrations during most of the reaction phase are high enough not to be limiting the conversion rate. Thus, the conversion rates only drop at very low substrate concentrations. Because of this, effluent concentrations can reach virtually zero without lowering the overall conversion rates.

Batch operation also enables a continuous monitoring of the kinetics of the granular sludge. Figure 11.10 gives the online monitored concentrations of ammonium, nitrate and phosphate during an operational day in a Nereda® plant. During influent feeding (Q_{inf}) the concentrations (measured at the top of the reactor) are equal to the effluent

concentrations. When feeding stops and aeration starts the reactor gets completely mixed leading to a sharp increase in ammonium and phosphate concentrations and decrease in the nitrate. These changes are proportional to the volume exchange rate of the SBR operation. The subsequent decrease in ammonium and phosphate immediately give the rate of conversion at the prevailing oxygen concentration which can be used for process monitoring and control purposes.

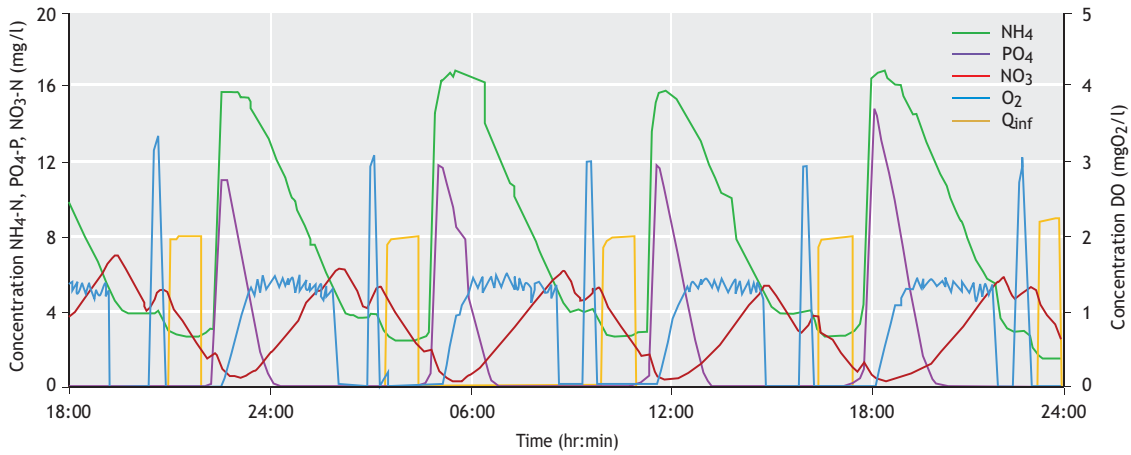

Figure 11.10 Online monitored concentration of ammonium, phosphate, nitrate, oxygen and influent flow rate during an operational day in a Nereda® plant.

In the reaction phase, the NH_4^+ is converted into NO_2^+, NO_3^- and N_2. An advantage of the AGS process is the simultaneous nitrification and denitrification. When oxygen is present in the water phase, NH_4^+ will be converted into NO_2^- by the ammonia-oxidizing bacteria (AOB). Because of the anoxic conditions within the core of the granule, and the presence of PHA stored by the PAO/GAO population, part of the NO_2^- will be converted directly into N_2 gas. The rest of the NO_2^- will be converted into NO_3^- by the nitrite-oxidizing bacteria (NOB). Similar to the NO_2^-, the NO_3^- can also be denitrified in the anoxic core of the granule. The amount of simultaneous denitrification depends on many process conditions, such as temperature, dissolved oxygen concentration in the reactor, and the amount of VFAs in the influent. In practice, this value can vary between 20% and 80%. If enhanced denitrification is required to achieve the treatment targets, simultaneous denitrification can be introduced by including additional denitrification periods (*i.e.* by switching on and off aeration during the reaction phase).

11.4 PROCESS CONTROL

11.4.1 The Nereda® cycle

The aerobic granular sludge process is a sequencing batch process. A reactor goes through a series of steps to clean the wastewater. These consecutive steps (simultaneous feeding from the bottom and drawing effluent from the top, the aeration phase and a short settling period) are called the Nereda® cycle (Figure 11.11).

The steps, or phases, in a Nereda® cycle are not fixed. The length of the different phases can be adapted according to the process circumstances. Also, the total cycle time can be changed by the process control software. More importantly, detailed online

monitoring (Figure 11.10) enables one to follow the progress of the biological conversion. This can be used to adjust the phases and process set-points. Therefore the AGS process is highly adaptable if compared to a conventional continuously-fed activated sludge plant. The AGS process is controlled to meet effluent requirements for COD, N and P, to limit the amounts of suspended solids in the effluent, to optimize the SRT and to manage the variations in the influent flow. In the following paragraphs the most common approaches are shown.

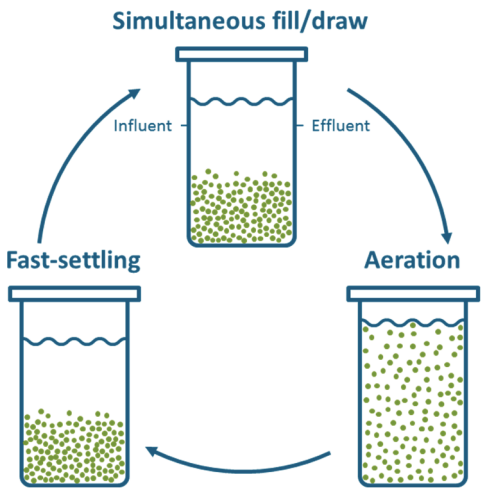

Figure 11.11 The Nereda® cycle

In the first step the influent is fed from the bottom through a settled bed into the reactor, while the cleaned water from the previous batch is pushed out at the top. Due to the good settling properties of aerobic granular sludge this can be done at relatively high up-flow velocities, without the risk of spilling sludge into the effluent. Up-flow velocities in the range of 5 m/h are not uncommon for an AGS reactor.

During the aeration phase the reactor is aerated for conversion of COD and nutrients. Also, anoxic conditions can be applied for the removal of nitrite and nitrate. In practice aeration is controlled based on a series of different strategies, such as oxygen concentration, redox, pH, ammonium, phosphate and NO_x concentrations.

The settling phase has two purposes. Before starting to feed again in the next cycle, the sludge bed must settle to prevent the washout of sludge when the simultaneous feeding and decanting starts. Granules will settle faster than flocs, which form a layer on top of the granular bed. The waste sludge is taken from the top of the sludge blanket at the end of the settling phase. Here the granular sludge is favoured above the flocculent sludge, which is an important step in the granulation process (as discussed in Section 11.2.3).

11.4.2 Batch scheduling

In most wastewater treatment plants the flow towards the plant can vary greatly. It is not uncommon that during a rain event the flow to the treatment plant is 5 times higher than under dry weather conditions, while overall pollution loads often remain the same. To treat this increase in flow rate in an as compact as possible reactor, the duration of each cycle can be adapted. This process is called batch scheduling. An example of a schedule under both dry weather conditions and wet (rainy) weather conditions is given in Figure 11.12. The example shows the scheduling for a system with two reactors and an influent buffer. Under dry weather conditions the reactors run with a total cycle time of 6.5 hours, a feeding phase of 1 hour, a reaction phase of 5 hours and a settling/wasting/idle phase of 0.5 hour. The reactors are fed from the influent buffer that stores the water when one reactor is in the feeding phase.

During wet weather conditions the cycle shortens. The total cycle time is only 3 hours, but the duration of the feeding phase is increased to 90 minutes. Also, there is no longer any time between the feeding phases, so one of the reactors is always feeding. The total time available for aeration is reduced by these changes in the scheduling. This can partly be compensated by increasing the oxygen levels in the reactor. So, depending on the effluent requirements and the local hydrograph, these wet weather

conditions can be the determining factor for the reactor sizes in the design.

11.4.3 Nutrient removal

The process control for effective biological phosphorus removal is minimal. The anaerobic time is mainly decided by the hydraulic constraints of the process. In general, the feeding time is long enough to ensure enough COD uptake by PAO to allow good growth and phosphorus uptake under aerobic conditions. Only in the case of wastewater with very low soluble COD could it be necessary to control the length of the anaerobic period. In the aeration stage the phosphorus uptake is in general faster than the nitrification rate and again no specific control is required. As for the stable operation of any enhanced biological phosphorus removal (EBPR) process, it is necessary to minimize aeration after the phosphate is removed from the liquid in order to prevent reduction of PAO-activity by extensive over-aeration. Where necessary the overall biological phosphate removal can be supplemented with some chemical precipitants. This for example might be required for very strict effluent demands or in the case of low COD/P ratio in the wastewater or during periods with excessive hydraulic loading such as during peak wet weather flows. The amount of this supplementary chemical dosing can be easily optimized and controlled effectively. Since phosphorus uptake is linear after the first hour of aeration, and phosphate can be monitored online, the effluent phosphorus concentration achievable by biological removal can be accurately predicted while the treatment batch progresses. This information can be used to determine online if and how much precipitant is required and provide set points for the control of chemical dosing.

Nitrogen removal is controlled in the aeration phase. Nitrification is sensitive to the dissolved oxygen concentration. Online monitoring of ammonium gives the nitrification rate under the applied oxygen concentration. The oxygen concentration can be adjusted by changing the aeration rate. A decrease in aeration rate results in a decreased nitrification rate and increased simultaneous denitrification. With the process control the optimal ratio between both processes can be set. Since PAOs can use both oxygen and nitrate as the electron acceptor there is no big influence of aeration control on the P-removal process. If the simultaneous denitrification is insufficient, extra (post-) denitrification can be obtained by applying on/off aeration in the second phase of the cycle. The batch-wise operation and consequent direct monitoring of the concentrations give useful information for the online control of on/off aeration cycles.

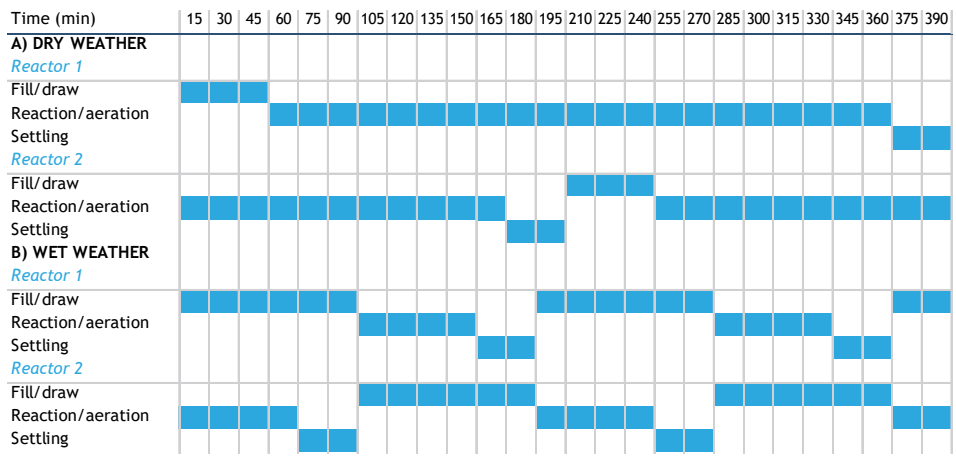

Figure 11.12 Batch scheduling for a plant with two reactors and a buffer under dry weather and wet weather conditions.

11.4.4 Effluent suspended solids

Effluent suspended solids removal performance, and measures to optimize this removal are similar to activated sludge plants. To prevent washout of small floating material with the effluent, as in clarifiers, baffles are placed in front of the fixed effluent weirs. Further, as for activated sludge clarifiers, it is important to prevent undesired denitrification resulting in an increase of effluent suspended solids. Rising sludge due to N_2 gas production is a known problem in clarifiers in conventional activated sludge plants. If no measures are taken, sludge rising due to denitrification can happen in the feeding and decant phase of the Nereda® process. Due to the high conversion rates in an AGS reactor and the plug-flow feeding from the bottom, degasification of N_2 gas can occur more easily. The gas deficit for N_2 gas between a completely stripped-out water phase and saturation is between 5 and 10 mg/l, depending on the water temperature and the ambient pressure. A higher temperature will give a lower solubility of N_2 gas in water and a lower N_2 gas deficit. A higher ambient pressure (atmospheric pressure plus the pressure due to the water column) gives a higher N_2 gas solubility and a higher N_2 gas deficit.

The equilibrium concentration of N_2 gas during aeration can be calculated with Eq. 11.2. With a gas fraction of N_2 gas in air (f) of 0.79, the minimum concentration of N_2 in the water phase can be calculated at every water depth (h). Also the saturation concentration can be calculated at every water depth by setting f to 1. If the N_2 concentration in the water phase becomes higher than the saturation concentration, small gas bubbles will form, leading to rising sludge. The actual N_2 concentration in the water phase is not only influenced by transfer from and to the gas bubbles during aeration and the denitrification process, but also by the mixing of the reactor. This complex process is shown in Figure 11.13.

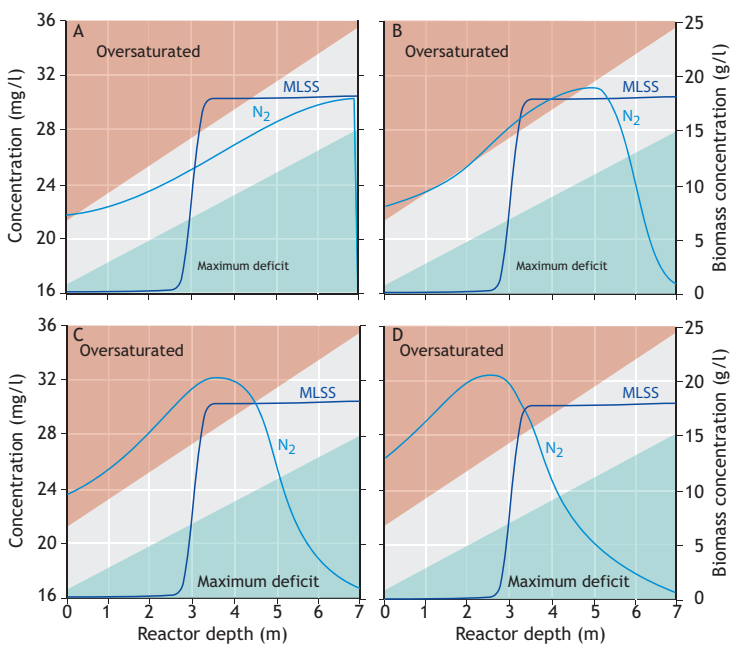

Figure 11.13 Stripping of nitrogen gas from fully saturated situation (A) and after 5 minutes (B), 15 minutes (C) and 60 minutes (D) of stripping at 20 °C. The red area marks an over-saturated concentration and the green area marks a deficit bigger than equilibrium with air.

$$C_{eq} = k_H \cdot f \cdot (P_{atm} + h \cdot \delta \cdot g) \quad (11.2)$$

where:

C_{eq}	equilibrium concentration of N_2 gas (mol/m^3)
k_H	Henry coefficient N_2 gas ($mol/m^3.Pa$)
f	gas fraction of N_2 gas in the gas bubble (-)
P_{atm}	atmospheric pressure (Pa)
h	water depth (m)
ρ	density of water (kg/m^3)
g	gravity acceleration (m/s^2)

During feeding, the potential for degasification is dependent on two processes. Firstly, the anoxic conditions lead to production of N_2 gas due to denitrification and thus a reduction of the N_2 gas deficit. Secondly, the plug-flow feeding from the bottom moves the water upwards in the reactor, lowering the ambient pressure and resulting in a lower saturation concentration. To prevent degasification during the feeding phase, a short stripping phase can be added at the end of the cycle if and when needed. By intensely aerating the reactor for a few minutes, the N_2 gas is stripped out and degasification of N_2 gas will not occur. A more in-depth view and description of the model is given by Van Dijk *et al.,* 2018.

11.4.5 Solids retention time

As mentioned earlier, while the average SRT of the sludge in a typical AGS process is similar to conventional activated sludge, the individual sludge fractions have a wide distribution of various SRTs. Since the granular fraction has an SRT allowing nitrification at all temperatures, the control of SRT is less sensitive than for activated sludge plants. The SRT of the flocculent fraction is not explicitly set, but optimally it should be as short as possible if the waste sludge is to be digested. In a properly designed AGS reactor, preferentially only smaller granules and flocculent sludge are regularly wasted. To limit the size of the granules and maintain optimal biological phosphorus removal efficiencies, it is necessary to control the SRT of the larger granules. This is done by occasionally purging sludge from the bottom of the settled bed. Large granules lower the specific biofilm surface area of the reactor. Thereby the potential aerobic conversion rates are decreased because these are dependent on the oxygen mass transfer rate from liquid to granule. Note that since these large granules are relatively old and well mineralised, the digestibility of this sludge is relatively low. Because the digestibility of the flocculent sludge fraction is higher than that of large granules, the overall digestibility of AGS waste sludge is similar to, or slightly higher than, activated sludge.

11.5 DESIGN CONSIDERATIONS

The AGS process contains similar treatment steps to those required by conventional activated sludge plants, but there are some key differences. The requirements for pre-treatment are similar to activated sludge; they depend on the wastewater characteristics and include screening, grit and if necessary oil, fat and grease removal. For stringent effluent requirements a range of post-treatment steps can be used, both chemical (such as metal dosing, activated carbon) and mechanical (such as sand filter, cloth media filter, ultrafiltration) systems. These pre- and tertiary technologies will not be further addressed in this chapter as they have similar designs to those for activated sludge effluents.

11.5.1 Plant configuration

Since the AGS process is a batch process, the treatment of a continuous influent flow requires multiple reactors. Each reactor goes through a cycle of feeding - reacting - settling. While a reactor is reacting or settling, it cannot receive influent. This is solved by applying multiple reactors, so that there is always one reactor that can receive influent. Subsequently, this approach stipulates that a minimum of three reactors are needed for continuous operation and feeding times will therefore be 1/3 of the total cycle time (Figure 11.14, left).

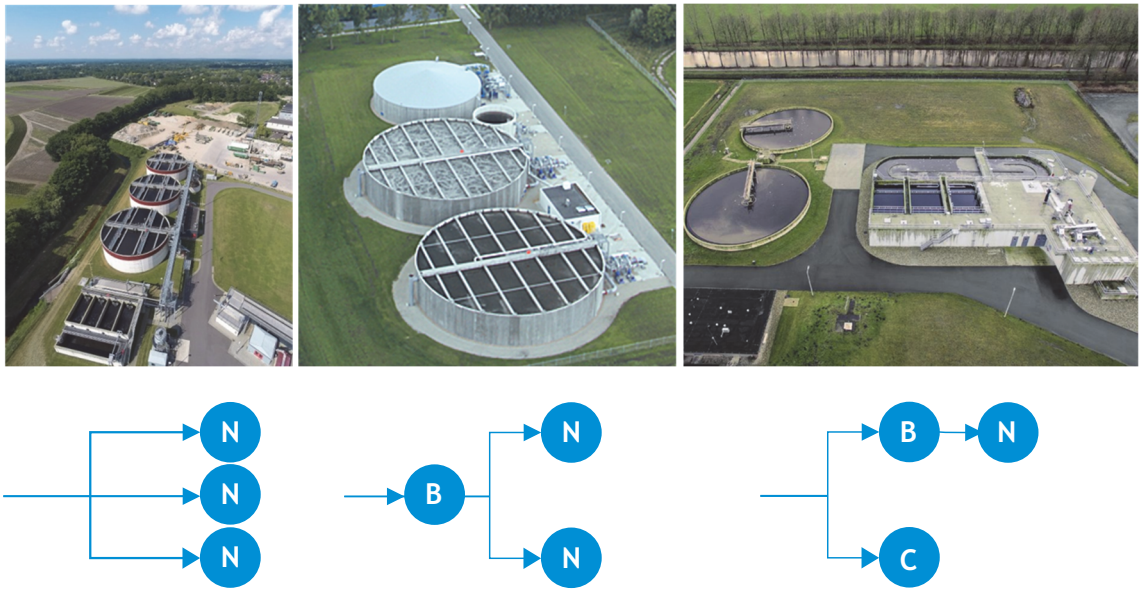

Figure 11.14 Different configurations for Nereda® plants in the Netherlands: (left) in Epe, three reactors (N) without a buffer, (middle) in Garmerwolde, two reactors (N) with a buffer (B), and (right) in Vroomshoop, a hybrid set-up where a Nereda® plant (with a buffer) is operated in parallel with a conventional activated sludge process (C).; here the waste sludge from the granular sludge plant is discharged in the activated sludge plant. Below is a depiction of the configuration.

This could result in feeding times that are longer than required for anaerobic COD uptake and phosphate release and results in larger than strictly necessary reactor volumes. This however makes biological phosphate removal more stable. Such over-design can be limited by applying more reactors (n), reducing the feeding time to $1/n^{th}$ of the total cycle time.

For smaller plants, building three treatment tanks might be uneconomic and an influent buffer can be used to overcome the need to have always one reactor in feed mode (Figure 11.14, middle). Using such an influent buffer only one reactor would suffice for treatment. Also, for larger plants with more than three reactors an influent buffer could be used to minimize reactor tank volume as the feeding phase can be shorter with higher flows. When minimising the anaerobic feeding time the minimal anaerobic time for biological phosphorus removal needs to be characterised which will depend on the local ratios of COD/P, $RBCOD/COD_{total}$ and the fraction of hydrolysable substrate.

Another design option for granular sludge is a hybrid configuration. In this configuration, the AGS reactor is built parallel to a continuously-fed activated sludge reactor (Figure 11.14, right). Certainly when upgrading existing activated sludge treatment plants this is often an interesting option. In the hybrid configuration the AGS reactor treats only part of the total influent and discharges the waste sludge into the activated sludge plant. By doing so, the AGS waste sludge improves the settleability of the activated sludge and the nutrient removal of the activated sludge reactor, making optimal use of both systems.

11.5.2 Design volume

The design volume is mainly dependent on the local influent flow rate and its variation, total COD and nitrogen load, water temperature and effluent

requirements. The sum of the times for the three phases in the cycle (feed, aerate and settle) and the amount of wastewater determines the total volume of the AGS reactors.

In the anaerobic feeding phase, wastewater is fed from the bottom through a settled granular bed into the reactor. Simultaneously the clean effluent is pushed out from the top of the reactor. As a result, the volume in the reactor is constant. In most conventional SBR plants this is simply not possible due to the flocculent sludge settling properties. The amount of exchange between fresh influent and clean effluent is expressed in the exchange ratio, which is a number between 0 and 100%. A properly designed AGS reactor can have an exchange ratio up to 65% and sometimes even more. This maximum exchange ratio is influenced by the level of (vertical) plug flow behaviour in the reactor. A sub-optimal plug flow will lower the maximum exchange ratio that can be reached and will lead to a higher required reactor volume. One of the parameters influencing the plug flow is the up-flow velocity (m/h). During the anaerobic feeding period RBCOD is taken up from the influent by the granular bed. To allow for the hydrolysis and uptake of RBCOD, generally feeding times of 0.5-3.0 hours are required, depending on the type of wastewater. In general, the anaerobic time is set by the hydraulic constraints on the specific treatment site.

The processes in the reaction phase depend on the effluent requirements. If the goal is to only remove COD, the reaction phase consists of a relatively short aeration period where only the COD and stored PHA is oxidized. In many cases this will also result in near-complete phosphorus removal, because PAOs play an important role in the granulation process and in COD removal. However, if nitrification is also required, the reaction phase will require a longer aeration period leading to larger volumes. The time needing to be allocated for denitrification is very dependent on the type of wastewater and sewer system used.

The total volume of the plant (Nereda® reactors + influent buffer) depends on the sludge loading rate, the flow variations and cycle configuration. In the following example we have a Nereda® plant with three reactors (on the left in Figure 11.14). To determine the plant volume, first a cycle configuration needs to be established. The total cycle time can be calculated by:

$$t = t_{feed} + t_{react} + t_{settle} \quad (11.3)$$

where:

t	total cycle time (h)
t_{feed}	feeding phase duration (h)
t_{react}	reaction phase duration (h)
t_{settle}	time for sludge settling and decanting (h)

In a plant configuration with multiple reactors without an influent buffer the total cycle time must be a multiple of the feeding phase duration:

$$t_{feed} = (t_{react} + t_{settle}) / (n_{reactors} - 1) \quad (11.4)$$

where:

$n_{reactors}$	number of reactors.

Hence in a 3-reactor configuration, the duration of the feeding phase is half of the sum of the duration reaction phase and the settling phase. The feeding phase and the reaction phase scale together, so a longer feeding phase leads to a longer reaction phase. Only the settling/wasting phase has a fixed duration of typically 20-30 minutes. As a result the reactor volume is only marginally influenced by the cycle time chosen. In practice the duration of the feeding phase is determined by parameters such as the exchange ratio, the up-flow velocity, and the minimum anaerobic feeding time.

In the design process the reaction (t_{react}) time can be freely chosen, for example 4 hours. Later, this value can be optimized to minimize the total plant volume, as can be seen in Figure 11.15. Based on equations 11.3 and 11.4 and the chosen value of t_{react}, a total cycle time (t) can be calculated. The average reaction time per day per reactor can be calculated based on the number of cycles per day:

$$n_{cycles} = 24 / t \quad (11.5)$$

and

$$t_{react,day} = n_{cycles} \cdot t_{react} \quad (11.6)$$

where:

n_{cycles}	number of cycles per day per reactor
$t_{react,\ day}$	total reaction time per day (h).

The reaction volume can now be calculated based on the total reaction time per day and the design flow, the COD concentration and the sludge loading rate. The sludge loading rate is typically similar to activated sludge processes and range between 0.1-3.0 kgCOD/kgTSS.d. Nitrification occurs typically until 0.4 kgCOD/kgMLSS.d at moderate temperatures. Higher temperatures can allow nitrification with even higher loading rates.

$$V_{reactor} = \frac{Q \cdot COD}{\frac{t_{react,day}}{24} MLSS \cdot SLR \cdot n_{reactor}} \quad (11.7)$$

where:

$V_{reactor}$	volume of a reactor (m^3)
Q	flow (m^3/d)
COD	COD concentration (kg/m^3)
MLSS	sludge concentration (kg/m^3)
SLR	sludge loading rate (kgCOD/kgMLSS.d)

In a configuration with a buffer the duration of the feeding phase can be more freely chosen. Typically, the feeding phase is shorter than in a configuration without a buffer, allowing for a longer reaction phase. The reactor volume will be more efficiently used. The total volume of the buffer and the reactors combined will not be much smaller than in a configuration without a buffer. However, the reduction in reactor volume will give an overall cost reduction. The feeding time t_{feed} is not calculated anymore according to equation 11.4 but it is freely chosen in the design process taking into account upflow velocity limits. Typical values of 0.5-1.0 hour are used. The buffer volume is calculated based on the peak flow during dry weather conditions. During rainy weather conditions the reactor scheduling is changed as described in section 11.4.2 and the buffer will not be emptied during the feeding phase because inflow and outflow are balanced. The buffer volume follows from:

$$V_{buffer} = \frac{Q_{peak}}{n_{reactor} \cdot n_{cycles}} - Q_{peak} \cdot \frac{t_{feed}}{24} \quad (11.8)$$

where:

Q_{peak}	DWF peak flow (m^3/d).

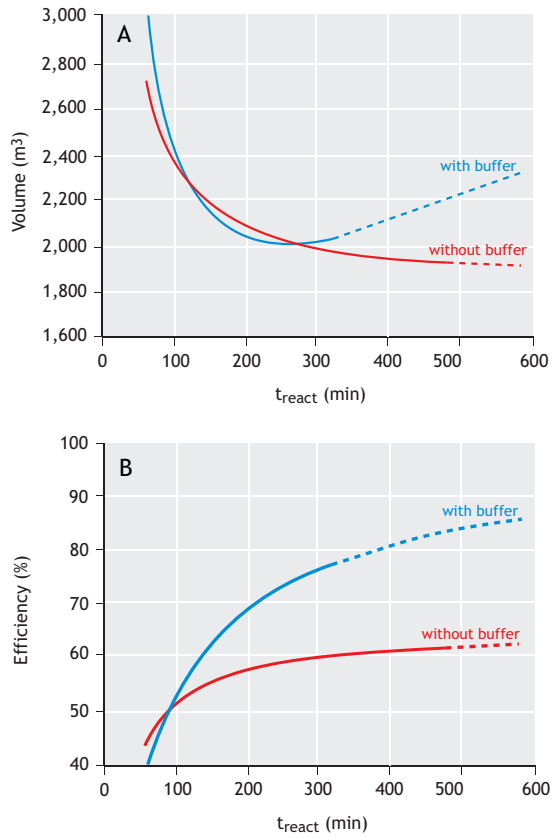

Figure 11.15 The effect of the application of a buffer. The application of a buffer (blue) does not lead immediately to a lower total volume (A), but the efficiency of the reactors (B) expressed as relative time available for reaction increases. The dotted lines show the design is limited by the exchange ratio.

Using all the above calculations, an optimal reactor configuration can be derived. Figure 11.15A shows that the total volume of tanks including the buffer (called the plant volume) does not change significantly when a buffer is applied. In contrast the remaining reactor volume is used more efficiently for aeration as shown in Figure 11.15B. This shows the efficiency, which can be expressed as the number of hours per day that the reactors are used in the reaction phase. Also, there is an optimal cycle configuration, where a minimum amount of reactor volume is needed.

In a hybrid configuration many variations are possible. It can be designed with or without a buffer, with one or many reactors. In the simplest form only one AGS reactor is built; here the conventional plant takes all the load when the AGS reactor is not feeding. Similar to the configuration with a buffer, the duration of the feeding phase and the reactor phase can be chosen based on boundaries from both the AGS process and the conventional process. As an alternative the AGS reactors can be built as a separate lane, continuously being fed a fixed fraction of the total flow coming into the plant. In this case a configuration either with or without a buffer can be applied.

11.5.3 Sludge treatment

Sludge is wasted in an AGS reactor for two reasons: the selection spill contains mainly slow-settling sludge and the SRT spill contains a mixture of granules. At the end of each cycle the worst-settling fraction of the AGS is wasted by removing the top of the sludge bed after a period of settling. This is called the selection spill and it is essential for the granulation process. The selection spill consists of sludge flocs, small granules, sheared off material from big granules and suspended solids from the influent such as cellulose fibres. The selection spill consists of relatively non-mineralised material. The digestibility of this selection spill is generally good. The treatment of the waste sludge is comparable with normal activated sludge, although recent developments show biopolymers can be recovered from the waste sludge (see Section 11.6).

11.5.4 Mixed liquor suspended solids

Aerobic granular sludge plants are generally operated at mixed liquor suspended solids (MLSS) concentrations which are higher than in typical conventional activated sludge plants. This is also an important parameter that drives the compactness of the Nereda® technology. A typical design concentration for an AGS plant is 8 g/l, but values as high as 15 g/l are not uncommon in operational plants. However, at values higher than 8 g/l the biomass concentration is not the limiting factor in the purification process. These high concentrations can be reached because of the dense biomass in the granules. Mature granules can have a MLSS concentration of 50-60 g/l. Therefore, with a typical bed porosity of 50%, a settled granular bed has a MLSS concentration of 25-30 g/l.

11.6 RESOURCE RECOVERY

The primary aim of wastewater treatment is the protection of human health and the environment. Nevertheless, in the last decade resource recovery has also become an important aspect of wastewater treatment process design (Kehrein *et al.,* 2020). The study of granular sludge formation has opened up new options for resource recovery. Especially the polymers that form the matrix of the granular sludge offer many new possibilities. The integration of resource recovery with Nereda® technology is briefly introduced below.

Traditionally energy in the form of biogas is recovered from waste sludge. The granular sludge process has a similar recovery potential for biogas as conventional activated sludge plants. In a conventional activated sludge plant the primary clarification is used to maximize biogas production. For granular sludge processes this has been integrated in the treatment reactor. Particulate COD from wastewater is incorporated in the floc fraction of the sludge. This fraction has a short residence time (0.5-5.0 days) and thereby a very good digestibility.

Granular sludge has a very long SRT and thereby a slightly lower digestibility than secondary flocculent sludge (200 *versus* 230 $mlCH_4/gVSS$). The total excess sludge gives a similar yield of biogas to activated sludge processes with combined primary and secondary sludge digestion (300 $mlCH_4/gVSS$) (Guo *et al.,* 2020).

One of the compounds in the flocculent sludge fraction is cellulose fibres (Figure 11.16) which form a significant fraction of the influent (20-30% of the particulate COD). This fraction ends up in the waste sludge and contributes to the higher biogas production from this fraction. The cellulose can be recovered by a sieving step either in the influent or in the waste sludge. The latter has the advantage of a smaller hydraulic flow (smaller sieve capacity needed). Recovered cellulose can be used to replace new cellulose in industrial processes. It can also be hydrolysed into sugars (and subsequently be fermented in volatile fatty acids) and dosed as substrate in WWTPs where the COD is limiting for phosphate removal or nitrogen removal.

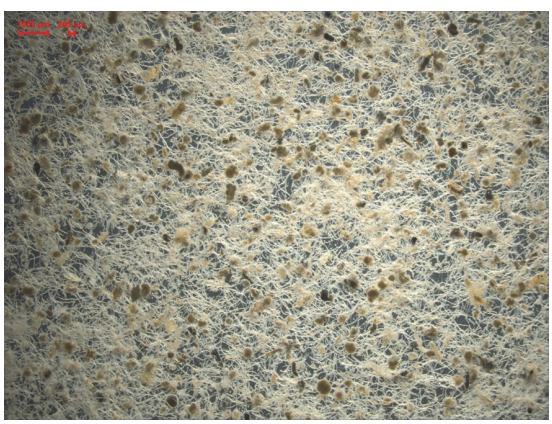

Figure 11.16 Image of sieved (≥ 200µm) excess sludge from a full-scale Nereda® plant showing cellulose fibres meshed with pieces of granular sludge.

Phosphate recovery can be integrated as in conventional activated sludge processes, usually in the form of struvite recovery from the digestate. When iron is dosed or present in the wastewater, vivianite is formed during digestion which can be recovered by magnetic separation (Prot *et al.,* 2019). An extra option in a batch granular sludge process such as in Nereda® plants is given by the large-scale release of phosphate during the influent addition. After the start of the aeration phase, phosphate concentrations are high. If the reactor liquid is retrieved at this stage, it is possible to recover phosphate as calcium mineral or as struvite. However, recovery at this stage should be balanced in order to leave sufficient phosphate available for effective biological phosphorus removal (Barat and Van Loosdrecht, 2006).

The most interesting raw material to recover is formed by the extracellular polymeric matrix of granular sludge. This can be extracted and used as the basis for novel materials. At the time of writing (2020), the extracellular polymeric matrix has in general not been well characterised (Seviour *et al.,* 2019) New analytical tools will unveil the composition in the coming years. Methods that have been used widely for sugar and protein analysis have shown to have large biases (Seviour *et al.,* 2019) and the composition is not a simple polysaccharide and protein mixture. In granular sludge, and probably also other biofilms, bacteria produce glycoproteins, sulphated glucosaminoglycan-like, Hyaluronic acid-like, and sialic acids containing compounds. These polymers are responsible for the very stable polymeric matrix that embeds the microorganisms which produce this matrix (Felz *et al.,* 2020).

Gel-forming polymers are generally extracted from biological sources and not produced from oil-based chemicals. The supply of these polymers is in general limited because separate production of them is too expensive. For instance, alginate supply is limited by the amount of naturally grown seaweed that can be harvested. Cultivation makes the polymer too expensive. The extracellular polymeric matrix of activated sludge or granular sludge can therefore form a very interesting new resource that can be recovered. The polymers from activated sludge have a more flocculent behaviour, while the polymers from granular sludge can form stable gels. The production

volume could be much larger than most current biopolymers; 5 kg/P.E.yr of biopolymers could be produced in a Nereda® plant. The harvesting and use of polymers from aerobic granular sludge are currently in the development stage. The first full-scale extraction facility was inaugurated in 2019 in the Netherlands (Figure 11.17) and the polymer is marketed under the brand name Kaumera gum. The current extraction process is similar to the harvesting of alginate from seaweed. First the polymer is solubilised by heating the sludge in an alkaline carbonate solution. After removing the non-solubilised material by centrifuging, the polymer is precipitated by pH neutralisation and calcium addition or by acidifying the liquid to a low pH.

Traditionally biopolymers are used mainly in the food and medical industry. This is due to their limited availability and high price. Wastewater-derived polymers are less suitable for these markets but the potential market volume is stimulating the development of new applications. First applications can be found in agricultural application, mainly also due to observed plant growth stimulation by the polymer. An interesting characteristic is the flame retardancy of the polymer (Kim *et al.,* 2020). This characteristic can be an added value when Kaumera is used for production of materials and coatings. Another interesting characteristic is the use of the polymer for composite material production. Where chemical polymers can be used for composite materials, they can bind up to 10-20% of inorganic filler material. Biopolymers can also be used but can hold up to 80% of filler material. Current investigations have revealed that clay-Kaumera composites resemble nacre materials (structured organic/inorganic material composite materials; *e.g.* mother of pearl). They have a very high tensile strength, are non-flammable and keep their strength up to 180 °C. Composites formed by Kaumera and cellulose produce a material resembling mother of pearl which has a high aesthetic value (Figure 11.18), showing that what is flushed through a toilet can be transformed into attractive products.

Figure 11.17 The Nereda® plant in Zutphen, the Netherlands with a Kaumera-extraction plant. In the background is the existing conventional activated sludge plant.

Figure 11.18 A: Flame-retardant composite plastic derived from Kaumera and clay. B and C: wastewater-based earrings and necklace. The main material is Kaumera and cellulose composite. The plastic sphere on the necklace is from PHA. The blue colour is vivianite and the red colour is extracted from anammox sludge (art created by Yuemei Lin).

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NOMENCLATURE

Symbol	Description	Unit
C_{eq}	Equilibrium concentration of N_2 gas	mol/m^3
$C_{s,o}$	Concentration of the substrate in the bulk	g/m^3
$C_{x,m}$	Maximum density of the biomass in the biofilm	g/m^3
D_s	Diffusion coefficient of the substrate	m^2/s
f	Gas fraction of N_2 gas in the gas bubble	-
g	Gravity acceleration	m/s^2
G	Dimensionless growth factor	-
h	Water depth	m
k_H	Henry coefficient N_2 gas	$mol/m^3.Pa$
L_y	Biofilm thickness	m
$n_{reactors}$	Number of reactors	-
n_{cycles}	Number of cycles per day per reactor	-
t	Total cycle time	h
t_{feed}	Feeding phase duration	h
t_{react}	Reaction phase duration	h
$t_{react,day}$	Total reaction time per day	h
t_{settle}	Time for sludge settling and decanting	h
Q	Flow	m^3/d
Q_{peak}	DWF peak flow	m^3/d
P_{atm}	Atmospheric pressure	Pa
$V_{reactor}$	Volume of a reactor	m^3

Abbreviation	Description
AGS	Aerobic granular sludge
AOB	Ammonia-oxidizing bacteria
CAS	Conventional activated sludge
EBPR	Enhanced biological phosphorus removal
GAO	Glycogen-accumulating organisms
MLSS	Mixed liquor suspended solids
NOB	Nitrite-oxidizing bacteria
PAO	Phosphate-accumulating organisms
PHA	Polyhydroxyalkanoates
RBCOD	Readily biodegradable COD
UASB	Upflow anaerobic sludge bed reactor
VFA	Volatile fatty acids
SLR	Sludge loading rate
SRT	Sludge retention time

Greek symbols	Explanation	Unit
μ_m	Maximum specific growth rate	1/d
ρ	Density of water	kg/m^3

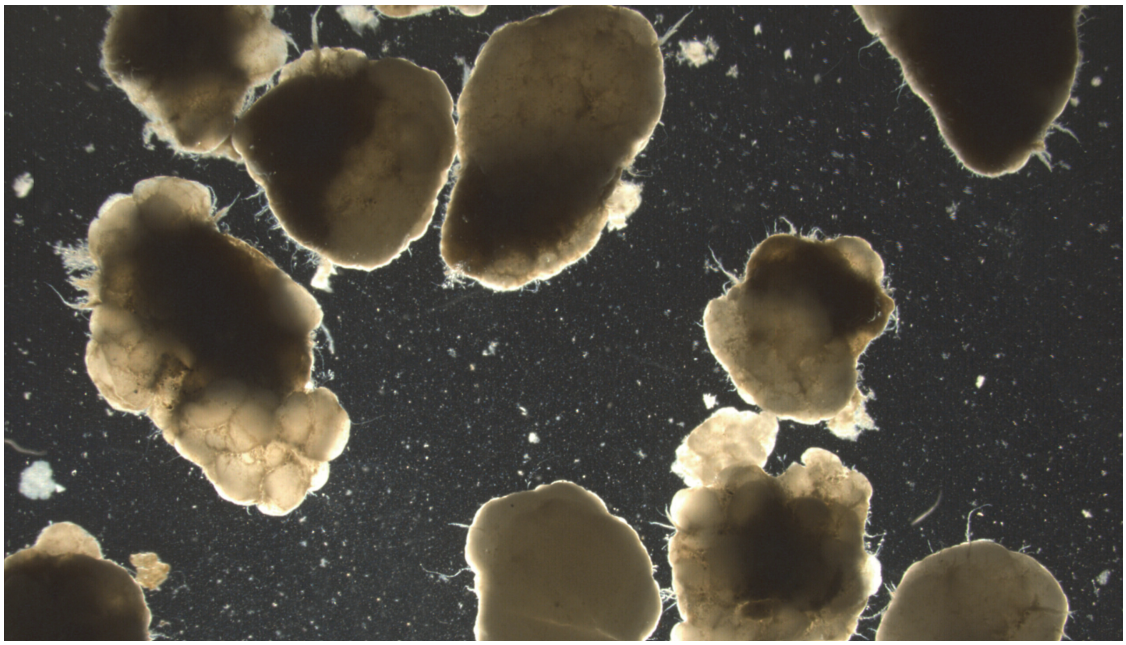

Figure 11.20 Aerobic granular sludge from a Nereda® plant (photo: Royal HaskoningDHV).

Figure 11.21 Data series presented in Figure 11.10 have been transformed into a sequence of notes connected in such a way that they are played or heard separately, the outcome is known as Nereda® Melody produced in cooperation between Royal HaskoningDHV and Foundation Pinta 021 (https://pinta021.org/muzika-vode/) (photo: Foundation Pinta 021).