NUTRIENTS REMOVAL IN MBRS FOR MUNICIPAL WASTEWATER TREATMENT

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ABSTRACT

Owing to increasingly stringent effluent quality requirements, intensifications of the conventional activated sludge process (ASP) are required. Due to high biomass concentrations employed, higher metabolic rates and better nutrient removal is possible in membrane bioreactors (MBRs). Decoupling of hydraulic and solids residence times offers additional possibilities for process design and optimisation. Recently, unconventional concepts like post-denitrification and enhanced biological phosphorus removal in MBRs have emerged. The objective of this paper is to present current knowledge on nutrients removal in MBRs and trends in process optimisation in comparison with conventional ASP.

Keywords: MBR, maintenance, post-denitrification, pre-denitrification, enhanced biological phosphorus removal

INTRODUCTION

The conventional activated sludge process (ASP) is the most common biological process in municipal wastewater treatment. Discovered in 1914 by Arden and Lockett, then commercialised in 1920 by John and Atwood as a continuous process (Védry, 1996), ASP is nowadays well understood and mathematically modelled. However, increasingly stringent effluent quality requirements in industrialised countries and rising needs for water reclamation call for further developments of ASP. Current and impeding legislation on wastewater treatment effluent has led to the need for improved treatment processes capable of removing higher percentages of nutrients, suspended solids, bacteria etc. Several different minimum standards for effluent concentrations are in existence (some examples are given in table 1). Requirements for effluents depend on the type of receiving water (e.g. lakes, lagoons, rivers, aquifers) and its quality category (e.g. in Japan), on regulations about the wastewater treatment technology (e.g. in the USA: Best Practical Technology Standards of Environmental Protection Agency), as well as on special demands locally adapted to the particular receiving water.
Membrane bioreactors (MBR) allow significant process intensifications and better effluent qualities due to the following changes of boundary conditions:

- They are operated at higher biomass concentrations with resulting high metabolic rates. This allows decreased reactor volumes and footprint.
- Since hydraulic and solids (biomass) residence times are independent of each other, MBRs offer an additional degree of freedom for process control. Degradation kinetics can thereby be optimised beyond ASP performance. Even slowly growing microorganisms with particular degradation features can be established.
- Since no gravitation settler is needed, operation is independent of sludge parameters.
- The membrane being a barrier for suspended solids, MBRs produce a more hygienic effluent.

- Although biological degradation of organic material and nutrients removal is basically identical, the utilisation of maintenance energy demands offers the possibility of decreased excess sludge formation.

A number of MBR plants have been established over the last couple of years. First applications of MBRs in wastewater treatment date back to the early 70s. In the meantime, three generations of MBR treatment plants have been developed and an increasing number of technical plants is coming into operation. Although several practical experiences and data are available for MBR processes there is still considerable optimisation potential.

The objective of this paper is to present current knowledge on nutrients removal and trends in process optimisation in comparison with conventional ASP.

Table 1: Effluent standards for municipal wastewater treatment plants (all ranges depend on size), [a] Loudoun County (Virginia) Sanitation Authority, [b] Magnetic Island (sensitive waters), [c] discharge to inland waters.

	Unit	EU (EC 1998)	Germany (AbwV 2002)	China (EPA 2000)	Japan (EA 1993)	USA[a] (LCSA)	Australia[b] (de Haas et al., 2004)	Australia[c] (Mallia et al., 2001)
Hygienic parameter:								
Total coliforms	no. $(100\ mL)^{-1}$	500		$2\text{-}3 \cdot 10^5$	$3 \cdot 10^5$	1	< 5	200
Organic load:								
COD	mg L^{-1}	125 or 75 %	75-150	100-250	160	10		
BOD_5	mg L^{-1}	25 or 70-90 %	15-40	30-80		1	< 5	5
Nutrients:								
NH_4-N	mg L^{-1}		10				< 1	2
N_T	mg L^{-1}	10-15 or 70-80%	13-18		120	1	< 3	10
P_T	mg L^{-1}	1-2 or 80 %	1-2		16	0.1	< 0.1	0.5
SS	mg L^{-1}	60-35 or 70-90 %		30-80	200	0 (0.5 ntu)		10

CARBON REMOVAL

Carbon being the main substrate is converted into biomass and CO_2, in MBRs usually by aerobic mechanisms in a single stage (compare Fig. 1):

$$C + O_2 \rightarrow \text{biomass} + CO_2 + \text{energy} \quad (1)$$

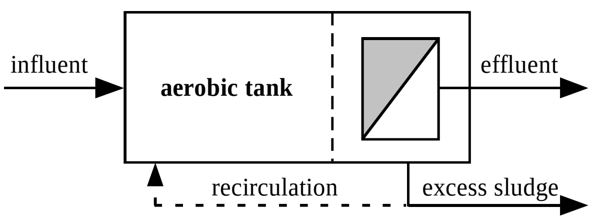

Fig. 1: Basic carbon and nitrogen removal MBR plant scheme

The relative yields of biomass and CO_2 depend on the sludge's physiological level. At low growth rates, microorganisms utilise available substrates mainly for maintenance purposes (Pirt, 1965). This effect is observed as lower sludge yield $Y_{B/S}$ at high sludge ages (van Loosdrecht and Henze, 1999) or at high biomass concentrations (Low and Chase, 1999), respectively. As excess sludge must be disposed of at costs which may account for 60% of the total plant operating costs (Horan, 1990; quoted in Low and Chase, 1999), wastewater treatment plants (WWTP) should be designed and operated such that pollutants are diverted from assimilation via biosynthesis to energy requiring functions associated with non-growth activities. The recognition that microorganisms satisfy their maintenance energy requirements in preference to producing additional biomass has revealed possible methods to reduce the generation of excess biomass. Other processes which account for lower biomass yields but cannot be distinguished macroscopically are endogenous respiration or lysis and successive cryptic growth. Currently, when subjected to severe substrate deficiency, organisms are considered to fall into dormancy rather than die (van Loosdrecht and Henze, 1999). Only predation or adverse conditions like pH and temperature or the presence of toxic substances or viruses lead to death (necrosis). Whichever phenomenon prevails, most processes can be well described by Pirt's equation, which in other words means $Y_{B/S} = f(\mu)$:

$$-\dot{r}_S = \frac{\dot{r}_B}{Y^g_{B/S}} + k_{m,S} \cdot c_B \tag{2}$$

Knowledge of the parameters maintenance coefficient $k_{m,S}$ and true yield $Y^g_{B/S}$ permits modelling of the process including final biomass concentration c_B and substrate removal rate $\dot{r}_S$. Considering literature data on aerobic and anaerobic growth of autotrophic and heterotrophic organisms (bacteria, fungi, plant cells, mixed cultures) in chemostats at temperatures ranging from 5 to 75 °C, Tijhuis et al. (1993) derived a thermodynamically based correlation for maintenance energy requirements. They concluded that the true yield is constant at values of 0.45 – 0.5 kgCOD $(kgCOD)^{-1}$, with biomass given here in kgCOD, too. Anomalies of this concept have been reported for very low growth rates (below 10 % of μ_{max}), i.e. severe substrate limitations caused by high biomass concentrations at a given loading rate, where maintenance demand is reduced by significant amounts (Pirt, 1987; Low and Chase, 1999; Müller and Babel, 1996). In a study cultivating *Pseudomonas fluorescens* on synthetic wastewater, however, Bouillot et al. (1990) found that neither true yield nor maintenance coefficient were influenced by operating parameters such as dilution or breeding rate (even below 5 % of μ_{max}). Wisniewski et al. (1999) concluded that maintenance demand does not depend on HRT, while Bulthuis et al. (1989) found that both the maintenance coefficient and the true yield decrease with decreasing HRT. While quite a few variations of the maintenance coefficient have been observed, no general correlation for its dependence on growth rate, HRT or sludge age has been reported yet. For MBRs, Bouillot et al. (1990) and Wisniewski et al. (1999) suggest $k_{m,S} \approx 0.04$ mgCOD $(mgVSS\ h)^{-1}$ and $Y^g_{B/S} \approx 0.36$ mgVSS $(mgCOD)^{-1}$.

Simply due to the high number of microorganism in MBRs, the substrate uptake or reaction rate can be increased. This leads to better degradation in a given time span or to smaller required reactor volume. COD and BOD_5 removal are found to increase with MLSS concentration. Arbitrary high MLSS concentrations are not employed, however, as oxygen transfer is limited due to higher and non-Newtonian viscosities (Rosenberger et al., 2001). Kinetics may also differ due to easier substrate access. In ASP, flocs may reach several 100 µm in size (Wisniewski et al., 1999). This means that the substrate can reach the active sites only by diffusion which causes an additional resistance and limits the overall reaction rate (diffusion controlled). Hydrodynamic stress in MBRs

reduces floc size (to 3.5 µm in sidestream MBRs (Cicek et al., 1999)) and increases the apparent reaction rate.
In various lab and pilot-scale studies, reported loading rates range between 1.2 to 3.2 kgCOD m^{-3} d^{-1} and 0.05 to 0.66 kgBOD m^{-3} d^{-1}. The process has been found to be rather insensitive to HRT between 2 and 24 h (Stephenson et al., 2000) resulting in very high removal percentages, while for anoxic COD-removal Wisniewski et al. (1999) found an increased removal with increased HRT (2 – 10 h). Sludge age (reported range 5 to 3500 d) also appears to have little influence on effluent quality at values above 30 d (Stephenson et al., 2000). Reported sludge yields range from 0 – 0.34 kgMLSS $(kgCOD)^{-1}$. Little or no sludge is produced at sludge loading rates of 0.01 kgCOD $(kgMLSS\ d)^{-1}$ (Stephenson et al., 2000) which are lower than commonly employed F:M ratios. Table 2 gives a comparison between ASP and MBR sludge yields at different sludge ages.

Table 2: Sludge yield at different sludge ages in ASP and MBR (Stephenson et al., 2000)

Sludge age	d	12	24	102
ASP yield	kgSS $(kgCOD\ removed)^{-1}$	0.28	0.26	0.07
MBR yield	kgSS $(kgCOD\ removed)^{-1}$	0.22	0.18	0.02

At technical scale, no significant decrease in excess sludge production is reported for sludge ages < 30 d in the German WWTPs Markranstädt and Rödingen. In order to significantly reduce excess sludge formation, sludge age needs to be increased to > 100 d (Kraume and Bracklow, 2003). At the MBR plant on Magnetic Island (Australia), 0.48 kg MLSS $(kg\ COD)^{-1}$ are produced, including precipitation solids. With an estimated 0.25 kg MLVSS $(kg\ COD)^{-1}$, biological sludge yield is along the line of conventional ASP (de Haas et al., 2004).
As given in table 3, sludge ages or SRTs in realised MBR plants and processes exceed those in ASP only slightly. HRTs are in the same range. With around 15 kg m^{-3}, MLSS concentrations in the MBR process are three times higher than in ASP. BOD_5 loading rates can thereby be increased accordingly, yielding F:M ratios in a similar range. With regards to organic load, discharge standards are always met. Effluent BOD_5 is always < 10 mg L^{-1}. The higher removal rates can also be attributed to complete particulate retention of suspended COD and BOD_5, high molecular weight organics, and biomass (no washout problems as encountered in ASP) (Stephenson et al., 2000).

NITROGEN REMOVAL

Nitrogen removal is one of the main concerns in modern wastewater treatment. Currently, the most widely applied technology for N-removal from municipal wastewater with activated sludge systems uses nitrification combined with denitrification (Mulder, 2003). This is similar for ASP and MBR plants. In municipal wastewater, most nitrogen is present in the form of ammonium. In an aerated step the ammonium is transformed to nitrate. In a successive non-aerated, anoxic step, nitrate is converted to elementary nitrogen.
Since microorganisms need nitrogen for growth, N is also removed with the excess sludge.

Table 3: Process conditions and degrees of removal in typical conventional ASP and selected MBR plants for municipal wastewater treatment ([a] Mudrack and Kunst, 1985, [b] MUNLV, 2003, [c] Cicek et al., 1999, [d] Hotchkies, [e] Kraume and Bracklow, 2003, [f] Tazi-Pain et al., 2002, [g] de Haas et al., 2004, [h] Mallia et al., 2001, [j] Gander et al., 2000).

	Unit	Conventional ASP[a,b,c]	MBR[b]	MBR[c]	ZenoGem Milton, (USA)[d]	6 German 750-12,000 EP plants[e]	BIOSEP (France)[f]	Magnetic Island (Australia)[g]
SRT	d	10-25	< 30	30	> 15[h]	25-28	> 20	30
HRT	h	4-8[j]	> 6	8[j]	3	< 10		
MLSS	kg m^{-3}	5	12-16		15-20[h]	8-16, mainly 12	15	15
BOD_5 loading rate	kg m^{-3} d^{-1}	0.25 0.32-0.64[j]		0.4-0.7	2.5	0.32-0.79		
BOD_5 (F:M)	kg kg^{-1} d^{-1}	0.05	< 0.08		< 0.2	0.02-0.066		
BOD_5 removal effluent conc.	% mg L^{-1}	85-95[j] 15		98-99[j]	> 99 < 2	98 < 5	> 97.5	 < 3
COD removal effluent conc.	% mg L^{-1}	94.5	 < 30	99		96.1 < 25	97	
TSS removal TSS turbidity	% mg L^{-1} NTU	60.9 10-15	 0	99.9	> 99 < 2 < 0.1	 0	99.8	 0.6
N_{total} removal effluent conc. NH_4^+ removal	% mg L^{-1} %	 < 13 98.9	 < 13	 99.2	> 96 (TKN) < 2 (TKN) < 0.5 mg L^{-1}	92 < 10 < 1 mg L^{-1}	98.6 (TKN) 0.4 (TKN)	 < 2 (with sugar dosing)
P_{total} removal effluent conc.	% mg L^{-1}	88.5 0.8-1	 < 0.3	96.6	> 99 < 0.1	86.5 1		 < 0.3

NITRIFICATION

The design and operation of a nitrification activated sludge system is similar to that operated for BOD removal only (Fig. 1). Nitrification is implemented in two steps by autotrophic bacteria. In the first step ammonium is oxidised to nitrite by e.g. *Nitrosomonas*:

$NH_4\text{-}N + O_2 \rightarrow NO_2^- + \text{energy}$. (3)

In the second step nitrite is converted to nitrate by e.g. *Nitrobacter*:

$NO_2^- + O_2 \rightarrow NO_3^- + \text{energy}$. (4)

Since the amount of energy gained by nitrification is relatively low, nitrifiers are slow growing and a minimum sludge age of > 5 days is necessary in order to ensure complete nitrification (Fan et al., 2000). Therefore, the design of MBR treatment plants is based on a minimum SRT of 8 to 10 days at 10 °C, as can be seen from plant data shown in table 3.

Nitrification is an aerobic process where oxygen is used as the electron acceptor and is therefore necessary for the process to occur. The half-saturation constant for dissolved oxygen (DO) has been reported to be in the range of 0.3 - 1.3 mg L^{-1} (Charley et al., 1980). MBR plants are usually operated at high MLSS concentrations which lead to an increase in viscosity and a change of rheology (Rosenberger et al., 2002). As a consequence, the degree of mixing decreases and anoxic (nitrate but no oxygen present) micro zones can be present in the aerated tank resulting in simultaneous denitrification. On the other hand, exceeding MLSS concentrations cause problems with membrane performance and oxygen mass transfer rate because of high sludge viscosity (Rosenberger et al., 2001). These considerations currently lead to optimal MLSS concentrations of

around 15 g L^{-1} for most effective MBR operation. Typically, MBR plants of technical size achieve total nitrification with effluent ammonia concentrations below 1 $mgNH_4$-N L^{-1} (see table 3).
The maximum specific nitrification rates reported are e.g.: 1.7 - 2.0 $mgNO_3$-N $(gVSS\,h)^{-1}$ for municipal wastewater (Fan et al., 2000), 0.91 - 1.12 $mgNO_3$-N $(gVSS\,h)^{-1}$ for domestic wastewater (Harremoes and Sinkjaer, 1995), and 0.78 - 1.81 $mgNO_3$-N $(gSS\,h)^{-1}$ for synthetic wastewater (Muller, 1994). While the mean nitrification activity has been demonstrated to be more than twice that of an equivalent ASP: 2.28 gNH_4-N $(kgMLSS\,h)^{-1}$ for an MBR compared to 0.96 gNH_4-N $(kgMLSS\,h)^{-1}$ for an ASP (Zhang et al., 1997), other authors found the opposite (e.g. Liebig et al., 2001). This change in specific nitrification rates can be attributed to a shift in the microbial community. Less *Nitrobacter* sp. and less *Nitrosomonas* sp. were found in MBRs but more *Nitrospira* sp. (Liebig et al., 2001, Witzig et al., 2002).

DENITRIFICATION

Denitrification is the dissimilative reduction of nitrate to molecular nitrogen. The reduction is stepwise and the following intermediates are produced:

$$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2 \; . \qquad (5)$$

Numerous mostly heterotrophic organisms are able to perform denitrification. Since heterotrophic organisms need an organic carbon source, the available C-source influences the denitrification rate. Easily degradable substrate (e.g. acetate) leads to higher denitrification rates (up to 20 $mgNO_3$-N $(gMLVSS\ h)^{-1}$) than a substrate like raw water that is harder to degrade (1 - 6 $mgNO_3$-N $(gMLVSS\,h)^{-1}$). Lowest rates (0.2 - 0.6 $mgNO_3$-N $(gMLVSS\,h)^{-1}$) are achieved by endogenous denitrification, i.e. when no external C-source is present (Kujawa and Klapwijk, 1999).
A big variety of technical schemes is used for denitrification, but the most common are pre- and post-denitrification.

PRE-DENITRIFICATION

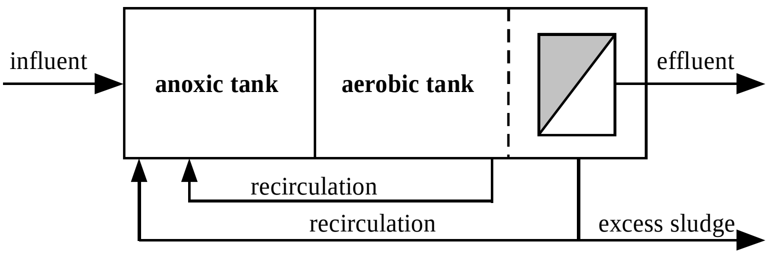

Figure 2: Flow sheet for MBR with basic pre-denitrification (modified Ludzack-Ettinger process).

In basic pre-denitrification systems (Fig. 2), against the obvious order, the anoxic denitrification tank is passed before the aerobic nitrification step. Nitrate is recycled from the aerobic tank to the anoxic zone. In most conventional WWTPs and MBR plants nitrogen removal is achieved by pre-denitrification for two major reasons. Firstly, biodegradable organic matter available in the anoxic zone improves denitrification rates, hence reducing the required reactor volume. Secondly, the oxidation capacity of nitrate degrades part of the organic matter, hence reducing oxygen demand and achieving savings in aeration requirement. On the other hand, N-removal depends on the recycling ratio characterising the transport of nitrate produced by nitrification in the aerated zone

back to the anoxic zone and is therefore limited to 75 to 90 %. Nitrogen removal can be constrained by raw sewage characteristics with a low ratio of COD (or BOD) to TKN. In these cases, like in ASP, a supplementary carbon source is required to achieve high N-removal (Magnetic Island plant, see table 3).
Kubin et al. (2001) found that cascading the aerated tank is favourable. Minimal aeration for complete nitrification (effluent ammonia-N concentration significantly below 1 mg L^{-1}) can be adjusted, hence less aeration is required (energy saving) and the development of anoxic micro zones is promoted, leading to better nitrogen elimination. Additionally, oxygen transfer to the anoxic zone can be minimised and the anoxic reactor volume can therefore be reduced.
Instead of taking place in two separate tanks, nitrification and denitrification can also be implemented in one frequently aerated tank providing aerobic and anoxic time phases. In these systems, nitrogen elimination is connected to aeration control and can reach up to 90% with an elaborate control concept (Boës, 1991). This so-called intermittent denitrification is used in a number of MBR installations especially in France for both industrial and municipal waste water treatment (Tazi-Pain et al., 2002). In these BIOSEP® plants (see table 3), COD elimination, nitrification, denitrification and filtration are realised in one tank, reducing footprint to a minimum.

POST-DENITRIFICATION

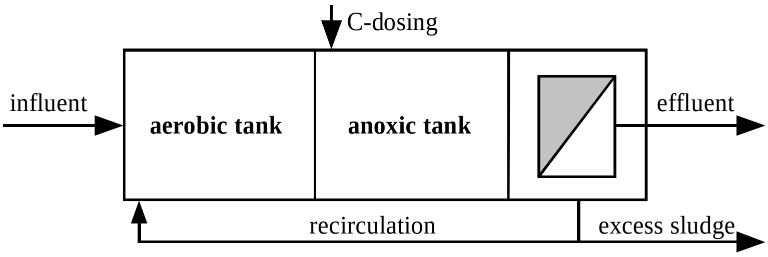

Figure 3: Basic flow sheet for post-denitrification

In post-denitrification systems (Fig. 3) the obvious order of nitrification and denitrification is realised. The elimination rate is therefore not limited by the recycling ratio. Since most organic matter is degraded already in the aerated zone, in most realised plants an external C-source is dosed to the anoxic tank in order to achieve higher denitrification rates and to keep the denitrification volume small (Sadick et al., 2000). In ASP, a second aerated zone is installed after the anoxic tank to ensure complete COD consumption. In MBRs this step is not necessary because of the heavily aerated membrane unit. As shown above, pre-denitrification has been traditionally implemented for nitrogen removal, also for MBRs. However, some characteristics of MBR technology could render post-denitrification an attractive alternative, even without a supplementary C-source (Gnirss et al., 2003). These specific features are summarised here:
(i) Low denitrification rates reported in current pre-denitrification MBR plants due to the combination of several detrimental parameters: high operation sludge age, high oxygen carry-over to the anoxic zone from the membrane system, separated or submerged in a sequenced aerated reactor; one single totally mixed anoxic reactor instead of a multi-stage design, etc. In some cases, reported denitrification rates approached or even fell below the endogenous rate. MUNLV (2003) suggests a volume ratio $V_{denitrification}/V_{nitrification}$ of 1:1 for pre-denitrification MBR plants while the ratio for conventional plants is only 1:3. In case of an anaerobic zone being installed at the inlet of

certain MBR processes (see Fig. 4), this effect can theoretically be enhanced, since a considerable amount of the easily biodegradable substrate does not reach the denitrification zone.
(ii) Insignificance of higher air requirements with post-denitrification mode: the savings due to nitrate recycling are minor in comparison with the importance of air requirements for membrane aeration in MBR.
(iii) Less equipment and energy requirement, as the aerobic/anoxic sludge recirculation loop is not required.
(iv) Better biomass redistribution due to the sludge recirculation pattern over the entire reactor volume: this leads to less sludge being in contact with the membrane, while more sludge is present in the anoxic zone.
For these reasons post-denitrification is identified as a promising configuration in MBR technology when enhanced nitrogen removal is required. In a lab and pilot-scale study post- and pre-denitrification were compared. N-elimination was around 96 % in post-denitrification and effluent values were around 2.5 $mgN_{total}\ L^{-1}$ while with pre-denitrification only 84 % of N were eliminated (Gnirss et al., 2003). The conclusion of the study was that post-denitrification even without C-dosing is a reliable technology delivering far better effluent values than pre-denitrification and can be competitive when an anaerobic reactor for enhanced biological phosphorus removal (EPBR) is additionally installed at the inlet (Gnirss et al., 2003; Lesjean et al., 2003). Storage compounds within the cells built up in the EBPR process possibly act as C-sources for denitrification, leading to denitrification rates above endogenous rates (Vocks et al., 2004). A small post-denitrification MBR full scale plant is now planned in the vicinity of Berlin.

PHOSPHORUS ELIMINATION

PRECIPITATION

Phosphorus removal processes can be divided into two basic groups: chemical and biological processes. Chemical P-removal is achieved by transforming phosphate into hardly soluble and thus precipitating iron, aluminium or calcium salts. These salts are withdrawn together with the excess sludge. The necessary precipitants can be dosed at different points of the reactor.

ENHANCED BIOLOGICAL PHOSPHORUS REMOVAL

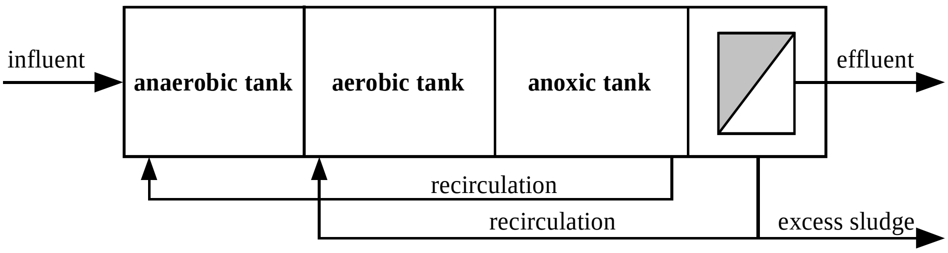

Figure 4: Flow sheet for MBR with enhanced biological phosphorus removal and post-denitrification

A part of the phosphorus is always removed biologically because P is one of the essentials needed for bacterial growth (1.5-2.5 % (w/w) based on dry weight). In addition, an enhanced biological phosphorus removal (EBPR) is possible. This is reached by special phosphate accumulating organisms (PAO) which can build up a polyphosphate storage under aerobic conditions resulting in accumulated phosphorus levels of 6 to 8 %. *Acinetobacter* and other phosphorus removal organisms consume this storage under anaerobic conditions for energy production and growth. Therefore, an anaerobic reactor has to be installed for EBPR to enrich PAOs in the sludge (Fig. 4). The anaerobic stage serves two important purposes: it provides a fermentation zone to produce simple hydrocarbons used by phosphorus removal bacteria, and it provides an environment that gives these organisms a competitive edge to ensure their survival in the system. The phosphate is removed by drawing off the excess sludge. Hence, a constant excess sludge withdrawal is necessary for EPBR. Since MBRs usually work at high sludge ages, chemical P-removal was generally installed in the past when P-elimination was required. However, recent studies show that EBPR is also possible in MBRs operating at sludge ages of up to 26 days (Adam et al., 2002). P-elimination of over 99 % was reached (effluent values below 0.05 $mgP_{total}\ L^{-1}$) without dosing of precipitants (Gnirss et al., 2002). Adam (2004) reached P- concentrations of up to 7 % (w/w) in an MBR operated at 15 days SRT. EBPR will also be attempted in real scale in a planned MBR plant in the vicinity of Berlin. In sensitive areas a combination of EBPR and precipitation is a promising technology (Adam, 2004) to constantly ensure very low effluent P-concentrations.

In Fig. 5, typical spatial nutrient concentration profiles over a cascaded plant are presented. They were measured in a cascaded lab scale plant operating with EBPR and post-denitrification at 15 d SRT. Phosphate release in the anaerobic tank and complete P-uptake in the aerobic become apparent. Ammonium is completely converted to nitrate in the aerobic zone and nitrate is degraded in the anoxic. Nitrate-N effluent concentration is 4.8 mg L^{-1}.

Ahn et al. (2003) achieved EPBR using a sequencing anoxic/anaerobic membrane bioreactor (SAM) process. In this process, sequencing anaerobic and anoxic conditions were implemented in a single tank by switching on and off the recirculation from the following aerated reactor where also the membranes were installed. In their lab scale trials phosphorus elimination was around 93 % and corresponding average effluent P concentration was 0.26 mgP L^{-1}. Nitrogen removal was relatively poor at 60 % and process optimisation still has to be done.

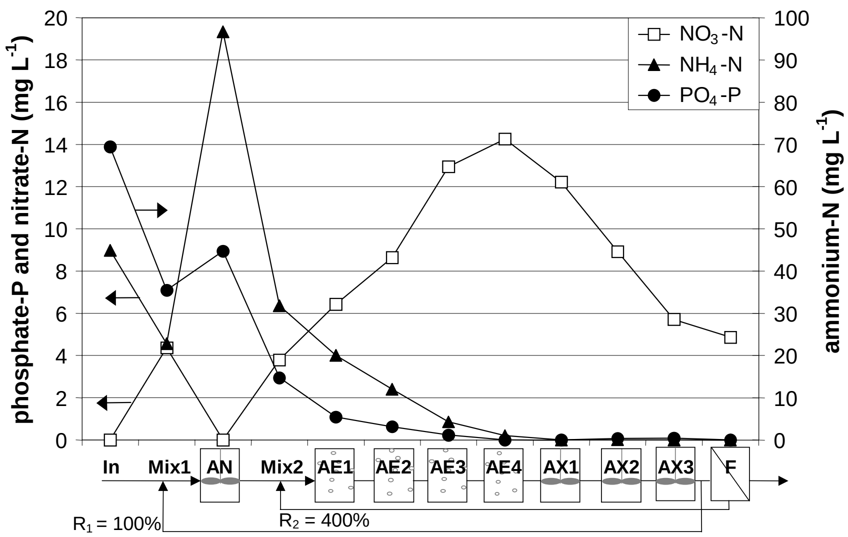

Figure 5: Concentration profiles of phosphate, nitrate and ammonium in a cascaded lab scale MBR operated with municipal wastewater (AN: anaerobic, AE: aerobic, AX: anoxic, F: filter chamber).

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