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Start-up and experimental analysis of a pilot-scale SBR as a basis for the design and economic evaluation of full-scale plants

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Summary

Abstract

Wastewater treatment is strictly regulated in Europe and Italy to prevent the pollution of the natural water bodies. At European level the law was recently updated in the EU directive 2024/3019 of the European Parliament and of the Council of the 27 of November 2024. The majority of the plants for domestic wastewater treatment consist of a conventional activated sludge process (CAS), that comprehends an aerated tank in which the degradation of the organic matter is carried out by microorganisms in steady state conditions. Sequencing Batch Reactor technology emerges as an innovative way to treat wastewater, that switches from a continuous, mixed flow to a batch system that evolutes in time and not in space; this helps with process flexibility and allows to avoid the settlers, since sedimentation can occur directly inside the reactor, saving a lot of space and energy. In this context, a laboratory experience with a pilot scale SBR was carried out to deeply investigate the work of this type of technology. All the devices that constitute the reactor are interfaced with an Arduino, which acquires all the signals and sends them to a computer. A Matlab code controls the system by processing the data and sending commands to the devices. Thanks to this automation, it is possible to let the reactor work continuously through the four phases that it manages, that are biological phase, sedimentation, discharge and charge phase. The evaluation of the efficiency of the COD removal is encouraging, since is almost always over 90%, with a concentration of COD in the effluent that respects BAT limits. A calibration test was performed to estimate the parameters that govern the microbial activity, following the Activated Sludge Model 3, but the results were bad, highlighting the need of further investigations. The rate of microbial growth was investigated through a solids analysis, to make an estimation of the sludge retention time to manage the periodical discharge of the sludge. Results affirm that a SRT of 4.8 d and a daily discharge of 4.16 L/d are needed. Even a brief analysis on nitrogen removal has been carried out, but this aspect needs more time to be studied. As a conclusion, since the efficiency of removal is good, a comparison between a CAS and a SBR at industrial scale was carried out, to quantitatively demonstrate the economic advantages of SBR, revealing that it would guarantee a 62% of land saving and 534 €/y saved during operational phase.

1. Introduction

1.1. Italian regulation on wastewater

Due to human life and activities, it is possible to distinguish between several types of wastewaters, that are classified and regulated, in Italy, inside the law decree 152/06, also called "Testo Unico Ambientale", that unifies all the main environmental laws.
In the document, the classification comprehends:

- Domestic wastewater: it comes from residential settlements and services and resulting mainly from human metabolism and domestic activities. [1]
- Industrial wastewater: any type of wastewater discharged from buildings or installations in which commercial activities, or the production of goods are carried out, other than domestic wastewater and stormwater runoff. [1]
- Urban wastewater: domestic wastewater or the mixture of domestic, industrial wastewater or stormwater runoff discharged into collecting systems, even if separate, and originating from urban agglomeration. [1]

All of these types of wastewater can be discharged in a receiving body (maybe after specific treatments), only if they respect the definition of wastewater discharge: any discharge carried out exclusively via a stable collection system, that seamlessly connects the production cycle of the effluent wastewater with the receiving body (surface water, soil, subsoil and sewerage system), irrespective of the polluting nature of the effluent, even subject to prior purification treatment. [1]
Wastewater discharges can be sent to different receptors, but there are some conditions that must be respected. The sewerage network can always receive domestic wastewater, without any preliminary treatment, since it ends into specific plants that are able to treat this kind of wastewater. Industrial wastewater needs authorization (AUA or AIA) and compliance to emission limit values (ELV), to be discharged into sewerage system; the Integrated Water Service Manager establishes in a regulation the condition to be respected, since the ELV may vary from the standards of the law, depending on the characteristics of the wastewater treatment plant of the area. In any case, industries must create some self-inspections of the water that they discharge.
Regarding surface water bodies, all the types of wastewaters need authorisation for the discharge; domestic ones are regulated by regional laws, while industrial ones need AUA or AIA and the compliance with the emission limit values, that are stricter than the previous ones.
The wastewater discharge in soil or subsoil surface layers, or in subsoil or groundwater is generally forbidden, with just few exceptions. [1]

1.2. European regulation on urban wastewater

The legislation on urban wastewater treatment was recently updated by the European Parliament and the Council, in the EU directive 2024/3019 of the European Parliament and of the Council of the 27 of November 2024, concerning urban wastewater treatment. [2]

This directive consist in the updating of 91/271/EEC and "should continue to pursue the same objective while also contributing to the protection of public health in accordance with the One Health approach, which is aimed at sustainably balancing and optimising the health of people, animals and ecosystems, when for instance urban wastewater is discharged into bathing waters or into water bodies used for the abstraction of drinking water, or when urban wastewater is used as an indicator for parameters relevant for public health". [2]

It should also increase synergies with the principles of climate change adaptation and contribute to the reduction of greenhouse gases emissions that could come from wastewater collection and treatment.

One of the main updates of the directive is the reduction of the threshold of agglomerations that must have a collecting system for urban wastewater. The threshold has been modified from 2000 to 1000 population equivalent, where p.e. is a unit of measurement that refers to the biodegradable organic load that needs a BOD_5 of 60 g of oxygen per day, to be degraded. It's assumed that one population equivalent discharges 200 l/d. Just a few declared exceptions can be accepted. The updates for plants between 1000 and 2000 p.e. should be ready by the end of 2035.

Other sources of pollution that needs to be managed with this new version of the regulation are storm water overflows and urban runoff; they represent sources of pollution that cannot be neglected, especially considering big agglomerates, of 100000 p.e. or more. Solution to manage them should be found locally and integrated with urban wastewater management plans; green areas, for example, promote drainage, as well as improving air quality.

Furthermore, all discharges of urban wastewater from agglomeration equal or greater than 1000 p.e., must undergo secondary treatments, that consist in biological treatments, to protect public health and the environment. For agglomerations of 150000 p.e. or more, instead, also tertiary treatments should become mandatory, that consist in the removal of nitrogen and phosphorus, that may cause eutrophication. If the area under exam is particularly subjected to eutrophication, the regulation becomes stricter, with the limit of 10000 p.e. In any case, the updates should be integrated by the end of 2039.

Another aspect that is underlined into the new directive is the one of micropollutants: nowadays they are routinely detected in all waters of the European Union and some of them are hazardous for environment and human health, even in low concentrations.
The term micropollutants refers to pharmaceutical products, antibiotics, cosmetics, PFAS, microplastics, etc. Some micropollutants are removed also with primary, secondary and tertiary treatments, but in some cases quaternary and more specific treatments can be necessary, on the basis of precautionary principle and a risky-based approach. The first aspect on which quaternary treatments should focus on are organic micropollutants, since some removal technology has already been designed. As a general rule, all plants that treat wastewater from agglomerations of 150000 p.e. or more, should install quaternary treatments by the end of 2045, with some exceptions specified in the full text of the directive. Micropollutants may come also from wastewater that are not domestic, but industrial assimilated to the domestic ones; in this case, plant needs also to be precisely informed on what they are treating, in order to reduce the emissions. These types of additive treatments will generate an increase in the costs of new equipment and monitoring devices; to face these additional costs, in accordance with the polluter-pays principle, the producers of goods that will generate micropollutants must take responsibility for the additional treatments required. This method will limit the financial impact on the users and will encourage the production of greener products.
Another aspect underlined in the directive is the one regarding energy: all wastewater treatment plants should reduce their energetic consumption and also try to produce some renewable energy exploiting their processes. In particular, the objective is, for plants treating a loading corresponding to 10000 p.e. or more, to consume an annual amount of energy that do not overcome the quantity of renewable energy produced. This objective can also be calculated on a national basis, not for every single plant, and have to be achieved by the end of 2045.
Furthermore, the directive has the prospective of preventing epidemics and/or pandemics, imposing the control on specific viruses or pathogens inside wastewater, in order to be able to manage them before the total spreading. This monitoring is also important to analyse and improve the research on antimicrobial resistance, since urban wastewater is the major source of antimicrobial agents and their metabolites; this aspect is aligned with the 'One Health' approach, recognised by the WHO.
Another relevant aspect is the circular economy applied to urban wastewater treatment: it's important to treat water and sludge in order to reuse them, minimizing further withdrawals from surface water. The reuse of sludge must be developed together with the control of the nutrients inside it, eventually removed and reused as fertilizers. [2]

1.3. Domestic wastewater

Focusing on domestic wastewater, its composition is almost regular in time and everywhere. For this reason, it's easier to build a plant to treat it, with respect to industrial wastewater, that is specific for each production line. The most frequent way to treat domestic wastewater is to discharge it into sewerage system and send it to a wastewater treatment plant (WWTP).
In the figure below, is shown the typical composition of domestic wastewater.

Constituent	Weak	Medium	Strong
	(all mg/L except settleable solids)		
Alkalinity (as $CaCO_3$)[a]	50	100	200
Ammonia (free)	10	25	50
BOD_5 (as O_2)[b]	100	200	300
Chloride[a]	30	50	100
COD (as O_2)	250	500	1,000
Total suspended solids (TSS)	120	210	400
Volatile (VSS)	95	160	315
Fixed	25	50	85
Settleable solids, mL/L	5	10	20
Sulfates[a]	20	30	50
Total dissolved solids (TDS)	200	500	1,000
Total Kjeldahl nitrogen (TKN) (as N)	20	40	80
Total organic carbon (TOC) (as C)	75	150	300
Total phosphorus (as P)	5	10	20

[a]To be added to amount in domestic water supply. Chloride is exclusive of contribution from water-softener backwash.
[b]For newer, tighter collection systems, or where water conservation is practiced, these numbers may be considerably higher.

Figure 1: typical composition of domestic wastewater [3]

After the treatment, the water must comply with the emission limit values before the discharge in a surface water body. These ELV are reported in detail both in the Italian law decree (as already explained above) and also in BREFs documents, that are valid in Europe, not only in Italy.
In particular, BREFs are BAT Reference documents, that describe the techniques present on the market and the relevant related environmental performances. BREFs are drafted and kept up to date by the European IPPC Bureau within the European Commission's Joint Research Centre. They contain the BAT, that stays for "Best Available Techniques", that represent the best technologies available in the market at industrial scale, that guarantee a low level of emissions to the environment. [4]

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DECISIONS

COMMISSION IMPLEMENTING DECISION (EU) 2016/902

of 30 May 2016

establishing best available techniques (BAT) conclusions, under Directive 2010/75/EU of the European Parliament and of the Council, for common waste water and waste gas treatment/management systems in the chemical sector

(notified under document C(2016) 3127)

(Text with EEA relevance)

BAT-AELs for direct emissions of TOC, COD and TSS to a receiving water body

Parameter	BAT-AEL (yearly average)	Conditions
Total organic carbon (TOC)	10-33 mg/l	The BAT-AEL applies if the emission exceeds 3,3 t/yr.
Chemical oxygen demand (COD)	30-100 mg/l	The BAT-AEL applies if the emission exceeds 10 t/yr.
Total suspended solids (TSS)	5,0-35 mg/l	The BAT-AEL applies if the emission exceeds 3,5 t/yr.

BAT-AELs for direct emissions of nutrients to a receiving water body

Parameter	BAT-AEL (yearly average)	Conditions
Total nitrogen (TN)	5,0-25 mg/l	The BAT-AEL applies if the emission exceeds 2,5 t/yr.
Total inorganic nitrogen (N_{inorg})	5,0-20 mg/l	The BAT-AEL applies if the emission exceeds 2,0 t/yr.
Total phosphorus (TP)	0,50-3,0 mg/l	The BAT-AEL applies if the emission exceeds 300 kg/yr.

BAT-AELs for direct emission of AOX and metals to a receiving water body

Parameter	BAT-AEL (yearly average)	Conditions
Adsorbable organically bound halogens (AOX)	0,20-1,0 mg/l	The BAT-AEL applies if the emission exceeds 100 kg/yr.
Chromium (expressed as Cr)	5,0-25 µg/l	The BAT-AEL applies if the emission exceeds 2,5 kg/yr.
Copper (expressed as Cu)	5,0-50 µg/l	The BAT-AEL applies if the emission exceeds 5,0 kg/yr.
Nickel (expressed as Ni)	5,0-50 µg/l	The BAT-AEL applies if the emission exceeds 5,0 kg/yr.
Zinc (expressed as Zn)	20-300 µg/l	The BAT-AEL applies if the emission exceeds 30 kg/yr.

Figure 2: BAT ELV for the main pollutants [5]

In Figure 2 are reported the ELV from the BAT that the water that exits from a WWTP has to respect before entering a surface water body.

2. Conventional WWTP for domestic wastewater

A conventional civil wastewater treatment plant is structured in an almost standard way, since domestic wastewater have a composition that can be defined as constant.
There are some pretreatments used to remove coarser impurities and suspended solids, then the water passes to the main units, to remove nutrients and dissolved substances. While water become cleaner, sludge remains apart and needs to be treated too. For this reason, WWTP also comprehend the sludge line that treat it to be reused or to recover some energy from it.
The plant situated in Castiglione Torinese, owned by SMAT, is the biggest WWTP in Italy, and it treats the domestic wastewater that comes from more than 3 000 000 population equivalent, that correspond to more than 260 million cubic meters of wastewater per year. Due to its size and its primacy, SMAT plant of Castiglione Torinese will be used in the following pages as an example of a typical wastewater treatment plant. [3]

2.1. Pretreatments

After the equalization basin, used to homogenize flow rates and concentrations and fix eventual problems due to occasional pollution, the first pretreatment that the water meets is the screening unit. The screening phase is useful to remove coarse solids, both for environmental reasons (to avoid the spread of microplastics) and to protect the plant from blockages and clogs. It consists of a simple screen inside a channel that blocks coarse bodies moved by the flow. To design the screens is important to take into account the water velocity, since it has to be fast enough to avoid sedimentation, but slow enough to not damaging the grids. The best range of velocities is between 0.6 and 1 m/s, that helps also to limit the head losses.
At this point, water goes into the sand and oil removal unit: sand has to be removed mainly to not compromise hydraulic and electromechanics parts of the plant, and the removal is really efficient. The unit is combined with oil removal, so the oil remains in the upper part of the basin, while sand precipitates. The system can also be aerated to avoid the sedimentation of the organic fraction. [3]

2.2. Primary sedimentation

At this point, the wastewater treatment enters into the core part, with primary sedimentation, that is used to remove suspended solids (not colloids). Sedimentation refers the physical removal from suspension. Maintaining a laminar flow, it is possible to calculate the sedimentation velocity with a balance between drag force, weight force and Archimede's force, as shown in the following formula:

$$v_s = \frac{\left[D_p^2 * (\rho_p - \rho_l) * g * C_c\right]}{18\mu_l}$$

Where D_p is the particle diameter, ρ_p and ρ_l are, respectively, particle and liquid densities, C_c is the friction coefficient and μ_l is the dynamic viscosity of the liquid.

To correctly design the sedimentation basin, the two components of the velocity have to be taken into account: in addition to settlement velocity, also the velocity of the water flux is important. In particular, the settlement time has to be shorter than the time in which the water passes through the basin, to be sure that the solid particles remain trapped at the bottom and don't reach the exit of the basin.

The most used configuration of sedimentation basin is the circular and dynamic one, that is shown in Figure 3.

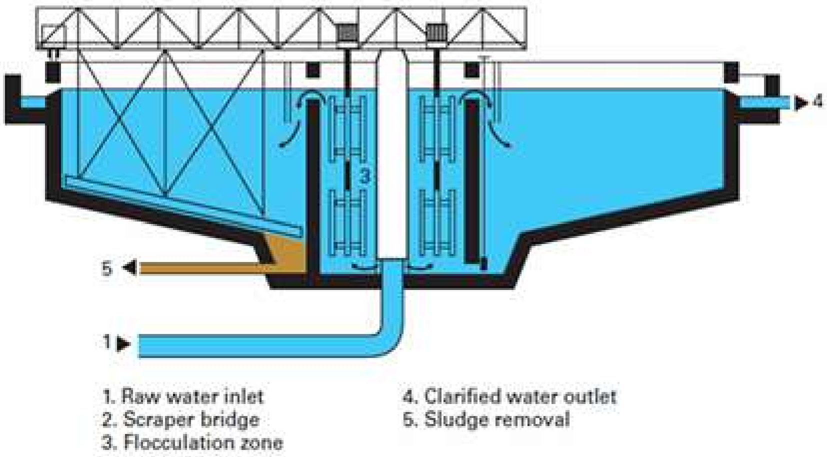

Figure 3: sedimentation basin [3]

In this first sedimentation unit, usually, the efficiency is pretty high and 90-95% of solids than can settle are removed (50-90% of the total suspended solids). Sedimentation is a simple process in which is relevant not only the density of the single particle, but also the overall number of particles and their interaction. [3]

2.3. Coagulation and flocculation

This part of the treatment is dedicated to the removal of non-settleable solids, also called colloids. Their removal is necessary since they make water turbid and coloured; furthermore, some of them are pathogens or they may adsorb toxic substances on their surface. Colloids have a diameter between 100 nm and 1 μm: for this reason, all forces are significative in their motion, and no one prevails (electrostatic forces, Van der Waals, etc). In particular, chemical coagulation consist of the modification of zeta potential, so the charges are neutralized and then a gelatinous mass is formed to trap or bridge particles, thus forming a mass large enough to settle. Flocculation, instead, is a physical-chemical treatment that helps with the formation of flocs, so that they can settle in an easier way, using a slow mixing. The velocity of the mixing must be taken under control, because it has to enhance collisions between particles, but it does not have to break the already formed flocs. Coagulation and flocculation can take place in a unique pit, called clarifier, in which there is also time for sedimentation to remove the flocs. If the two steps are placed in the clarifier, also the retention time has to be accurately designed, because they need different times when taken separately. The best option is to consider an average HRT. Usually, in wastewater treatment plants, these steps happen in the same pit together with primary sedimentation, so the efficiency of sedimentation can be higher; the sludges in this phase sediment thanks to physical-chemical interactions and they're called primary sludges.

There are lots of coagulant agents used in water treatment plants; the most common are shown in Figure 4.

Classification	Chemical Formula	Molecular Weight, g/mol	Application
Coagulants			
Aluminum sulfate	$Al_2(SO_4)_3 \cdot 14H_2O$	594.4	Primary coagulant
Sodium aluminate	$Na_2Al_2O_4$	163.9	Used with alum; provides alkalinity and pH control
Aluminum chloride	$AlCl_3$	160.3	Used in blends with organic polymers
Polyaluminum chloride (PACl)[a]	$Al_a(OH)_b(Cl)_c(SO_4)_d$	Variable	Primary coagulant
Polyaluminum sulfate (PAS)[b]	$Al_a(OH)_b(Cl)_c(SO_4)_d$	Variable	Primary coagulant, produced onsite
Polyiron chloride[c]	$Fe_a(OH)_b(Cl)_c(SO_4)_d$	Variable	Primary coagulant, produced onsite
Ferric chloride	$FeCl_3$	162.2	Primary coagulant
Ferric sulfate	$Fe_2(SO_4)_3$	400.0	Primary coagulant
Coagulant aids			
Activated silica	SiO_2	60.0	Coagulant aid used with alum during cold winter months
Sodium silicate	$Na_2O(SiO_2)_{3-25}$	242–1562	Coagulant aid, produced onsite
Bentonite	$Al_2Si_4O_{10}(OH)_2$	258	Used to provide nucleation sites for enhanced coagulation
Alkalinity and pH adjustment			
Calcium hydroxide	$Ca(OH)_2$	56.1 as CaO	Used to provide alkalinity and adjust pH
Sodium hydroxide	NaOH	40.0	Used to provide alkalinity and adjust pH
Soda ash	Na_2CO_3	106.0	Used to provide alkalinity and adjust pH

Figure 4: common inorganic coagulants, coagulant aids and pH and alkalinity adjusting chemicals used in WWTP [3]

In any case, the most suitable coagulant can be chosen after specific jar tests, since every water can have its specific needs. [3]

2.4. Biological treatments

Biological treatments exploit microorganisms to degrade the organic pollutants present in wastewater. There are a lot of microorganisms in water, and bacteria are the most common; they are unicellular, prokaryotic and they multiply by binary fission. Bacteria can have different shapes and dimensions. Bacteria use 3 main processes to degrade the organic substance (also known as biodegradable organic substance, bCOD), that is roughly indicated with the following brute formula:

$$COHNS$$

Differently, microorganisms in chemical reactions are indicated with:

$$C_5H_7NO_2$$

The first step is oxidation, or catabolism: in this phase chemotrophic bacteria destroy organic matter oxidating it to gain some energy. The chemical reaction that governs the process is the following:

$$COHNS + O_2 + biomass \rightarrow CO_2 + H_2O + NH_3 + energy + other\ final\ products$$

The following step is called synthesis, or anabolism: using the energy previously produced, bacteria generate other cellular material, so they are multiplying themselves (growth).

$$COHNS + O_2 + biomass + energy \rightarrow C_5H_7NO_2$$

The final process is endogenous respiration, described as follows:

$$C_5H_7NO_2 + 5O_2 \rightarrow 5CO_2 + NH_3 + 2H_2O$$

While degrading organic substances, the growth of the bacteria colony is not constant, but it follows different phases, shown in the figure below.

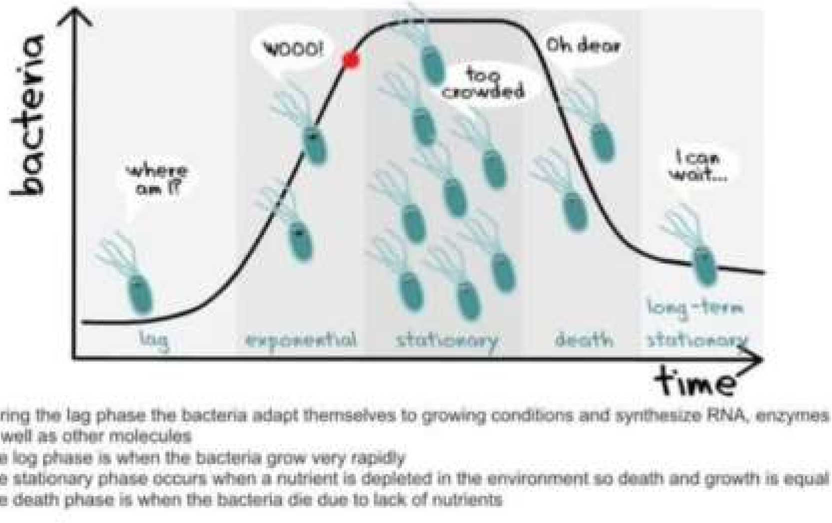

Figure 5: bacterial growth phase [3]

The first phase is called lag phase and is the one in which bacteria have to adapt to the environment. After that, the log phase starts, in which there is a lot of organic matter to degrade, so bacteria can growth almost unlimitedly. After some time, organic matter starts to be significantly reduced, so the bacterial growth becomes limited; this phase is called stationary, since newborns bacteria and dead ones are almost balanced. The last phase is the endogenous one: organic substance is almost absent, so bacteria try to survive but in the end they all die.

To design the aeration basins in which biological treatments can take place, a more quantitative interpretation of the process is needed, that takes also into account the kinetic of the reactions. Michaelis-Menten equation describes the reduction of the substrate of organic matter that is supposed to be degraded:

$$\frac{dS}{dt} = \frac{\mu_m}{Y}\frac{S}{K_S + S}X$$

Where S in the substrate concentration, measured in $\frac{g_{bCOD}}{L}$; X is the biomass concentration (bacteria), measured in $\frac{g_{VSS}}{L}$; μ_m is the maximum growth velocity of the biomass, also expressed as k*Y and measured in $\frac{g_{VSS}}{g_{VSS}*d}$; K_S is the semi-saturation constant, so the substrate concentration for which the removal kinetic is equal to the half of the maximum one (k), measured in $\frac{g_{bCOD}}{L}$; and Y is the biomass growth coefficient, measured in $\frac{g_{VSS}}{g_{bCOD}}$.

Monod equation, instead, describes the net bacterial growth, that is a balance between births and deaths:

$$\frac{dX}{dt} = \mu_m\frac{S}{K_S + S}X - k_d X$$

With k_d that is the bacterial decay coefficient, measured in $\frac{g_{VSS}}{g_{VSS}*d}$. [3]

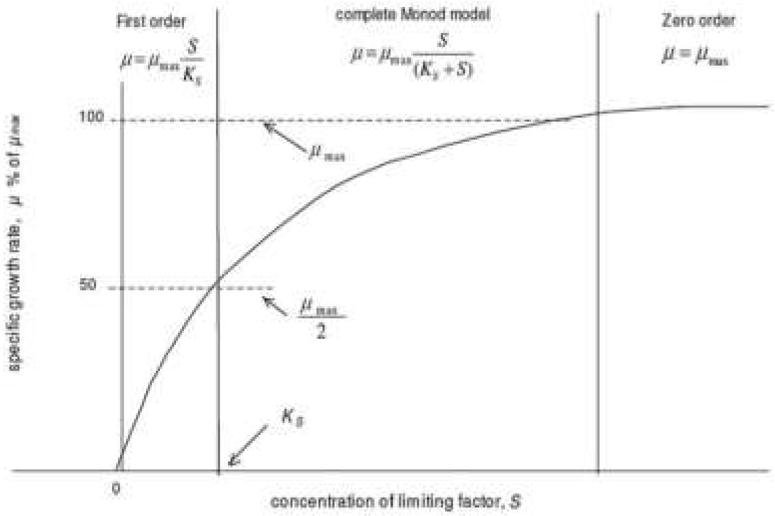

Figure 6: graphic representation of Monod equation [3]

At this point is possible to write an equation that represents the quantities of bacteria present in the activated sludge aeration basin inside conventional WWTP, that work to degrade organic matter in stationary conditions. The goal is to achieve a concentration of bacteria that is constant in the basin, while the substrate is continuously injected and degraded. To reach this condition, it is necessary to recirculate some of the sludge with bacteria that is accumulated at the bottom of the secondary sedimentation unit, placed just after the aeration basin. The mass balance on microorganisms' concentration becomes:

$$V\frac{dX}{dt} = Q_0X_0 - [(Q_0 - Q_w)X_e + Q_wX_R] + r_g'V$$

With:

$$r_g' = Yr_{su} - k_dX$$

And

$$r_{su} = -\frac{dS}{dt} = -k\frac{S}{K_s + S}X$$

In this way it is possible to find the concentration of substrate that remains in the treated water after a specific retention period, the sludge retention time (SRT).

$$S_e = \frac{K_s[1 + k_dSRT]}{SRT\ (\mu_m - k_d) - 1}$$

The concentration of bacteria in the basin, instead, is calculated as:

$$X = \frac{SRT}{HRT} * \frac{Y(S_0 - S)}{1 + k_dSRT}$$

Where HRT is the hydraulic retention time of water. [3]

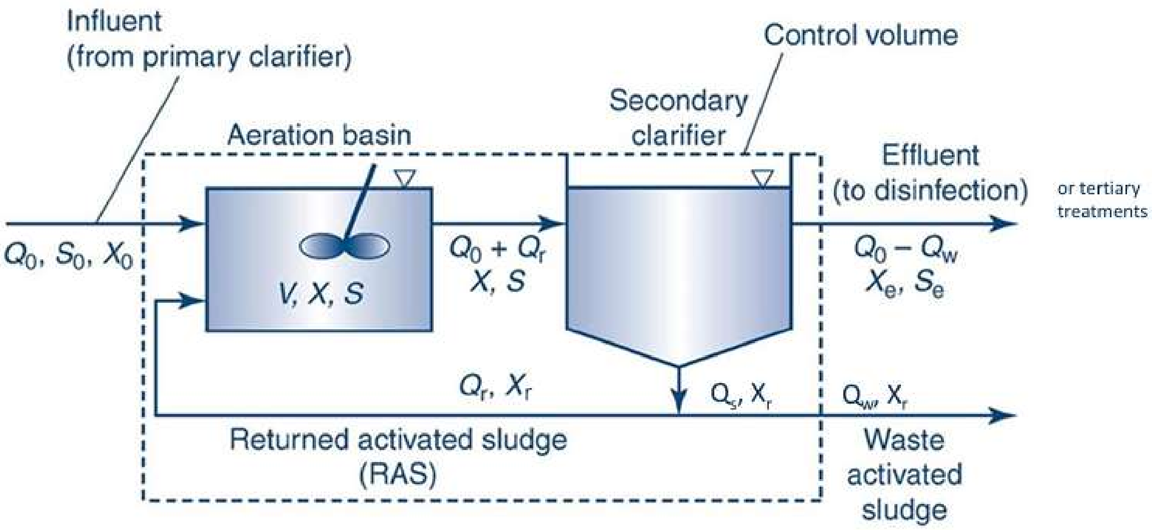

Figure 7: mass balance of the aeration basin [6]

Thanks to these balances and formulas, it's also possible to design the effective dimensions of the aeration basin (volume). In fact, from the following equation:

$$M = V * X = P_{MLVSS} * SRT$$

It is possible to find the mass of bacteria knowing SRT and P_{MLVSS}

$$P_{MLVSS} = \frac{YQ(S_0 - S)}{1 + k_dSRT} + f_dk_dSRT * \frac{YQ(S_0 - S)}{1 + k_dSRT} + Q[nbVSS]$$

that is the production of mixed liquor volatile suspended solids. In this formula, the first term is referred to the volatile solids, that are the main part of the living organisms (active biomass); the second term is referred to the accumulation of bacteria's cells inside the tank, while the third one is referred to the non-biodegradable volatile solids, and depends only to wastewater characteristics, not to microorganisms.
In the equation, f_d $[\frac{g_{VSS}}{g_{VSS}}]$ represent the amount of residual m.o. that remains after the endogenous respiration and can't be degraded by biological processes.
After that, the bacteria concentration X is known form the formula cited above, so the tank volume is the only unknown variable. [6]
Another variable that is relevant for the design of the aerated basin is the sludge organic load, that represent the effective quantity of substrate that bacteria need for feeding themselves, in a unit of time. [3]

$$C_F = \frac{F}{M} = \frac{Q * S_0}{V * X} \quad [\frac{g_{BOD5}}{g_{VSS} * d}]$$

In conventional WWTP the most used sludge organic load is between 0.2 and 0.3 $\frac{g_{BOD5}}{g_{VSS}*d}$, with a concentration of bacteria of 3-4 $\frac{kg_{VSS}}{m^3}$ and an HRT of 2 hours.
As already said, the biological process happens in aerobic conditions, so it is possible to evaluate the amount of oxygen needed by bacteria to oxidize the organic matter and to produce new biomass. Form a mass balance, the final formula for oxygen demand is the following:

$$OD = Q(S_0 - S) - 1.42 * P_{MLVSS,bio}$$

The factor 1.42 comes from the chemical reaction of endogenous respiration and corresponds to the grams of oxygen needed to oxidize 1 g of m.o., also known as chemical oxygen demand (COD). This value is useful to design the aeration devices that are needed inside the basin. [3]

2.5. Nitrification and denitrification

Nitrogen is one of the nutrients that is mostly present into domestic wastewater, and there are two prevalent forms: organic nitrogen and ammonia nitrogen. Referring to the two forms, together, it is possible to talk about Kjeldahl nitrogen; the reference parameter is Total Kjeldahl Nitrogen (TKN).

Nitrogen removal from wastewater is divided in three main steps: ammonification, nitrification and denitrification.

- Ammonification: in this step all the organic nitrogen undergoes hydrolysis and become ammonia nitrogen (ammonium ion).

$$NH_2CONH_2 + H_2O \rightarrow CO_2 + 2NH_4^+$$

- Nitrification: at this point, all the ammonia nitrogen is oxidized into nitrite and then nitrate; this step causes acidification of the water because of the release of H^+ ions. To complete the following reaction, 4.33 g of oxygen per gram of $N\text{-}NH_4^+$ are needed. The process is held by nitrous and nitro bacteria, that are different families with respect to the ones for the biological processes; they have a high oxygen demand and there is a limited cellular growth since they are autotrophic bacteria. The complete nitrification reaction is the one written below.

$$NH_4^+ + 2O_2 \rightarrow NO_3^- + 2H^+ + H_2O$$

  This step can also take place inside the aeration basin together with biological treatments, but in this case, in the oxygen demand formula, it's necessary to consider also the quantity of oxygen for nitrification.
- Denitrification: in the end, nitrates are reduced into gaseous nitrogen (N_2) through heterotrophic bacteria. This step must take place in a separate basin, because bacteria need to use the oxygen linked to the nitrate to reduce it; if some free oxygen was present, the denitrification reaction would stop.

The most common way to introduce nitrogen removal in the line of conventional WWTP is the pre-denitrification: the basin in which denitrification takes place is upstream with respect to the aeration basin for nitrification and removal of organic matter. This configuration allows to save oxygen inside the aeration basin, since the oxidation of organic matter starts in the anoxic tank during the denitrification of nitrates: only 2.8 g of oxygen per gram of NO_3^-N needed (-62.5%). Ammonia nitrogen will pass undisturbed; then, it will be oxidized in the aeration tank and a fraction of treated water undergoes recirculation, to let the new nitrates pass into the anoxic tank, to become gaseous nitrogen, that can enter the atmosphere. The fraction that recirculates is calculated to be sure that the amount of water that exits the system already respect the nitrogen law limits. This configuration is a little bit difficult to manage, but it's self-sustaining, bacteria for the denitrification don't need extra carbon to work, but they can use the one still present in the sludge. In this way, it's also easier to manage the acidification problem. [6]

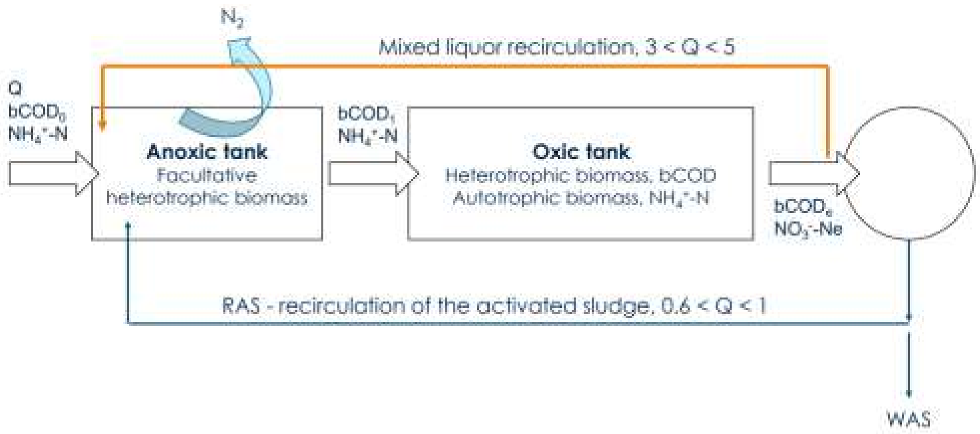

Figure 8: pre-denitrification configuration [6]

To design in the right way the aeration basin of the WWTP, taking into account also nitrogen removal, the amount of biomass has to consider also the autotrophic bacteria for the nitrification phase (AOB). These microorganisms have their specific equation for the net growth rate, where the amount of ammonia nitrogen is a limiting factor, and OD is another limiting factor. The new formula for the P_{MLVSS} become:

$$P_{MLVSS} = \frac{YQ(S_0 - S)}{1 + k_d SRT} + f_d k_d SRT * \frac{YQ(S_0 - S)}{1 + k_d SRT} + \frac{Y_A Q(NO_x)}{1 + k_{dN} SRT} + Q[nbVSS]$$

Where the third term is the one referred to AOB bacteria for nitrification process. [6]

2.6. Phosphorus removal

Phosphorus is another prevailing nutrient that can be found inside urban wastewater and that must be removed. The removal can happen chemically or biologically; the first one is the most diffused, since biological removal is more challenging.

Chemical removal of phosphorus consists of the precipitation of dissolved species thanks to chemical reagents such as ferrous or aluminium salts or hydrated lime. The precipitates that result from the reactions are iron or aluminium phosphates or hydroxyapatite. There are three main operative schemes that can be used inside WWTP: pre-treatment, simultaneous treatment or post-treatment, where reagents are injected, respectively, before primary sedimentation, inside the aeration tank, or after the secondary sedimentation unit. Post-treatment is the more efficient scheme, since TSS that remains are not a huge number and the amount of reagent needed is reduced; on the other hand, a third sedimentation unit is needed. The simultaneous treatment, instead, can be critical because the two processes may conflict with each other due to the eventual coagulation of suspended solids.

Biological removal of phosphorus needs specific bacteria, since with the previously described biological processes P removal is minimal. The specialized bacteria are called PAO, phosphorus accumulating organisms, and they contain a small amount of P inside (6-7%). The process consists of two tanks, one with anaerobic conditions and one aerated. In the first step bacteria are stressed until the point at which they barely survive, since they are aerobic. Thanks to this situation fermentation of PAO starts. Introducing in the environment some volatile fatty acids (VFA), that are easily biodegradable, PAO use them to produce PHB (polyhydroxybutyrate). During this process, bacteria take energy from themselves, releasing the accumulated phosphorus in the water. At this point, bacteria are transferred in the aeration tank, where they can find optimal conditions to survive: they can find oxygen and organic matter from the previously produced PHB. New PAOs are generated by bacteria activity, so they accumulate phosphorus in their formation, removing it from wastewater. In this second step bacteria are 20 times more active than the previous one, so the net result is the removal of P, even if in the first anaerobic tank some phosphorus is released. This process is very efficient, but it’s challenging to combine it with the other biological processes; in particular, the best configuration in the one with the independent recirculation pathways, as shown in the last section of the figure below. [3]

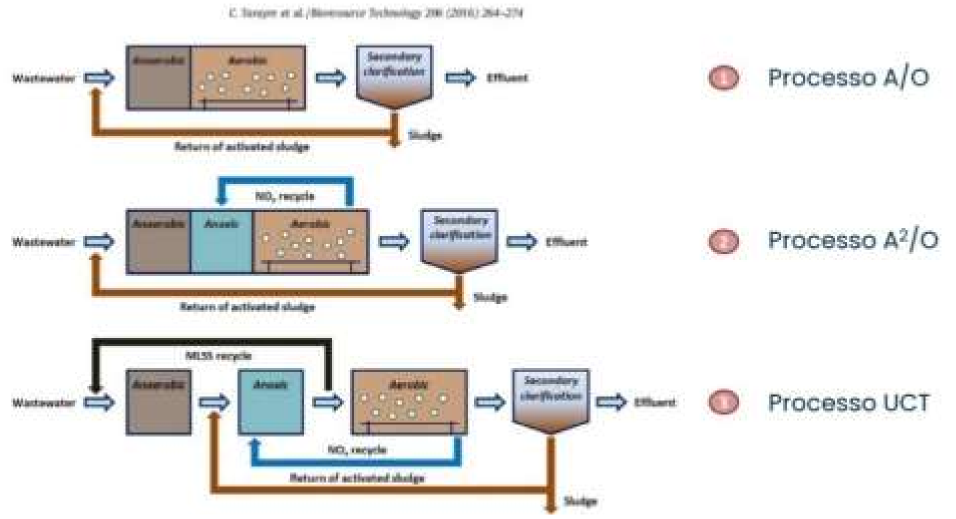

Figure 9: enhanced biological phosphorus removal configurations [3]

2.7. Refinement processes

In some WWTP such as the one of Castiglione Torinese, is present a refinement unit downstream the main ones, that consists of filtration in granular media. This section is useful to remove fine suspended solids that cannot be removed by secondary sedimentation. In this unit, is also possible to integrate disinfection: in plants of these huge dimensions, disinfection is used only in particular situations, if bacteria concentration increases for some problems inside the plant. In regular functioning of the plant, disinfection is not carried out because due to the high flow rates treated, the amount of disinfectant needed would be too high, causing a pollution source to the river in which treated water is discharged. This section is particularly useful if the final destination of water is the reuse: in this case the parameters are stricter, and disinfection is mandatory. Reuse of water would be a nice improvement for the plant, aligned with the new European regulations described in the previous chapters; furthermore, it would contribute to the water demand being a constant source, limiting the environmental footprint of water supply. [3]

3. The innovation of Sequencing Batch Reactor technology

The Sequencing Batch Reactor technology is an innovative way to treat wastewater which complements the much popular activated sludge process (ASP) described in the previous chapter. SBR technology switches from a continuous flow to a batch system that evolutes in time and not in space; this helps with process flexibility and introduce alternative process controls, useful because wastewater contains always more specific pollutants and micropollutants. If the SBR process is effectively automated, it can save more than 60% of the operating expenses needed for the conventional ASP and can achieve high effluent quality in a short aeration time. [7] In particular, SBR is more efficient in BOD removal, reaching 90%, with respect to the 60-95% of ASP; also, the concentration of suspended solids can reach values under 10 mg/L. [8]
In areas that are densely populated, SBR is the favourite solution because it has a lower requirement of area with respect to the conventional technology; as a consequence, even the manpower for operation and the energy demand are reduced. The main aspect is that is possible to select the best operative conditions for each stage of removal, to maximise each of them. [7]

3.1. The process

The treatment process in the SBR is composed by 5 stages, that take all place in a single tank in different times, creating a sort of cycle. The 5 phases, or operating modes, are called Fill, React, Settle, Draw and Idle and each of them can have a different retention time. [7]

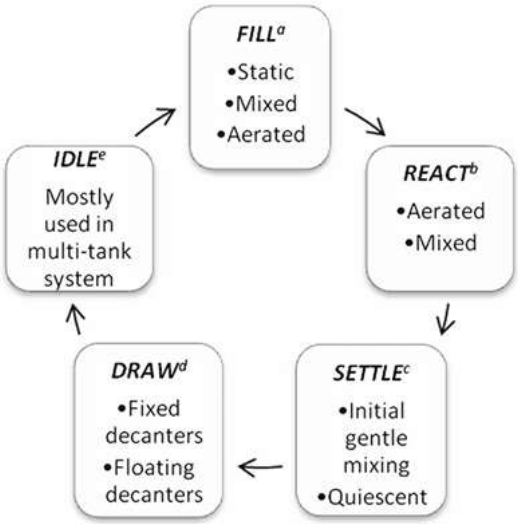

Figure 10: phases of SBR operation cycle [7]

The first step of the cycle is the Fill phase, in which raw wastewater enters the system and comes in contact with microorganisms left from the previous cycle. Depending on wastewater characteristics and the target removal of organics and biological nutrients, it is possible to perform a static fill, a mixed fill or an aerated fill. Static fill consists in the injection of wastewater in the environment with the already present biomass without mixing; this situation is similar to the plug flow one, with a high food to microorganisms ratio (F/M), also known as sludge organic load. This configuration promotes the formation of floc-forming bacteria in spite of filamentous ones, that have a worst effect in the environment, since they tend to remain suspended. Furthermore, static fill enhances PAO activity for the removal of phosphorus. [7] The duration of the fill phase is usually 25% of the entire cycle duration; changing the length of this stage may alter the productivity of the reactor. [8]

During the React phase, the biological degradation of organics takes place, but it is often designed to enhance the nutrient removal as well. In this step the air injection is a crucial parameter, since it can generate both aerobic, anoxic or anaerobic conditions. This stage can be aerated or mixed: aerated react consist of letting aerobic reactions, such as nitrification, occur after the aerated fill; mixed react, instead, consist of creating anoxic conditions for nitrogen removal (denitrification) and anaerobic conditions for phosphorus removal, after the aerobic phase. This control of biological reactions through aeration regulation is one of the main strengths of the SBR. [7] This second phase is the longest, since it uses almost the 50% of the whole cycle time; the change in the duration of this phase determine its efficiency. [8]

The following step is the Settle phase, in which the reactor acts as a clarifier: in batch mode, sedimentation is much more efficient, since there is not the flow velocity that contrast particle's settlement. [7] The settle phase lasts between 0.5 and 1.5 hours. [8]

At this point, during Draw phase, the clarified supernatant is discharged in a decanter, that can be fixed or floating. [7] During this mechanism is important to avoid floating material discharge; this phase lasts typically 45 minutes. [8]

Idle phase, in the end, is the period of time that occurs between draw phase and the next fill phase: this is useful if there are more SBR working in parallel, as a buffer in time. This is also the phase to manage the sludge, removing some excess or mixing it, if the operating conditions need it. [7]

The high concentration of microorganisms reduces a lot the treatment duration. The problem of sludge bulking, that consist in a high suspended solid content, is reduced in SBR with respect to ASP, as already underlined, but there are some cases in which, also in SBR configuration, the quantity of suspended solids is too high. To overcome this problem, is possible to provide special bioreactors, selectors, that enhance the floc-forming bacteria growth over the filamentous bacteria growth. [7]

3.2. Nitrogen removal

Biological nitrogen removal in SBR technology is again based on two steps, nitrification and denitrification, as described in the previous chapters. During nitrification step, organic nitrogen is converted into ammonia nitrogen through an oxidation reaction, that is composed by two steps, from organic nitrogen to nitrite and from nitrite to nitrates. The reaction is carried out by chemoautotrophic bacteria and catalysed by two families of autotrophic bacteria. This step requires aerobic conditions.

After that, denitrification takes place, in anaerobic conditions thanks to heterotrophic bacteria; the reaction generates gaseous nitrogen that can leave the aqueous phase, starting from the nitrates previously produced.

The parameters of the reactions that is important to control are temperature, pH, dissolved oxygen and solid retention time (SRT). From literature is find out that the optimal range of temperature for nitrification is between 25 and 35 °C, while the optimal concentration of DO to obtain the maximum nitrification rate is beyond 2 mg/L. Increasing the concentration of bacteria responsible for nitrification is a way to increase the rate of the reaction; to do this is necessary to increase the SRT, lowering the flow rate of waste activated sludge. If pH goes beyond the range 7.5-9.8 the nitrification rate drops to half of the optimum. [7] In this phase carbon dioxide is used as carbon source and its stripping causes an increase to pH values. [8]

On the other side, when denitrification starts, DO must drop below 0.5 mg/L, and the optimum pH range is between 6.5 and 9. [7]

SBR is a valid technology that allows to perform simultaneous nitrification-denitrification, thanks to the possibility of changing the aeration conditions during the same treatment cycle. The key parameter, apart from the others previously cited, is the C/N ratio: with the correct value, that is find out to be 11.1, it is possible to obtain the total COD and ammonia removal. If the ratio is lower, instead, may cause unbalanced conditions that result in a poor denitrification performance. The reaction rate is lower with respect to separate basins designs, because just a percentage of the total biomass is involved in nitrification or denitrification steps, but the efficiency is really close to the maximum. [7]

The creation of anoxic conditions right after the aerobic ones may result in a need of carbon supplementation; this aspect can be reduced at its minimum with optimized regime control of the process in term of dissolved oxygen and oxidation-reduction potential (ORP). [7]

Another way to perform nitrogen removal from wastewater can be the partial nitrification and denitrification process, also known as short-cut nitrogen removal process. This technique consists of the partial nitrification followed by the denitrification and allows to reduce the aeration requirement by 25% and the external carbon supplementation by 40%, with respect to the simultaneous nitrification-denitrification process. In this way, also the energy demand is reduced. Short-cut nitrogen removal also allows higher denitrification rate and lower waste sludge production; it demonstrated good results when a low C/N ratio is present. [7]

There is another way that can be used to remove nitrogen from wastewater with SBR technology, that is the anammox process: it utilizes anaerobic ammonia oxidizing bacteria, that work completely in anaerobic conditions, so there is no need to change the aeration conditions inside the reactor during the process. Furthermore, even the additional source of carbon is not needed anymore; in fact, is demonstrated that anammox bacteria work well in low COD conditions and with low C/N ratios. The process firstly consists of the half oxidation of ammonia nitrogen into nitrite, then the nitrite reacts with the rest of the ammonia nitrogen to generate nitrogen gas and nitrate. The disadvantage of this process is the slower rate of the reaction due to the slower growth rate of this kind of bacteria, with respect to the simultaneous nitrification-denitrification. The two stages of the reaction have to be strictly controlled inside the SBR system, with a particular attention to the ORP; as a conclusion, the best option is to perform an interval feeding with an interval aeration. [7]

3.3. Enhanced biological phosphorus removal

The classical way to biologically remove phosphorus from wastewater is using PAO (polyphosphate accumulating organisms), that accumulate phosphorus during an aerobic phase, after being stressed during a previous anaerobic period, as already mentioned in the previous chapters. Performing this type of P removal in SBR, taking advantage from the possibility of changing aeration conditions, allows to reach a really high efficiency, until 90%. This result is a huge improvement with respect to the conventional technology, that reaches a maximum efficiency of 10-20%. [7]

Glycogen accumulating organisms (GAO) may compete with PAO in their metabolism, but they don’t have any contribution to phosphorus removal; for this reason, is important to create optimal conditions inside the reactor to avoid GAO proliferation. Temperature is one of the parameters that has to be controlled, and cold ones favour PAO growth. High values of pH are good for PAO over GAO, with optimum values between 7.2 and 8. In general, if the ratio COD/P is high (> 40) a low effluent P concentration can be achieved; in particular, easily biodegradable COD, such as VFA, helps the PAO during their activity. At the end of the process, PAO and the accumulated phosphorus are removed as waste sludge. [7]

Some families of bacteria can be added to the sludge to enhance PAO's activity; Acinetobacter lwoffii, for example, increase the reaction rate, while Halobacter halobium, that are salt tolerant organisms, improve nutrient removal performance in saline wastewater. [8]

3.4. Simultaneous removal of nitrogen and phosphorus

Inside a Sequencing Batch Reactor, it is expected that nutrient removal, such as nitrification, denitrification and EBPR processes, take place simultaneously; this means that the control of the system must be really precise, otherwise may generate the failure of the treatment. There are some intermediate products or reactant that may interfere with some bacteria; for example, nitrite and its acidic counterpart provide disadvantages to PAO growth over GAO in the phosphorus removal process. 2 mg/L of nitrite are sufficient to inhibit PAO's activity. Some studies underlined the poor phosphorus removal in environments rich in nitrates, because nitrates disrupt anaerobic conditions. Furthermore, denitrifiers and PAO both require organic substrate in anaerobic conditions, so they enter in competition with each other. On the other hand, some PAO called denitrifiers PAO, have the ability to accumulate phosphorus using nitrate as a terminal electron acceptor, so they can be useful for the simultaneous removal of nutrients; they also have a lower cell yield and sludge production. Some others PAOs are able to use both nitrate and nitrite, in addition to oxygen, as electron acceptors; they generate a good removal of nutrients using an anaerobic-aerobic-anoxic-aerobic system or an anaerobic-aerobic-anoxic system, as discovered in many studies. [7]

When N and P removal are requested together, a longer cycle time is needed and is necessary to operate with minimum sludge recycle ratio. Moreover, the alternate aerobic, anoxic and anaerobic conditions have to be managed carefully, with specific durations. Nutrients removal can be managed also modifying pH and ORP: in particular, the first one plays a crucial role during the aerobic phase, while the second one during the anoxic phase. Another crucial parameter is the oxygen uptake rate (OUR); with these three parameters is possible to catch the time at which nitrification, carbon oxidation and denitrification end. Being able to control this time allows to increase reactor productivity also thanks to a good adaptation of load variation. More than 98% of nutrients removal has been obtained in this way from a piggery wastewater. Of course, nutrients removal is affected by the characteristics of the inlet wastewater. [8]

3.5. Strategies to control the process

One of the main advantages of SBR over conventional treatment processes is that it can be used both in steady and unsteady conditions; the high level of control guarantees a better removal of COD and nutrients, low energy consumption and a better control of filamentous bacteria. [7]

Classical SBR control is done without adaptation of cycle length; real time control, instead, can provide better flexibility for adaptation in different conditions. Real time control requires feedback at start and end points of every biological reaction, while continuous monitoring of direct parameters is not possible nowadays, but the control can be performed using pH, DO and ORP. [7]

ORP is directly correlated with nitrification rates and biological reactions in anoxic conditions. Normally, ORP is negative during anoxic phase and positive in the aerated one, assuming a range of values between 0 and 50 mV; in the anoxic phase, instead, the range of values is between 0 and -300 mV, with a continuous dropping profile during time. When denitrification phase ends, the ORP profile presents a steep drooping, also called nitrate knee; this is the signal that allows to switch from anoxic phase to the next one. [7]

The pH profile is useful to understand the differences between nitrification and denitrification, since in the first phase pH decreases, while increases during the second one. In particular, during nitrification some acids are produced, that lower the pH; when all the ammonia has been oxidised, the acids production stops, so the pH curve stabilizes, creating what is defined ammonia valley. During denitrification pH rises continuously until it reaches a maximum called nitrate apex, reached at the end of the denitrification reactions; in fact, the peak corresponds to the nitrate knee in the ORP profile. Sometimes the background alkalinity typical of wastewater provides a buffering capacity that minimizes noticeable pH variations. [7]

In aerated phases, degradation of COD implies the consumption of dissolved oxygen; since there is a constant supply of DO and COD level is fast decreasing, the overall profile of DO is increasing in time. The formation of nitrites requires just 25% of the total OD required during nitrification, so when all nitrites are formed the profile of DO rises sharply; this inflexion is called DO breakpoint. This breakpoint corresponds to the ammonia valley of the pH profile. At the end of the aerated phase, DO profile falls to zero and remains constant until the rest of the anoxic phase; for this reason, it is not a good parameter to control denitrification. [7]

Another parameter that is becoming popular is the oxygen uptake rate (OUR): it corresponds to the DO consumption per unit time in unit volume of the reactor, and it

can be evaluated from DO measurements through a calculator. This parameter is useful to control short-cut nitrification process and enhanced biological phosphorus removal. The end of the phosphorus uptake process is clearly visible because the derivative of OUR reaches a breakpoint, from negative to positive values. As the DO, of course, it is not representative for anaerobic phases. [7]
Thanks to these characteristic profiles, it is possible to use algorithms that change the aeration conditions inside the SBR when the curves reach some specific values, optimizing energy consumptions. [7]

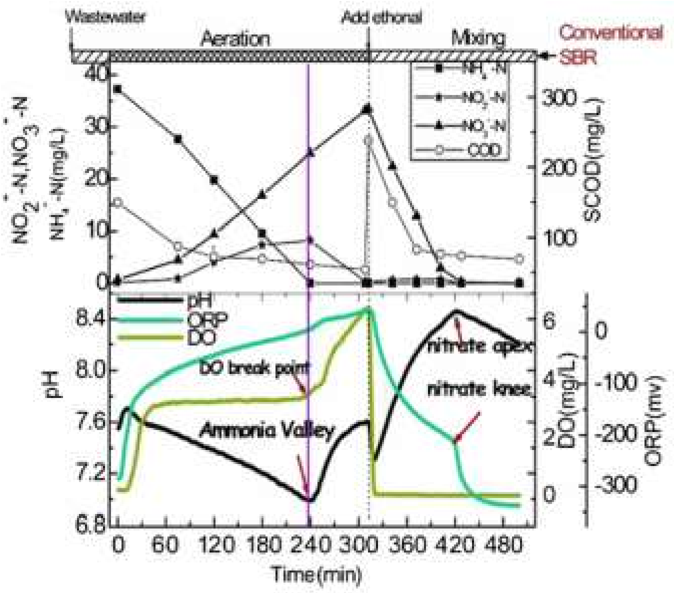

Figure 11: variation of DO, pH and ORP values and nitrogen compounds during nitrification and denitrification processes [7]

3.6. Different operation modes

Design and operation conditions of the SBR have to take into account the requirements of the biological processes, to ensure the quality of the treatment. The optimization of the process must consider the cycle time and the fill strategy, that involves influent characteristics. In particular, non-aerated fill mode guarantee better results if the influent has a low phenol concentration, because it inhibits filamentous bacteria growth. On the other side, aerated fill is better for higher phenol concentrations. [8]
It is known that the best aeration rate that provides the optimal performances of the reactor is 0.8 L/min: it provides a removal of COD of 97%, nitrogen removal of 94% and phosphorus removal of 97%. [8]
Some studies demonstrate that longer feeding times guarantee a better degradation of toxic compounds, especially if mixed; in fact, at shorter times, only some toxic species

are degraded, with the inhibition of the degradation of the others. The best results are obtained using step feeding. [8]
Different durations of aerobic and anaerobic phases within a single cycle can modify the reactor efficiency; the findings of some studies concluded that removal of nitrogen and COD decreases if the organic loading increases or if HRT decreases; these two parameters are really important for SBR efficiency. It was also demonstrated that increasing HRT improves the organics removal, in particular with mixing and aeration conditions. [8]
The number of operating stages is another aspect that can affect the system efficiency; studying three, four and five steps, was found out that the five step process (An/Ax/Ox/Ax/Ox) guarantee the highest nutrients removal: 94% for COD, 90% and 64% for ammonia nitrogen and nitrates and 57% for phosphorus. If the organic load is particularly high, nutrients removal can be negatively affected, especially if the toxicity level is high too. Wastewater with high C/N ratio are treated well using intermittent feeding and sub-cycles of anoxic and aerobic periods. [8]
The most expensive part of wastewater treatment is, without any doubt, sludge management; a good solution can be, for this reason, trying to reduce the amount of sludge produced, and SBR technology can accomplish with this aspect, increasing sludge age. Another way to reduce the sludge produced is increase the quantity of dissolved oxygen. [8]

3.7. Variants of SBR technology

With the developing of the research on SBR technology, different variants of the classic system have been created. An example is the Cyclic Activated Sludge System (CASS), that consists of a single basin with a variable volume which can operate in alternating mode. It works as a plug flow reactor in the initial zone, then there is a completely mixed zone where aeration changes can take place. The activated sludge is recirculated inside the reactor. The plug flow system placed upstream allows the mixed tank to perform a faster, stable and uniform metabolic activity. The system is almost indifferent to variations of organic concentration or flow rate and perform better with respect to the classical technology concerning the simultaneous removal of nitrogen and phosphorus.[7]
Another example is the UNITANK technology; its configuration consists of a single tank divided into three compartments in series. There is not recirculation of the sludge between the compartments and all of them have an aeration system. A cycle inside this type of reactor consists of two stages with three steps each, performed symmetrically.

This technology is really efficient in nutrient removal and, thanks to its simple structure, require less land occupancy and is most cost-efficient. [7]
A further enhancement of the standard SBR is the Intermediate Cycle Extended Aeration System (ICEAS), which principal characteristic is the continuous inflow of wastewater. Variable inflows are managed by a distributor box that splits the flow between various tanks to avoid overloading. There is a pre-react zone with high F/M ratio that acts as a selector, then the main-react zone operates in three modes, aeration, settle and draw. The equal loading of all the basins simplifies the control of the process and reduces capital costs. [7] This type of technology is used in Italy, near Verona: it is the first example of Italian WWTP that utilizes an SBR reactor instead of the conventional scheme. The plant was opened in 2020 and works with a load of 11250 p.e.; this type of technology allows to reduce the land use and the energy consumption, especially due to the elimination of pumps for the recirculation of the sludge. [9]
Hybrid SBR systems are another option, that guarantee carbonaceous oxidation and nitrification in the same tank. Porous biomass carrier SBR is an example of hybrid technology where porous biomass represents the support; the efficiency has been improved using alternate anoxic and oxic phases and high biomass level. Moreover, plastic media at the bottom of the reactor increased removal efficiency and sludge quality. Other hybrid media used are powered activated carbon and polyurethane foam.[8]
Another solution is the sequencing batch biofilm reactor: the attached-growth biofilm work as a sort of sink for nutrients and inorganic materials, being able to temporarily store substrate during peak loading situations and use it when the influent loading is scarce. The strength of this reactor consists of high concentration and stable activity of biomass. [8] Biofilm formation happens at a solid-fluid interface; the selection of support material depends on the type of wastewater to be treated. [7] The utilization of materials to increase the surface area for biofilm formation, such as plastic media or flexible fibre bundles, allows to increase the removal efficiency.[8] A disadvantage of this technology is that is unsuitable for high TSS influents. [7]
Another interesting application is a modified SBR with bio-floc technology: bio-floc are macro aggregates of microorganisms able to accumulate nitrogenous compounds and convert it into microbial proteins. [7]
A further example is the anaerobic SBR, where the reaction phase is carried out totally in anaerobic conditions, producing methane and carbon dioxide. Its main advantages are simplicity, good quality control of the effluent, flexibility and high biogas yield. On the other hand, a good mixing is required and can be achieved by recirculating the liquid or the gas or mechanically; this is due to the lack of homogeneity in the reaction medium.[8]

GAC-SBR is a technology developed for the removal of volatile, semi-volatile and non-volatile organic contaminants, through physical adsorption on granular activated carbon. Even at low HRT, the combination of adsorption and biological treatment allow to reach a good removal efficiency. [8]

3.8. SBR and salinity

Salinity can be a popular characteristic for wastewater, both industrial and domestic ones, especially in coastal regions; this is due to the fact that seawater is utilized as source of water to face the shortage of fresh water. Some salty residues reach wastewater treatment plants: for this reason, is important to evaluate if salinity affects the treatments that clean the wastewater, to be sure that the quality of the effluent water is not compromised.
A Chinese university tried to answer this question studying the variation of the effluent quality of an SBR, evaluating different salinity conditions of wastewater, from neutral, to saline, to hypersaline. Wastewater is considered saline until a salinity of 10 g/L; above this value it is called hypersaline wastewater. [10]
The experiment consists of comparing two SBR with a volume of 2.5 L; one is the control reactor, while the second one evaluates different influents with a salinity that gradually increases. The values of salinity evaluated are 0, 5, 10, 15 and 20 g/L. Three cycles per day for each reactor were operated and effluent samples were collected for the analyses, after reaching the stability. From the results, it is clear that stability takes more time to be reached, increasing the salinity. The removal rate of COD, ammonia nitrogen and phosphorus decrease with the increasing of salinity, as shown in Figure 12. [10]

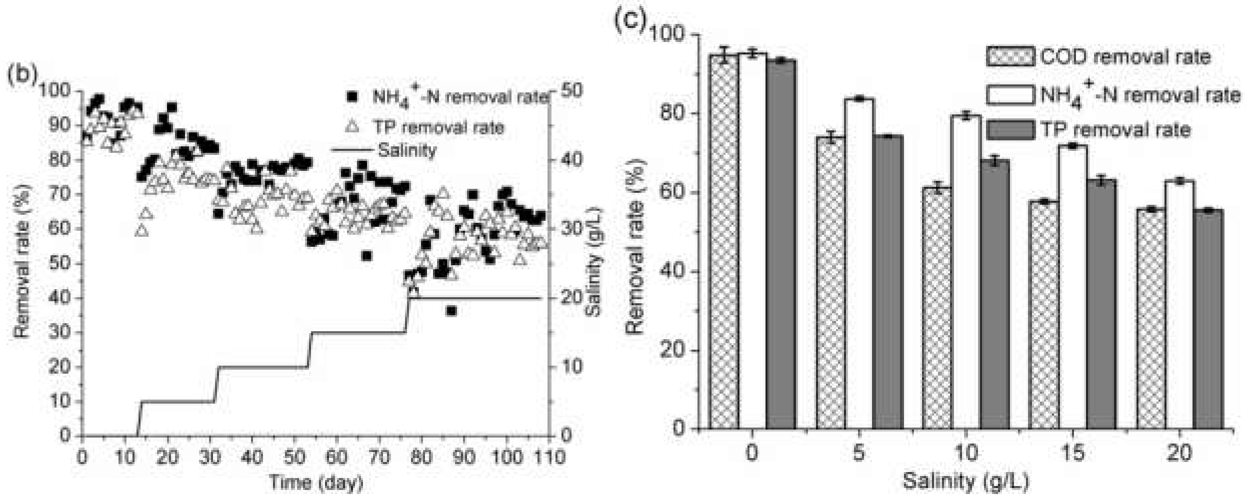

Figure 12: variation of removal efficiency with salinity [10]

In particular, it can be noticed that the COD removal rate gradually decreases while salinity increases, but the trend is steeper at the beginning; at higher salinity values the removal rate of COD varies less. Furthermore, can be said that phosphorus removal is more affected by salinity with respect to nitrogen removal; so, nitrobacteria and denitrifying bacteria are probably more salt-tolerant than PAO. All the bacteria show a greater inhibition when their tolerance to salinity is exceeded; in particular, at hypersaline conditions a further decreasing is evident. [10]

Going more in detail, results show that salinity affects, in particular, the aerated period of the SBR cycle: this means that the nitrification is inhibited, especially for hypersaline wastewater. This may be due to the fact that salt influences the dissolution of oxygen in the liquid phase. Increasing the duration of the aerated phase may partially overcome this problem. [10]

Furthermore, the sludge volume index (SVI) was measured to evaluate the sedimentation performance of the SBR with the salinity variation. Increasing the salinity, SVI value decreased from 110 to 60 mL/g: this means that the sedimentation process of the sludge was better. The reason may be that salt favours the formation of flocs in the sludge. The extracellular polymeric substances (EPS), that are products of microorganism's metabolism, increase with salinity: this can increase the resistance of activated sludge to toxic substances and protect the cells from not favourable environments, such as saline ones.

However, salinity inhibits the dehydrogenase (DHA) activity; DHA is an enzyme that reveals the degradation activity of bacteria, so it can be said that salinity affects the activity of the sludge, reducing also the diversity of microbial community. [10]

A possible way to enhance the removal efficiency for saline wastewater is to use salt-tolerant bacteria. [10]

4. Laboratory tests

4.1. Setup and initial conditions of the experiment

The experimental section of this research consists of analysing the behaviour of a laboratory-scale Sequencing Batch Reactor. The configuration comprehends, first of all, a tank with a volume of about 20 litres; inside the tank there are 6 air diffusers and 2 mixers to homogenise the conditions and parameters during the aerated phase. Moreover, there is a sensor that measures continuously the dissolved oxygen and the temperature, then 3 pumps, one for the charging phase, another one for the discharging phase and the last one connected to the air diffusers to provide air. The discharging is carried out through a floating system that allows to capture water always at the free surface, to avoid the suction of the sedimented sludge. Connected to the tank there are also 2 level sensors, used to understand when it’s necessary to start and stop charging and discharging phases.

All these devices are interfaced with an Arduino, which acquires all the signals and sends them to a computer. A Matlab code controls the system by processing the data and sending commands to the devices. Thanks to this automation done by the code, it is possible to let the reactor work continuously even if there are no operators in the laboratory, for, at least, a few days.

In this preliminary phase of the experiment, the wastewater treated in the reactor is not real, but is synthetically created in the laboratory, then stored in a tank with a volume of 100 L and connected to the charging pump, to be available during charging phase.

To create wastewater was used tap water with a source of simple organic matter, in form of readily biodegradable COD. Nutrients, such as nitrogen and phosphorous, have been tested just during the last days of experiments, so the results regarding them are not so significative. The COD source used were sodium acetate trihydrate and then sugar, to face the costs of acetate, since the amount needed was consistent. The objective was to insert a concentration that corresponds to 500 mg/L of COD, that is an average amount that is usually present in domestic wastewater. The quantity of COD correspondent to the compounds used, is calculated as the amount of oxygen needed to completely oxidate the molecule. In this way, it was found out that 500 mg/L of COD corresponds to 1063.8 mg/L of acetate and to 445.3 mg/L of sugar.

The source used for nitrogen and phosphorous was diammonium hydrogen phosphate (DAP); the amount needed was evaluated using a ratio of COD/N equal to 10, that is a good estimation of real wastewater quality. Following this ratio, being the amount of COD around 500 mg/L, the amount of nitrogen is around 50 mg/L, that corresponds to 235.9 mg/L of DAP. Phosphorous is indirectly inserted through the DAP.

Figure 13: laboratory setup

4.2. Logic of control of the system

Entering more in detail with the code, the process is divided into 4 phases; in fact, the Idle phase is not considered, since the analyses are carried out only in one reactor. The process starts into React or Biological phase, that is called, for simplicity, phase 1. The pumps for charge and discharge are off, while mixers are on; the aeration pump is controlled by a hysteresis logic: if the DO is lower than 2.0 mg/L the code switches on the air pump and maintains this condition until DO reaches 2.4 mg/L. In this way, microorganisms always have a sufficient amount of oxygen to degrade the organic substance at high rates. This velocity is measured by a parameter called Oxygen Uptake Rate (OUR): it is evaluated only in periods where the air pump is off, to be sure that the difference in the DO values is only due to the biological activity. In particular, the OUR is calculated starting from the curve of the DO values measured in the selected period, between 2.4 mg/L and 2.0 mg/L, to catch the oxygen consumption. The code takes the points inside this range, being sure that at least 5 points are present, and calculate the linear regression of them. The OUR corresponds to the slope of the line found, with the sign changed, since DO is consumed so the line has a negative slope. The quality of the interpolation is evaluated too, using the R^2 factor.

Using this logic, one OUR value is calculated for every time period in which the air pump is off, so the result is an evolution of the OUR in time. Since the goal is to create inside the reactor the best condition for bacteria's work, at the beginning of phase 1 OUR is really high; for example, after one month of activity, the colony is grown a lot and the maximum OUR reaches 150 ppm/h. During time, OUR decreases, since the organic matter inside the tank decreases; in fact, as written before, the degradation of the substrate in time follows the Michaelis-Menten equation. In this preliminary phase of the experiment, only the aerobic reaction is studied, so the creation of anaerobic conditions is not studied in detail.

The following phase is, for this reason, Settle or Sedimentation phase, also called phase 2. To enter this phase, the OUR value calculated must be lower than 1/3 of the maximum value registered during phase 1 and should not differ from the previous OUR value for more than 20%. If these two conditions are true, the script launch the beginning of phase 2. More in detail, this means switch off aeration and mixing, while the discharge and charge pumps are still off. Sedimentation is supposed to last 45 minutes, a reasonable time for a tank of these dimensions.

Once the fixed time has passed, the code switches to phase 3, the discharge of the reactor (Draw phase). At the beginning of phase 3 the discharge pump is turned on, and the clarified water starts to exit from the tank through the floating system mentioned

above. This phase lasts until the lower-level sensor, situated at 9.5 m from the bottom of the tank, passes in “low” condition. In fact, it is a floating sensor that returns a 0 when it’s floating, and a 1 when it is not floating, so the water level is lower with respect to the sensor. The code switches to the following phase when the change from 0 to 1 is detected and turn off the discharge pump. The treated water is discharged directly into the sewerage system.
The last phase is the charging one, also called Fill phase, or phase 4. At this point the charging pump is turned on and the level of the water increases. In the upper part of the tank there is another level sensor; when water reaches it, the sensor starts floating and there is a change in the signal, from 1 to 0: that change is detected from the script that switches off the pump and enters again into biological phase, to start another cycle.
The script is also able to save the useful data into files that can be consulted and analysed when needed. In this way is possible to elaborate data and have a remote control of what happens in every moment.

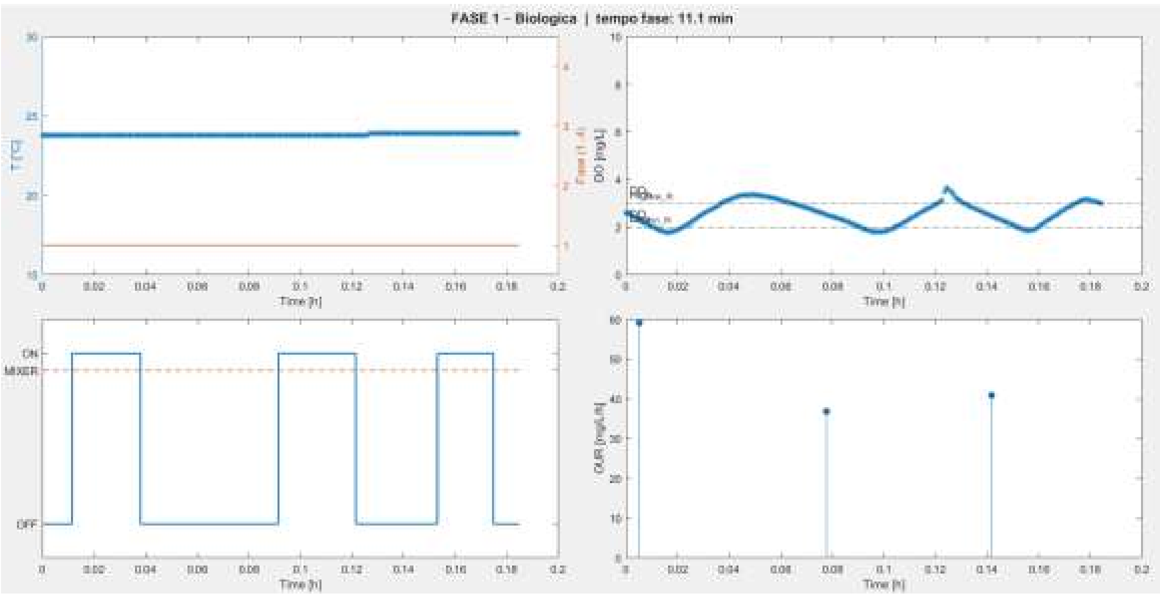

Figure 14: Matlab interface

4.3. Analysis and interpretation

4.3.1. COD removal efficiency

The first aim of the analyses was to evaluate the quality of degradation of COD during the biological phase. To do this, some samples were taken for the whole duration of phase 1, every 30 minutes, to easily see the trend of the microorganisms' activity.

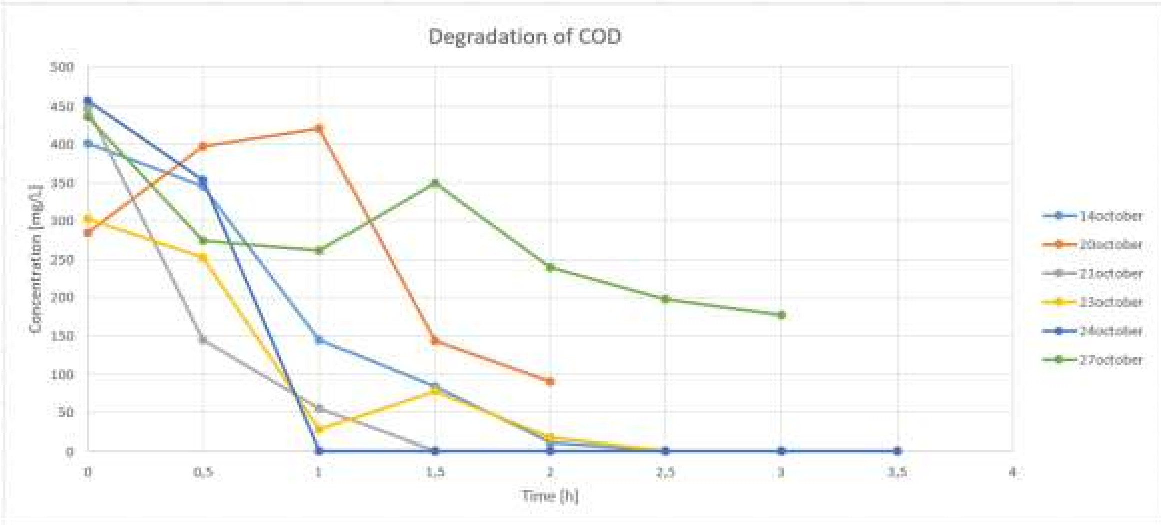

Figure 15: degradation of COD

As shown in the picture above, the biological phase doesn't have always the same duration, and the results are quite good. For the majority of the days in which samples were taken, the efficiency was maximum and the concentration of COD in the effluent reaches the zero. The important thing is that the concentration was under the BAT limit concentration for the discharge into surface water bodies, that corresponds to 65 mg/L on average. [5]

One noticeable result was the one of the 20th of October: COD concentration increases for the first hour. This result can be explained by the fact that a lot of sludge accumulates on the wall of the reactor, and sometimes it interferes with the sensors; on that day, in particular, the upper-level sensor was blocked from the sludge, so it was necessary to clean it. That sludge went inside the reactor as dead biomass, that represent a new source of COD, that explain the initial growth of the concentration; further checks should be carried out, but it is the most probable cause of that behaviour. Despite that, the final concentration of COD is supposed to be good, because the sampling didn't reach the end of phase 1. This event underlines one of the major criticalities of working with this type of technology: sometimes the reactor times does not fit with the hours of the day in which the laboratory is open, so it was not always possible to monitor it for the necessary interval of time.

Another interesting result is the one of the 27th of October: for the first time the efficiency decreased significantly, even if it was still around 60%. This may be due to the injection of a nitrogen source inside the synthetic wastewater, that slowed down the microbial activity because they need some time for the adaptation to the new conditions.
COD concentrations were evaluated trough a specific machine, TOC-TN [11], that measures the TOC inside each sample, previously filtered with filters with pores of 0.45 μm. After that, a conversion factor equal to 2.66 gCOD/gTOC was applied to the results.

4.3.2. Calibration test

The results on the degradation of COD, together with the OUR data directly evaluated by the script, were used to create a calibration model based on the Activated Sludge Model 3, to estimate some parameters that governs the activity inside the reactor. Activated Sludge Model 3 (ASM3) introduces interesting news to the classical equations used to estimate microbial activity, that are based on the ASM1: the new concept is based on the fact that microorganisms accumulate COD inside them before starting the degradation. If ASM1 assumes that heterotrophic bacteria grow directly on the readily biodegradable COD, ASM3 goes more in deep and affirm that the growth is not direct, but there is an accumulation first, as a polymer that grows inside the bacteria (X_{STO}), and then the colony of microorganism grow using that storage. Also the endogenous respiration assumes a different value: bacteria consume themselves to survive, but releasing new COD in an active way, that consumes oxygen, not in a passive way, as ASM1 declares. In this perspective, the source of oxygen is mostly used to increase the storage inside microorganisms. Due to this separation between the substrate consumption and the microbial growth, the equations are more complex with respect to the ones of ASM1. In fact, ASM1 describes the net microbial growth starting from Monod equation, considering the oxygen source as a limiting factor, as described in the formula below: [12]

$$\frac{dX}{dt} = \mu_m * \frac{S}{K_S + S} * \frac{S_O}{K_{O,H} + S_O} * X - k_d X$$

ASM3, instead, describe the net microbial growth with a system of two equations, as follows: [12]

$$\frac{dX_{STO}}{dt} = k_{STO} * \frac{S}{K_S + S} * \frac{S_O}{K_{O,H} + S_O} * X - \frac{1}{Y_{STO}} * \mu_H * \frac{X_{STO}/_X}{K_{STO} + X_{STO}/_X} * \frac{S_O}{K_{O,H} + S_O} * X$$

$$\frac{dX}{dt} = \mu_m * \frac{X_{STO}/_X}{K_{STO} + X_{STO}/_X} * \frac{S_O}{K_{O,H} + S_O} * X - b_h * \frac{S_O}{K_{O,H} + S_O} * X$$

Following the equations above, a Matlab script for the calibration of some parameters was created; the parameters chosen for the calibration were k_{STO}, the maximum specific storage rate, and K_s, the semi-saturation constant, because they were the most directly related to the data sets of the experiments explained above.

The calibration model is based on the concept of normalized least squares, so it starts with the literature values of these parameters (k_{STO} = 3.0 d^{-1} and K_S = 10.0 mg/L) [6] [12], and then tries to compare the equations with these values and the real data of the experiments; at this point, it evaluates the discrepancy between real data and the first fit. After that, it moves around the initial values of these parameters to compare the errors with the initial one and continues to the direction in which this error is lower. When the difference between the various errors is not relevant anymore, the script stops and choose the new values of the parameters. The final obtained values were: k_{STO} = 3.28 d^{-1} and K_S = 9.80 mg/L: these results seem to be quite good, since they're not so different from the literature. The script also plots the final version of the model next to the real data: the fits are really bad, so further studies are needed to create a calibration that fits well with the experimental data, as shown in the pictures below.

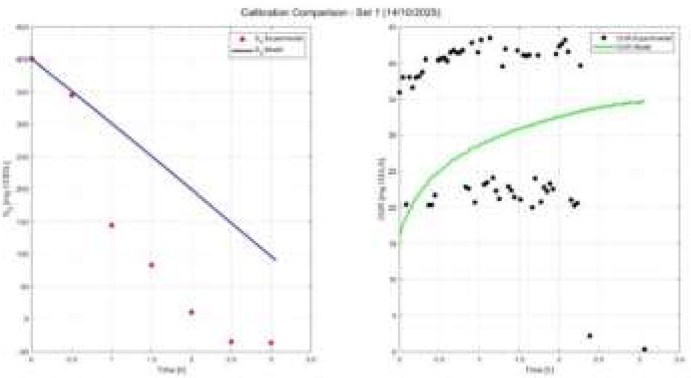

Figure 16: Set 1, 14th October

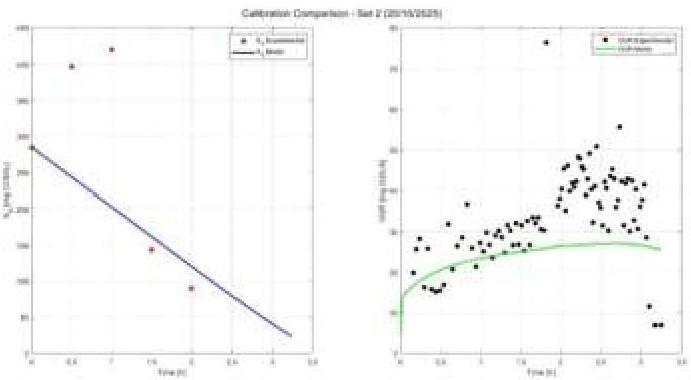

Figure 17: Set 2, 20th October

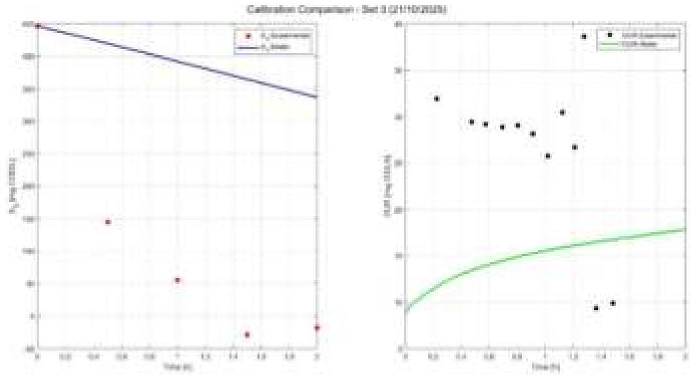

Figure 18: Set 3, 21st October

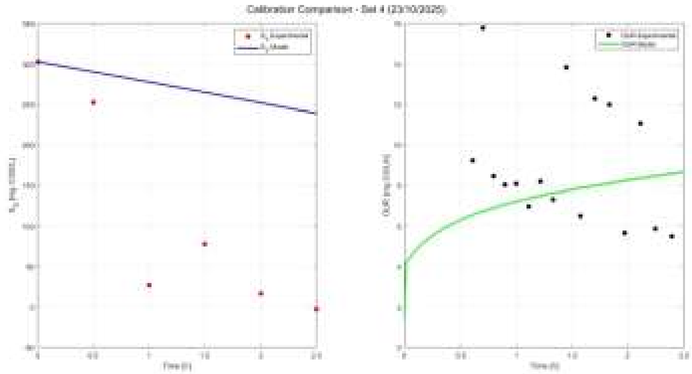

Figure 19: Set 4, 23rd October

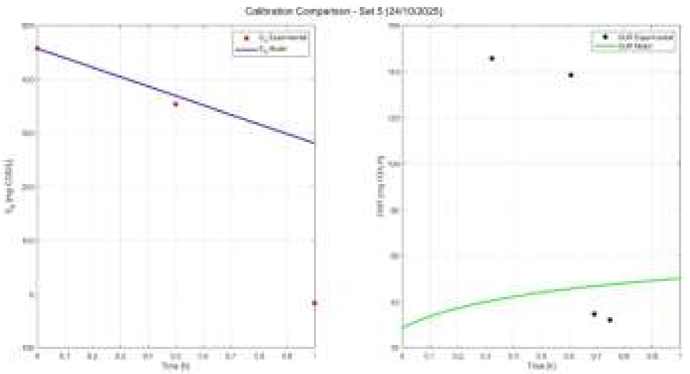

Figure 20: Set 5, 24th October

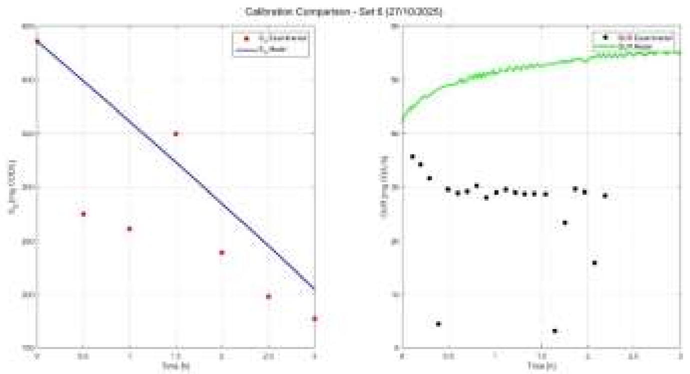

Figure 21: Set 6, 27th October

Going more in deep in the interpretation, it can be noticed that, in Set 1, the oscillation of OUR experimental data is really big, but the model is located in the middle, so it can be considered a quite good result, even if it doesn't catch the final decreasing of the OUR. The interpolation of the substrate concentration has a slope that is similar to the experimental data, but the line is not passing exactly over the last points, so it underestimates the biological activity. Regarding Set 2, the interpolation of the substrate degradation line is fitting quite well the points, even if it ignores the initial increasing: that's because the model can't predict the disturbance of real conditions that was present on that day, as already mentioned above. The OUR model is almost close to experimental data. Set 3 is probably the worst fit in both cases, while Set 4 has a bad fitting in the substrate concentration and OUR, too.
Set 5 model fits well only the first two points of the substrate concentration, resulting again in an underestimation of the biological activity; the OUR is bad. In the end, for Set 6, the line of the substrate concentration is almost good, but the same cannot be said for the OUR model.
These results are not good and cannot be taken into consideration. Probably the calibration found a local minimum with those values of k_{STO} and K_S, but the plots reveal a systematic error. The model is unable to replicate in the right way the biological activity observed in the experiments. This suggests that other fixed parameters may be incorrect and act as a bottleneck for the model. For this reason, even the fact that the parameters are close to the literature's ones, is not significant at all. Unfortunately, the time was not enough to catch more data samples and to deeply investigate all the other parameters, because of the difficulty of facing all the unexpected events that concern a laboratory experience. A suggestion could be a more thorough investigation of the temperature, that is constantly measured inside the reactor and influences a lot the microbial activity.

4.3.3. Microorganisms' growth and SRT

A further set of sampling was carried out to try to understand how many microorganisms are growing inside the reactor and what is the net growth rate; these data are important to estimate the sludge retention time (SRT) that is necessary to maintain inside the reactor, to guarantee a constant amount of active biomass and avoid an uncontrolled growth. In other words, it was necessary to fix how much sludge must be removed from the reactor every day; in fact, the SBR setup does not include the recirculation of the sludge as conventional systems, and this means that sludge must be manually removed from the tank with a certain regularity. The evaluation starts with a daily sampling of the sludge, to understand the trend with which microorganisms grow. The used parameter was Volatile Suspended Solids (VSS) since bacteria are suspended solids made of organic matter that can be oxidized at 600 °C.

Solids evaluation comprehends Total Solids (TS), Volatile Solids (VS), Total Suspended Solids (TSS) and Volatile Suspended Solids (VSS). TS are formed by TSS and Total Dissolved Solids (TDS), while VS comprehend VSS and Volatile Dissolved Solids (VDS). To evaluate TS and VS, samples of sludge as the ones showed in Figure 22 were taken, while, to evaluate TSS and VSS, samples need to be previously filtered with nitrocellulose filters with pores of 0.45 μm. In fact, by definition, suspended solids are the ones that remains trapped by a filter with pores of these dimensions. Filtered samples are shown in Figure 23.

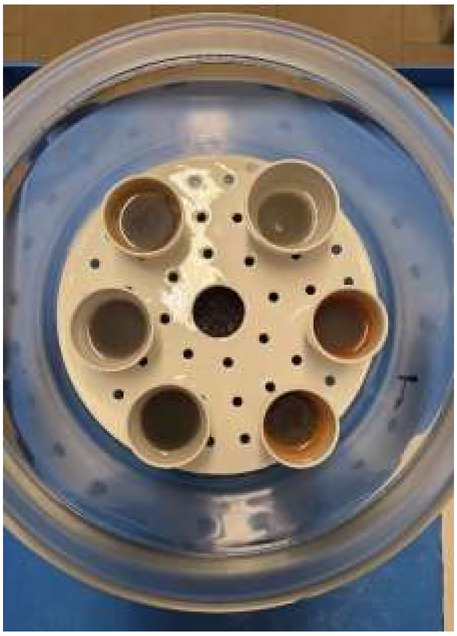

Figure 22: samples of sludge for TS and VS evaluation

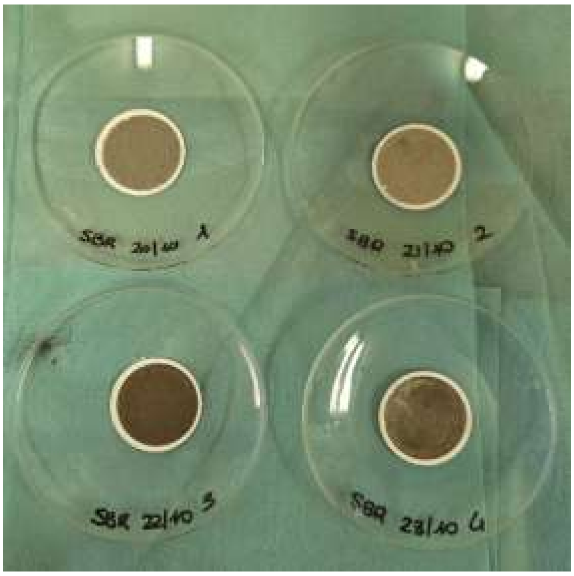

Figure 23: filters with samples for TSS and VSS evaluation

At this point, all the samples were put in an oven at 105 °C, to let water evaporate. After one day, dried samples were cooled inside a desiccator and then weighted. The difference between this weight and the tare previously evaluated indicates the amount of TS and TSS, respectively, inside the samples. The corresponding concentration was found dividing the weight by the amount of sludge used at the beginning. At this point, the dried samples were put for 2 hours in an oven at 600 °C; at that temperature, all the organic matter oxidize, and only ashes remain inside the cups. After cooling, samples were weighted again and VS and VSS correspond to the difference between TS and TSS, respectively, and the corresponding ashes left. Results are shown in the graphs below.

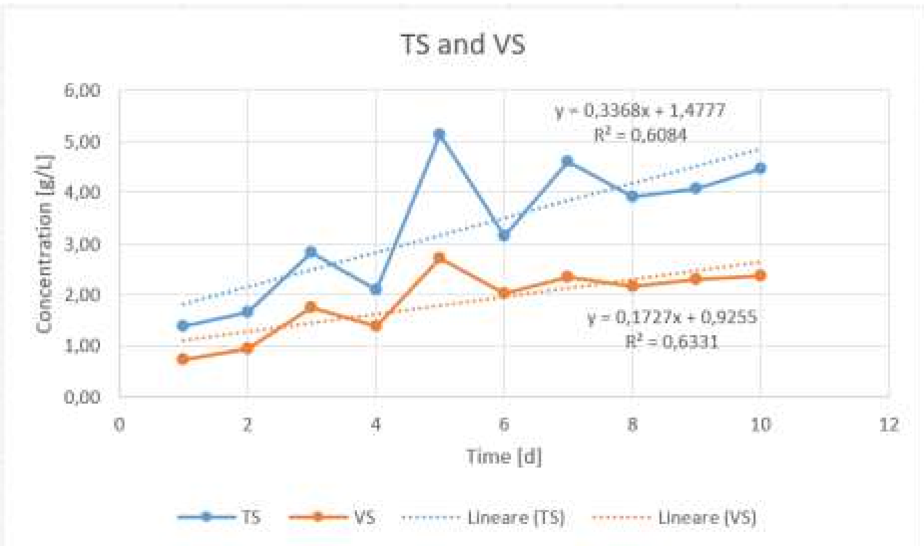

Figure 24: TS and VS trends

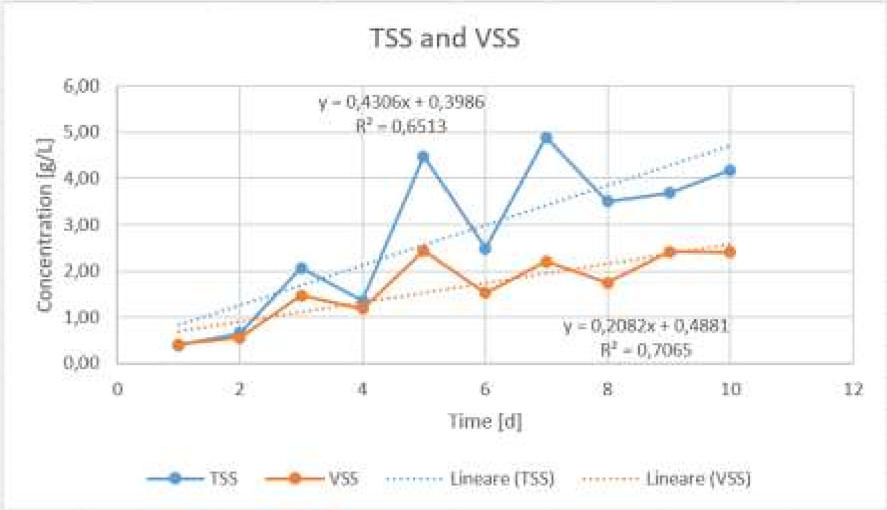

Figure 25: TSS and VSS trends

Going on with the estimation of the reactor conditions, it is necessary to linearize the trend of VSS: the slope of the trend corresponds to the net growth rate of the microorganisms, $\mu_{net} = 0.21\ d^{-1}$. The linearization is represented in Figure 25. This approach has been chosen because the microbial activity is limited by the amount of substrate treated, since the net growth rate found is far from the maximum one that is known from literature ($\mu_{max} = 2\ d^{-1}$); in these conditions the Monod equation can be approximated to a first order growth. [12]

In the perspective of maintaining a constant concentration of microorganisms inside the reactor, so stationary conditions, SRT must be equal to the opposite of μ_{net}, so SRT is fixed equal to 4.80 d. The target concentration wanted is fixed at 4 gVSS/L, that corresponds to 80 g of VSS, since the reactor volume is around 20 L, as already said. At this point, the mass flow rate that must be removed every day is calculated as:

$$M_{OUT} = \frac{M_{FIX}}{SRT} = \frac{80\ gVSS}{4.8\ d} = 16.7\ \frac{g\ VSS}{d}$$

To evaluate the volumetric flow rate that must be removed, it's important to underline that in laboratory conditions, it's impossible to remove solids from the bottom of the tank, after sedimentation, as should be done at industrial scale. To face this problem, solids have to be removed during the biological phase, in which everything is mixed; this kind of removal allows to catch a volume of sludge that is homogeneous and representative of the whole tank. In this way, to pass from the mass flow rate to the volumetric flow rate, the target concentration of solids, fixed at 4 gVSS/L, should be used, and the volumetric flow rate to be removed is equal to 4.16 L/d. Assuming an average of 4 cycles per day, a reasonable choice should be the one to remove around 1 L of sludge at every cycle.

4.3.4. Nitrogen addition

In the end, some considerations about the addition of nitrogen source: the TOC-TN machine used for TOC measurements [11], can also calculate the total nitrogen (TN) present inside the samples, so the trend of nitrogen concentration during the whole biological phase of the 27th of October, is known. The result is something strange, because TN decreases a lot during time, and the outlet concentration reaches 5 mg/L, that is perfectly aligned with the limit imposed by BAT documents. [5]

This seems good, but nitrogen removal follows three specific steps, mentioned in the previous chapters, that are ammonification, nitrification and denitrification: the first two steps can occur during the biological phase, since there are aerobic conditions, but denitrification should occur in anoxic conditions, so nitrites and nitrates can be reduced into molecular nitrogen by bacteria, which use the oxygen linked to nitrogen for their activity. Anoxic conditions, at this level of the experimentation, are not created, so nitrogen was not expected to disappear. During sedimentation phase, the air pump is switched off, so the concentration of DO is really low, but there's nothing that control the conditions and, first of all, nitrogen is degraded a lot even before sedimentation, where DO concentration is relevant.

To deeply investigate this behaviour, more analyses were conducted to determine how much nitrogen is ammonia nitrogen, nitrite nitrogen and nitrate nitrogen. Ammonium is measured through a colorimetric kit with some reagents that reacts with the sample and colour it with different intensities of green related to the ammonium concentration; after that, the treated sample is insert in a spectrophotometer UV-VIS [11] that return the corresponding concentration of NH_4^+ wanted. Nitrites and nitrates, instead, were measured through an ionic chromatographer [11] that returns the concentrations of the ions.

After that, the concentrations found must be converted into $N\text{-}NH_4$, $N\text{-}NO_2$ and $N\text{-}NO_3$ concentrations, through a proportion that exploits molecular weights. The sum of these three concentrations should be equal to the TN measured with the TOC-TN; these values are not perfectly identical due to the uncertainties and errors that are intrinsic of every machine. The final result is shown in Figure 26.

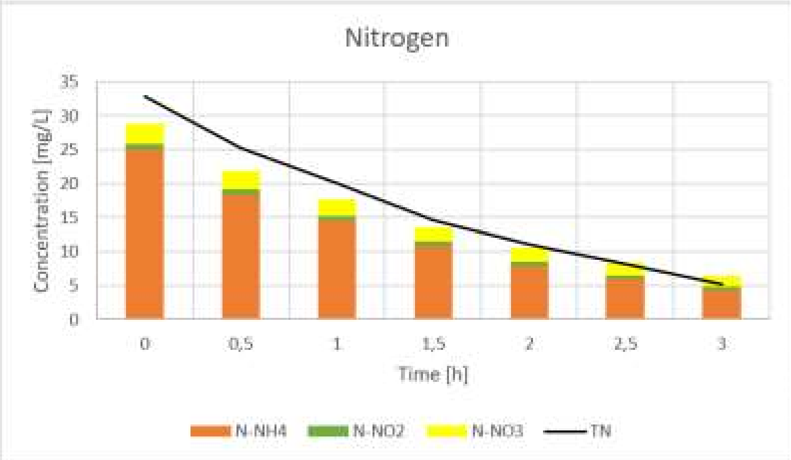

Figure 26: nitrogen trend

During time, ammonia nitrogen percentage decreases, while the percentages of nitrites and nitrates increase, inside a single sample, as shown in Figure 27. That should be due to the correct nitrification that occurs, while denitrification seems not present, as it should be. Nitrification occurs slowly, since not all the nitrogen is converted into nitrite and nitrate form; the reason should be linked to the fact that autotrophic bacteria, responsible for nitrification, were not developed yet, since the reactor has been maintained in nitrogen deficit for a while.

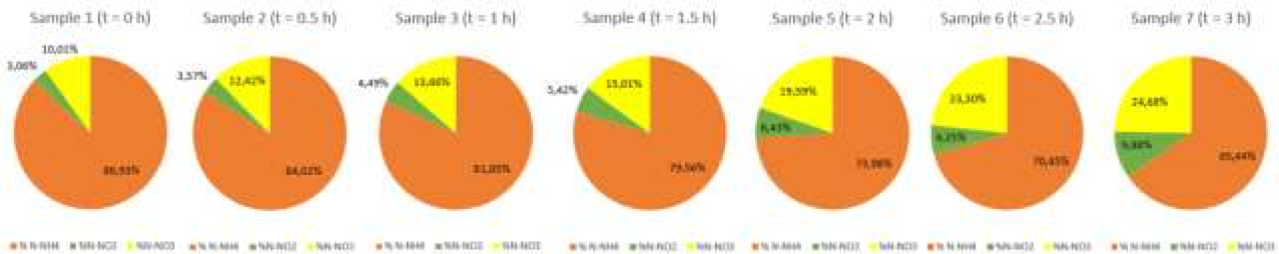

Figure 27: different composition of the samples

The problem that remains unsolved is the one of the overall degradation of nitrogen: the only probable answer is that microorganisms were in deficit of nitrogen, since they have been working for one month without it. In other words, all the "missing" nitrogen was assimilated by bacteria for their own growth; this result is aligned with the worse COD degradation, in fact the consumption of nitrogen slowed down COD degradation.
For these reasons, further analyses are needed to draw conclusions about the efficiency of nitrogen removal of the SBR.

5. Conclusions

To conclude the study, it can be said that further insights are needed to deeply understand the behaviour of microorganisms and to be able to control the fluctuations of the system due to real conditions. Despite that, the results of the analysis on the efficiency of the reactor are really good and they give hope to the implementation of SBR at industrial scale in the next future.

This is an encouraging result, because with the updates on the environmental laws described in the previous chapters, that are written in the EU directive 2024/3019 of the European Parliament and of the Council of the 27 of November 2024, a lot of small urban sites will need to construct a wastewater treatment plant by the end of 2035. The reason is that the directive reduces the threshold of agglomerations that must have a collecting system for urban wastewater from 2000 to 1000 p.e.. [2]

In this perspective, the construction of wastewater treatment plants that uses an SBR for the biological treatments instead of a conventional activated sludge (CAS) technology, would be a great advantage for the urban sites, because SBR need a small land occupation and are more energy efficient.

To demonstrate that, a simplified comparison has been taken out between a CAS and a SBR plant for an urban site of 1000 p.e.. For simplicity, the comparison is made only for the core part of the difference, not on the whole machineries and technologies that the construction of a new plant should need. In other words, for the CAS are considered the biological tank and the secondary sedimentation, while for SBR is considered just the reactor. The table below shows the initial conditions needed for the design that correspond to the dimensions of the urban site chosen.

Table 1: initial conditions

Initial conditions				
	1 p.e.		**1000 p.e.**	
Daily discharge	200	L/d	200 000	L/d
Q_in	0,2	m3/d	200	m3/d
Organic load (OL)	0,06	kg BOD5/d	60	kg BOD5/d

At this point it's necessary to design the volume of the tanks; to do this, the formula below is used for the CAS tank [3]:

$$\frac{F}{M} = \frac{Q * S_0}{V * X}$$

Knowing that $Q*S_0$ corresponds to the organic load and fixing all the other parameters as shown in Table 2, the volume needed for the CAS is found out to be 50 m^3.

Table 2: fixed parameters for CAS design [3]

Fixed parameters for CAS design		
Parameter	**Value**	**U.M.**
F/M	0,3	kg BOD5/(kgVSS*d)
X	4	kg VSS/m3

For what concerns SBR, the same value of F/M used for the CAS is not valid, since the batch technology allows to work at higher rates of degradation, because there is no need to guarantee for all the reaction time a low concentration of COD in the effluent. For this reason, the formula used to evaluate SBR volume is the following [3]:

$$V = \frac{P_x * SRT}{X} = 24.61\ m^3$$

Where SRT is considered equal to 4.8 d, as found out through the laboratory analysis explained in the previous chapter. P_x, instead, is the daily production of microorganisms, and it has been evaluated with the following formula [13]:

$$P_X = \frac{Y * OL}{1 + b * SRT}$$

Where OL is the organic load, Y is the biomass growth coefficient and b is the biomass decay rate. The used values are shown in Table 3 [12].

Table 3: fixed parameters for SBR design

Fixed parameters for SBR design		
Parameter	**Value**	**U.M.**
X	4	kg VSS/m3
Y	0,67	-
b	0,2	1/d
SRT	4,8	d

For the SBR, the tank volume should consider also the space for the sedimentation, so the active volume of 24.61 m^3 corresponds to the 65% of the total volume. For this reason, the SBR tank should be 37.86 m^3.

For plant safety and to better manage ordinary maintenance, it’s a good practice to divide the design volume by two and consider two lines working in parallel with a flow rate of 100 m^3/d each. This is particularly useful for the SBR since it doesn’t work in steady state; with two tanks the management of the wastewater flux will be easier. Table 4 reports the dimensions of each tank.

Table 4: dimensions of the tanks

Tanks dimensions				
	CAS		SBR	
Volume	25	m3	18,93	m3
HRT	6	h	4,54	h
Surface	7,14	m2	5,41	m2

Together with the CAS, it is necessary to design also the secondary sedimentation, while SBR can manage also the sedimentation phase. Two settlers are needed, one for each CAS tank. The parameter used to design each tank is the Surface Overflow Rate (SOR), that represents the velocity with which water is able to rise up while solids sedimentate. Of course, SOR must be lower than the sedimentation velocity, to obtain a good quality clarified water. A reasonable value for SOR is 1 $m^3/(m^2*h)$, from which it can be obtained the surface of the basin, dividing by the flow rate, fixed at 160 m^3/d, because it takes into account also the recirculation of the sludge. The theoretical diameter that is obtained from the surface was 2.9 m, value that is rounded up to 3 m, for safety reasons and to match catalogues parameters. At this point, the final characteristics of each sedimentation basin are found and shown in Table 5.

Table 5: sedimentation basin design

Sedimentation basin		
Diameter	3	m
Surface	7,07	m2
Volume	24,74	m3

The total area occupied by CAS and sedimentation basins is around 28.42 m^2, while the surface occupied by the two SBR is around 10.82 m^2. The SBR solution guarantee almost a 62% saving of land use. This can be a significant reduction in the costs of the land that is necessary to purchase, but also during the construction phase.

Another relevant expense that needs to be faced in the CAS solution is connected to the recirculation of the sludge, from the sedimentation unit to the biological tank, needed to guarantee stationary conditions inside the biological tank. A dedicated pipeline with pumps should be installed to face recirculation, that consist in a flow rate that is almost 60% of the entire flow rate; so, it's around 60 m^3/d. It must be considered also the pipes to bring water from the biological tank to the sedimentation one, in which water moves thanks to gravity force.

For what concerns pipes, it was estimated that around 10 m are needed for each line, so 20 m extra with respect to the SBR configuration. Hypothesizing a cost of 150 €/m for PVC pipes, the price will be around 3000 €. Regarding recirculation pumps, instead, one for each line is needed, and they must manage the flow rate mentioned above, 60 m^3/d; the cost of both of them is around 6000 €. Hypothesizing a standard pressure of 0.5 bar, that corresponds to a head loss of 5 m, the theoretical power is evaluated and it's equal to 0.03 kW; considering an efficiency of 85%, the effective electrical power corresponds to 0.04 kW. From this data is possible to evaluate the annual amount of extra energy due to these pumps, in the CAS configuration. In the energy evaluation it is considered also the nominal power of 0.18 kW for each sedimentation unit, that is used for the bottom scraping. Table 6 summarizes the extra cost of electricity.

Table 6: annual electricity cost

Electricity		
Total power	0,44	kW
Hours of operation	8760	h/y
Total energy	3,87	MWh/y
Electricity prize	138	€/MWh
Total annual cost	533,96	€/y

In this case, since the hypothetical plant is a small one, the annual saving is not so relevant, but for big plants, such as the one of Castiglione Torinese, for example, the saving would increase even of three orders of magnitude. On the other hand, being a small plant, maybe this value corresponds to a considerable percentage inside the whole economical balance.

Of course, it is necessary to mention that the major electricity costs are due to the air injection, for both configurations. In the CAS one, aeration is always active, but at a low regime, because the velocity of the reaction must be low to guarantee the effluent quality, as already mentioned; in the SBR, instead, the air injection is intermittent, but at a higher flow rate, because the reaction can occur faster. Due to these differences, it is supposed that the annual energy consumption for the air pumps is comparable between the two configurations, that's the reason why it was not analysed in detail. [13]

As a conclusion, the results about the savings of SBR technology are really encouraging, making it a valid alternative for wastewater treatment plants, that guarantee a good performance while helping to save land, money and energy.

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