Review

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*Review*

# Nutrient Removal and Recovery from Municipal Wastewater

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**Abstract:** With the ongoing amendment of the EU legislation on the treatment of urban wastewater, stricter requirements for the removal of pollutants are expected, which calls for the need for innovative wastewater treatment technologies. Biological systems are still the first choice. The survey of typical bioreactors applied in wastewater treatment is presented. Wastewater treatment objective, biochemical environment and microbial growth are selected as a main criterion for classification of these bioreactors. Hydraulic and kinetic aspects are considered, advantages and drawbacks of these bioreactors regarding the selection of appropriate type of the reactor as well as regarding operation of reactors are mentioned. The aim of this paper is to provide operators and designers with a brief overview of selected traditional and advanced processes, reactors, and technologies for nutrient removal from municipal wastewater. Another goal is to discuss the possibilities and limitations to comply with more strict effluent standards. From the evaluation of the published papers, it follows that the currently applied traditional methods of nutrients removal have the potential to convey also with the expected stricter limits.

**Keywords:** municipal wastewater; nutrients; removal; secondary; tertiary; treatment

## 1. Introduction

Wastewater is essentially water supply of community after a variety of uses. From the point of origin, wastewater may be characterized as a combination of the liquid and solid waste removed mainly from residences, social and industrial institutions, and similar facilities together with groundwater and storm water depending on the type of sewer system.

Domestic wastewater means wastewater from residential settlements and services which originates from the human metabolism and from household activities [1]. Municipal wastewater is water delivered to communities after various uses originating from domestic, industrial, commercial, and institutional sources within a given human settlement or community [2]. Urban wastewater includes both municipal wastewater and urban runoff.

The accumulation of untreated wastewater can lead to the production of malodorous gases due to decomposition of organic impurities. In addition, untreated wastewater usually contains numerous pathogenic or disease-causing microorganisms. It usually contains nutrients, which stimulate the growth of algae and other aquatic plants and consequently accelerate eutrophication.

A high content of organic and ammonium impurities can lead to excessive consumption of dissolved oxygen and undesirable changes in aquatic life in the receiving water. Wastewater can also contain toxic compounds. It is therefore a necessity to treat wastewater before discharging it into the receiving water.

Nitrogen and phosphorous are inevitable for the growth and vitality of living beings. However, the discharge of nutrient-rich wastewater into aquatic ecosystems leads to eutrophication. Thus, nutrient N and P effective removal from WW is way to meeting strict nutrient discharge standards [2]. In addition, nutrient recovery from WW is equally important for realizing a circular economy by preventing the depletion of limited resources.

On October 26, 2022, the European Commission published a proposal [3] for the revision of the Council Directive of May 21, 1991, on the treatment of municipal wastewater "Urban Wastewater Treatment Directive - UWWTD".

Although the presented proposal represents material that is still subject to debate and changes, it can be considered as the framework and basic theses that this proposal contains.

The proposed directive [3] emphasizes not only the protection of the environment but also the protection of people's health, the reduction of greenhouse gas emissions, better access to sanitation and wastewater treatment, the improvement of management and transparency, and the improvement of information sources on public health.

The original directive [2] went through several evaluations during its validity. The European Commission published its latest assessment (2019 REFIT EVALUATION) [3] in December 2019. This assessment shows significant reductions in pollution in discharged waste in terms of $BOD_5$, total nitrogen and total phosphorus, and thus a significant impact on the quality of lakes, rivers and seas in the EU are visible and tangible. However, the evaluation also pointed out three complex and ambitious challenges:

- Remaining pollution.

Decrease of pollution load from individual sources in the EU by reducing pollution discharged from overflows of relief chambers (StormWater Overflows - SWO), from agglomerations below 2,000 EO or inadequate individual treatment systems.

- Insufficient alignment of the directive with the objectives of the EU policy around the EU Green Deal.

Decreasing of total energy consumption and all greenhouse gas emissions in the whole EU is possible by improving the treatment process, better use of sewage sludge and increasing energy efficiency and a higher rate of use of renewable resource technologies. It is necessary to better integrate this area into the circular economy, improve sludge management and reuse of water and valuable resources.

- Insufficient and uneven level of management

The evaluation report also pointed out the different level of operation of sewage networks and wastewater treatment plants between individual EU member countries. The existing monitoring and reporting do not use all the possibilities of digitization. The results of the public consultation resulted in a request to address the area of micropollutants in wastewater and to better apply the "polluter pays" principle by introducing and expanding producer responsibility.

With the ongoing amendment of EU legislation on urban wastewater treatment [3], stricter requirements for pollutant removal are expected driving the need for innovative environmental technologies. Diverse pollutants in urban wastewater, including macronutrients and micropollutants, require advanced treatment technologies that integrate biological, physical, and chemical processes [4].

This review article is focused on the removal and recovery nutrient N and P from municipal wastewater. Provides the description of processes, reactors and technologies and new challenges in the context of the proposed directive [4]. The aim of this paper is to provide operators and designers with a brief overview of selected traditional and advanced processes, reactors and technologies for nutrient removal from municipal wastewater. Another goal is to discuss the possibilities and limitations to comply with more strict effluent standards.

Most wastewater treatment plants (WWTP) are equipped with mechanical pre-treatment, where floating substances (rakes) and sedimenting inorganic solids are physically removed by sedimentation (sand traps). In primary treatment, undissolved organic substances are separated in sedimentation tanks [5].

In secondary treatment, the remains of organic substances are removed by biological processes, while macronutrient (N and P) are also partially removed by their assimilation into the new cell mass. Thus, the main function of convention WWTPs is to remove mainly organic pollution. However, even after typical secondary treatment, some dissolved and suspended substances remain in WW along with the main part of N and P [6–8].

From the point of view of macronutrient, they therefore represent a dominant point source of nutrient pollution of the aquatic ecosystem with N and P [8].

A tertiary or advanced treatment system is required for their desired removal. Due to the dominant applications of biological processes using activated sludge in sewage treatment practice [9,10], these processes also began to be used in tertiary technologies for the removal of macronutrients.

In this context, however, it is necessary to state that the high operating and investment costs associated with the implementation of these processes and technologies require significant financial requirements and thus also limitations in their wide use in technological practice [5].

It turns out that similar limitations also apply to physical methods. Membrane applications are also costly. They are not very effective as they only remove approximately 10% of the total macronutrients. Other problems include membrane fouling [11,12]. Chemicals are also expensive. Handling and storing them is hazardous. Their applications lead to higher sludge formation, which increases maintenance and operation costs [13].

## 2. Nutrient Removal

It is well known that biological processes are widely used for the removal of organic, nitrogen and phosphorus from wastewater. One reason is the possibility to achieve high elimination efficiency of these pollutants from wastewater. In addition to this there are relatively low capital and operational costs in comparison to physicochemical or chemical processes [14].

Biological processes of wastewater treatment are environmentally friendly. Represent the intensification of the processes, which occur in nature, for example well known process of self-purification in rivers, or rotting of organics (break down, decay) in the absence of air. The intensification of natural biological processes is carried out in bioreactors and basically is related to objectives of treatment process (removal of organics, nitrogen, and phosphorus substances), to rate of treatment process and consequently to required reactor volumes, capital, and operational costs [15,16].

Practical applications show that the conventional activated sludge process with a gradual flow can be easily intensified to a system with biological phosphorus removal by creating an anaerobic zone at the beginning of the activation tank. This reconstruction requires installation of a baffle to prevent back-mixing, shutdown of aeration and installation of anaerobic mixing zone [17]. This system is highly efficient in terms of phosphorus removal if nitrification does not occur under oxic conditions. On the other hand, this system can also be modified for the purpose of nitrogen removal, namely by installing additional baffles, mixers and including appropriate recirculation of nitrates.

The draft of the new directive [3] establishes the conditions for ensuring the tertiary treatment of municipal wastewater (nitrogen, phosphorus).

The concept of sensitive areas is abolished, and the new Member States will have the obligation to define areas sensitive to eutrophication.

The main change is the fact that the requirements for nitrogen and phosphorus removal are not determined by the size of the agglomeration, but directly by the size of the load on individual WWTPs. In the case of WWTPs with a size of over 100,000 PE, tertiary treatment will be mandatory. For WWTPs with a size of over 10,000 PE, they will be mandatory if their treated wastewater is discharged into an area defined as sensitive to eutrophication.

Original concentrations for total nitrogen 15 mg/l (for 10,000-100,000 PE) or 10 mg/l (at more than 100,000 PE) is reduced to 6 mg/l. The corresponding original minimum reduction values for total nitrogen are 80%, or 70-80% will increase to 90, respectively. 85%.

Original values for total phosphorus 2 mg/l (for 10,000-100,000 PE), or 1 mg/l (at more than 100,000 PE) are reduced to 0.5 mg/l. The corresponding original minimum reduction value for total phosphorus of 80% is increased to 90%.

Depending on the local situation, either or both parameters may be used. One of the main topics of the transposition of the future directive [3] will therefore be the definition of areas sensitive to eutrophication.

Biofilm-based and granule/activated sludge-based microbial growth processes are most used in WW biological treatment. The development of requirements for the quality of treated water, requirements for cost reduction and the development of processes and technologies for treating WW are shifted significantly also towards hybrid processes that use combined forms of biomass [4].

Biological nutrient removal (BNR) is a term used to describe the application of microbes to take up nitrogen and phosphorus during WW treatment [18].

As the requirements for the quality of treated WW increase, so does the complexity of their treatment technologies. Individual biological processes for the removal of organic pollution, N and P, require the provision of different conditions in terms of the presence of oxygen (oxic, anoxic, anaerobic conditions) as well as the required form of carbon (inorganic or organic) [15,16].

No less important are the differences in the values of specific growth rates of dominant microorganisms for individual processes and their sensitivity to the presence of various pollution components, temperature changes, technological parameters, etc. These factors must be considered when designing processes and technologies for treatment WW as well as during their actual operation. On the other hand, the removal of nutrients from WW is also accompanied by higher costs [21].

### *2.1. Nitrogen Removal Processes*

Nitrogen is one of the basic elements of cell mass. However, its excessive release into water bodies can lead to the acceleration of eutrophication in receiving water. Nitrogen removal from wastewater can be carried out using physico-chemical, chemical, bioelectrochemical and biological methods. The authors [20] dealt with operating conditions, advantages, and disadvantages of used processes and influencing factors.

Ammonia stripping with air is the most common pretreatment process. However, it requires large spaces and is time-consuming. Its disadvantage is also the formation of calcium carbonate deposits.

Chemical processes showed the least efficiency around 20-30%, which is dependent on the pH of the process.

Biological treatment processes (Nitrification and Denitification, Anammox,) are among the most effective (in terms of achieved efficiency (> 95%), costs and an environmental origin) for the removal of nitrogenous pollutants from municipal WW.

Due to the complexity of wastewater, a combination of different methods using the advantages of individual systems is recommended for efficient and economical treatment of WW in accordance with the characteristics of WW and the quality requirements.

Ammonium nitrogen can be transferred to nitrite or nitrate in aerobic environment. Inorganic carbon, i.e., carbon dioxide is utilized by chemoautotrophic nitrification bacteria (*Nitrosomonas, Nitrobacter*) in synthesis of new microorganisms. The energy-yielding two step oxidation of ammonia to nitrate is generally represented as follows [17]:

*Nitrosomonas*

$$2NH_4^+ + 3O_2 \rightarrow 2NO_2^- + 4H^+ + 2H_2O \quad (1)$$

*Nitrobacter*

$$2NO_2^- + O_2 \rightarrow 2NO_3^- \tag{2}$$

Total reaction

$$NH_4^+ + 2O_2 \rightarrow NO_3^- + 2H^+ + H_2O \tag{3}$$

The removal of the products of nitrification can be accomplished in *anoxic conditions*, i.e., the level of dissolved oxygen is maintained near zero (less than 0,5 mg/l). The reduction of nitrite and nitrate to nitrite to nitric oxide to nitrous oxide to nitrogen occurs. In anoxic conditions, i.e., the level of dissolved oxygen is maintained near zero (less than 0,5 mg/l), the reduction of nitrite and nitrate to nitric oxide to nitrous oxide / nitrogen occurs according to Equation (4):

$$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2 \tag{4}$$

*Nitrate nitrogen* is utilized as the terminal electron acceptor, releasing nitrogen gas as the product in the process of denitrification. Organic carbon source is necessary in the process of denitrification.

By combining shortened nitrification, the so-called nitritation, significant savings in both investment and operating costs can be achieved with the anammox process. Nitrifying bacteria (AOB - ammonium oxidizing bacteria) oxidize 57% of the input $NH_4^+$ to $NO_2^-$ according to Equation (5), which saves approx. 50-60% of energy for aeration compared to nitrification. The reason is that only part of the ammoniacal nitrogen is oxidized biochemically, while the oxidation is already completed in the nitrite stage. Anammox microorganisms transform the remaining ammoniacal nitrogen and gaseous $N_2$ according to Equation (6) [21]. As the annammox microorganisms of the process belong, unlike denitrifying bacteria, to chemolithotrophic microorganisms, nitrogen is essentially removed without consumption of organic substrate. At the same time, up to 80% less excess sludge is produced. The anammox process transforms up to 13% of input nitrogen into nitrates. In practice, these nitrates are at least partially denitrified by heterotrophic microorganisms. The advantages of technologies based on the anammox process include high performance (the proposed load can be 0.35–2.3 kg$_N$/(m$^3$/d) [22], in some cases even 10 kg$_N$/(m$^3$/d) [23].

$$NH_4^+ + 1.5\, O_2 \rightarrow NO_2^- + 2H^+ + H_2O \tag{5}$$

$$\begin{aligned} NH_4^+ + 1.32\, NO_2^- + 0.066\, HCO_3^- + 0.13\, H^+ \\ \rightarrow 1.02\, N_2 + 0.26\, NO_3^- + + 0.066\, CH_2O_{0.5}N_{0.15} + 2.03\, H_2O \end{aligned} \tag{6}$$

Biological nitrogen removal (BNR) is a critical process in WW treatment. The authors of [24] present new microorganisms that can remove nitrogen through new metabolic pathways. These include ammonia oxidizing archaea (AOA), complete ammonia oxidation bacteria (COMAMMOX), anaerobic ammonium oxidation associated with iron reduction bacteria (FEAMMOX), anaerobic ammonium oxidation bacteria (ANAMMOX) and anaerobic methane oxidation denitrifying microorganisms (DAMO). Compared to current nitrifying or denitrifying bacteria, these new microbial groups have better physico-chemical tolerance and/or lower release of greenhouse gases. They represent promising microbial communities that could provide new technologies for high-performance and energy-saving nitrogen removal from WW.

### 2.1.1. Single Anoxic System

The development of nitrogen removal technologies gradually progressed from three-sludge to single-sludge systems. In the case of two-sludge systems, it is advantageous to operate the denitrification process in reactors with increased biomass (packed columns, rotating disk reactors, fluidized bed bioreactors). Systems with separated biocenosis are relatively stable and achieve high nitrogen removal efficiencies. However, their disadvantage is relatively high investment and operating costs.

The choice of nitrogen removal technology is marked not only by the characteristics of wastewater, the required quality of treated water and investment options, but also by the stage of decision-making and selection. In the case of the construction of a new sewage treatment plant

(WWTP), it is possible to consider more comprehensively all aspects, starting with requirements for the built-up area, ending with guarantees of the quality of the outflow and flexibility in terms of managing individual processes in the event of changes in the amount and composition of wastewater.

Currently, single-sludge systems are most often operated. Under single-sludge systems, the systems in which activated sludge is separated from treated wastewater in only one settling tank is meant. These systems make it possible to use organic wastewater pollution as a source of organic carbon for denitrification, while relatively high nitrogen pollution removal efficiencies are also achieved.

The most widespread arrangements of a single-sludge system regarding the removal of nitrogen compounds from wastewater include upstream denitrification and simultaneous denitrification. In the case of some industrial wastewaters, included denitrification appears to be advantageous.

These systems make it possible to achieve relatively high treatment efficiency also when using municipal WW as a source of organic carbon for denitrification. They are considerably flexible in terms of the possibility of controlling nitrification and denitrification processes. The most used single-sludge system technologies for removing nitrogen compounds from WWV are upstream denitrification and simultaneous denitrification.

Each system with a single anoxic zone (single anoxic system) should be able to achieve for sewage WW average annual output values of total nitrogen (TN) of 8-12 mg/l. [25].

Most of the biological stages of urban WWTPs can be characterized from the point of view of the hydrodynamic regime as systems with a gradual flow. The investigation of the influence of hydrodynamic conditions on the kinetics of the nitrification process Chudoba et al. [26] showed that the gradual flow system is more advantageous for the nitrification process compared to the mixing system [26]. Azimi and Horan [27] also presented the advantages of a gradual flow nitrification system under steady state conditions [27]. Similar results were also published by Horan and Azimi [28] based on the investigation of the responses of nitrification systems with different hydrodynamic conditions to a sudden increase in nitrogen pollution [28]. Both systems showed very similar responses for the monitored forms of nitrogen pollution. Thus, in the mixing system, the effect of dilution in favor of the nitrification process was not observed.

*Wuhrmann system*

Biological nitrogen removal (BNR) can occur in a two-step process that begins with the use of autotrophs to oxidize $NH_4^+$ to $NO_3^-$ (nitrification), followed by the reduction of $NO_3^-$ to $N_2$ by heterotrophic microorganisms (denitrification). The system was developed by Wuhrmann [29] and in literature is referred according to its developer or post-denitrification system.

In the case of domestic wastewater, the basic disadvantages of this system include the need for an external source of organic carbon as well as the increased cost of oxygen supply in connection with the oxidation of organic wastewater pollution in the aerobic stage. In this system, it is not possible to eliminate the decrease in neutralization capacity due to the nitrification process by hydroxyl ions produced in the denitrification stage. For this system, it is also necessary to include post-aeration before the settling tank.

Post-denitrification is advantageous mainly from the point of view of energy (smaller requirements for recirculation) but also from the point of view of the possibility of achieving higher efficiency of the process compared to pre-denitrification. However, to achieve required nitrogen removal efficiency, the supply of an external carbon source that does not contain ammonia or organic nitrogen, or with their small content, is inevitable. Monteith et al. [30], used as an organic carbon source for denitrification 30 different types of industrial wastes and compared the results with denitrification using methanol as an organic carbon source. Among the waste used were various food (brewery, distillery, wine, starch), chemical (organic acids, distillation residues, higher alcohols) and other waste products. In 27 cases, higher denitrification rates were achieved than when using methanol. In addition to liquid industrial waste, it is also possible to use gases, especially methane and natural gas. One of the basic conditions for all carbon sources for denitrification is their good biological degradability and low or no toxicity to the nitrification process [30].

*Modified Ludzak-Ettinger system*

Ludzack and Ettinger (LE) [31] developed concurrent system by placing anoxic zone ahead of the oxic zone. LE system was improved by Barnard **[25,32]** providing additional internal recirculation to return mixed liquor back from nitrification to denitrification zone. Such system is in literature referred to as Modified Ludzack-Ettinger (MLE) or pre-denitrification system.

In upstream denitrification in MLE system (Figure 1) WW first flows through an anoxic zone, in which the organically bound nitrogen is hydrolyzed to ammoniacal nitrogen. Ammonia contained in wastewater is consumed in the anoxic zone only for the synthesis of new biomass. In the second, oxic zone, ammoniacal nitrogen is oxidized to nitrites and nitrates. Oxidized forms of nitrogen are transported back to the anoxic zone by internal recirculation (IR) of mixed liquor suspended solids (MLSS) as well as external recycle of return sludge (RS). Heterotrophic microorganisms use organic pollution of WW as a source of carbon, while oxidized forms of nitrogen are reduced to $N_2O$ or nitrogen gas.

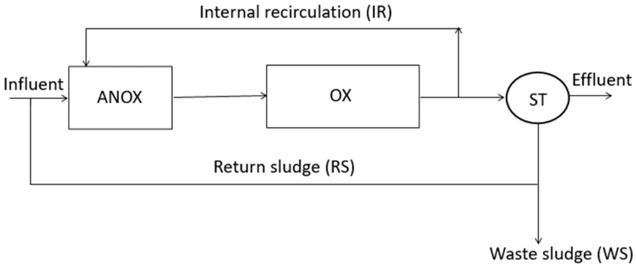

**Figure 1.** Scheme of Modified Ludzak-Ettinger system. ANOX - anoxic zone; OX - oxic zone; ST - settling tank. (ST).

The efficiency of the denitrification process depends on the amount of recirculation. On the other hand, with a higher recirculation, there is a danger that a larger amount of oxygen is introduced into the anoxic zone, which inhibits denitrification. It is possible to prevent this effect, e.g.**,** by including a small anoxic zone at the end of the oxic part of the system. Due to the higher specific rates of denitrification compared to nitrification, the volume of the denitrification part is generally smaller than that of the nitrification zone. This volume is also often used to increase the volume of the oxic part of the activation, especially in the winter months due to a significant decrease in the rate of nitrification compared to the summer period [25].

If there is a sufficiently low concentration of dissolved oxygen, microorganisms use organic pollution from wastewater as a source of organic carbon, and oxidized forms of nitrogen are reduced to $N_2O$ or nitrogen gas. The activity of denitrification affects the amount of recirculated liquid R = (RS + IR) according to the Eqn. (7) [27]:

$$E = \frac{R}{R+1} 100\% \tag{7}$$

From Eqn. (7) follows that at the value of the total recirculation ratio R is 4, about 80% of nitrates are removed by denitrification. A further increase in recirculation would be accompanied by a disproportionate increase in operating costs compared to the achieved effect in terms of denitrification efficiency. In addition, at higher values of recirculation, a larger amount of oxygen can be introduced into the anoxic zone, which inhibits the denitrification process. It is possible to prevent this effect, e.g., by including a small anoxic zone at the end of the oxic part of the system [25]. Another disadvantage of this arrangement is the decrease of the concentration gradient in the oxic part due to internal recirculation, which can be reflected in the deteriorated sedimentation properties of the sludge.

A MLE type of approach can be relatively easily retrofitted into an existing wastewater treatment plant through the installation of baffles, mixers, and an internal recycle capacity. In a diffused air system, the diffusers would have to be relocated to create an anoxic zone and to ensure that there is adequate air for nitrification in the oxic zone. Som diffusers could be left in anoxic zone to provide mixing if the dissolved oxygen concentrations were kept at a low level.

In the MLE system (Figure 1), which is equipped with internal recirculation, the required TN output concentration of 8 mg/l can be achieved. Without internal circulation (the original Ludzack-Ettinger (LE) system) a concentration of 12 mg/l TN could be achieved [27].

The operation of a biological system with complete nitrification is, in the case of sewage WW, accompanied by an approximately 50% increase in the energy required for aeration compared to the removal of organic pollution alone [17]. By appropriate inclusion of the anoxic zone in the technological line, part of the organic pollution is oxidized by anoxic respiration, which will be reflected in the reduction of operating costs associated with the necessary energy for the supply of oxygen. In addition, denitrification can theoretically reuse 62.5% of the oxygen consumed for nitrification. However, due to incomplete denitrification, the real possibilities are around 50%, which roughly covers the costs of mixing the anoxic zone and nitrate recirculation [17].

The authors of [28] present the feasibility of biological nitrogen removal (BNR) using a modified Ludzack Ettinger (MLE) system at a reduced hydraulic retention time (HRT) of 5.5 hours. Complete nitrification was achieved at 75–80% TN removal at a temperature of 12 °C. There was also a reduction in the net observed yield of biomass by 28% compared to the full-scale plant, to 0.31 $g_{VSS}/g_{CODr}$. Compared to the real conventional technology using activated sludge (CAS), the sedimentation properties were also improved (Sludge Volume Index (SVI) value decreased from 202 mlg-1 to 97 ml/g) The values of the heterotrophic biokinetic parameters of the CAS and MLE systems were very similar and consistent with published values for primary runoff. The values of the specific nitrification rates at a temperature of 20 °C in the MLE 0.14 $gNH_4\text{-}N/(g_{VSSd})$ system were 55% higher than in the CAS [28].

*RDN system*

Some shortcomings of the system with upstream denitrification are eliminated by the R-D-N system **[30,31]**. This system is an extension of the upstream denitrification system with a return sludge regeneration zone R. To suppress the excessive growth of filamentous microorganisms, the entrance part of the denitrification zone can be set aside as an anoxic selector (Figure 2).

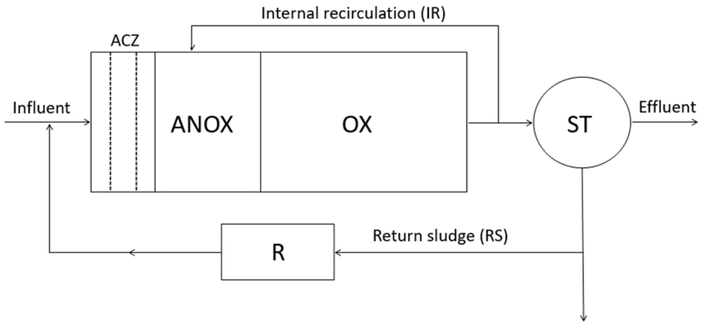

**Figure 2.** R-D-N system. ACZ – anoxic contact zone, ANOX – anoxic zone, OX – oxic zone, ST – sedimentation tank, R - regenerator.

This system is characterized by a 20-25% reduction in required volume compared to pre-denitrification while achieving the same denitrification efficiency. Simultaneously less internal recirculation is also required. The presence of the regeneration zone ensures the restoration of the accumulation capacity of microorganisms and an increase in the aerobic age (SRT) of the sludge. When controlling aeration in the regeneration zone, increase the share of simultaneous denitrification at the level of micro flakes (micro segregation). Increased phosphorus removal was also observed in this process compared to other nitrification-denitrification systems.

### 2.1.2. Cyclical Nitrogen Removal

Alternating aerobic and anoxic zones can be achieved in a batch or continuous flow activated sludge system by alternately switching on and off the aerators. This type of intermittent or pulsed aeration in activated sludge facilities is referred to as cyclical nitrogen removal (CNR) [25]. CNR processes can be advantageously applied in existing activated sludge systems for which the permits for nitrogen removal have also been revised. It only requires small process adjustments, e.g., installation of partitions or timers for the cycling of aeration equipment, or pumps to ensure internal recirculation, pipelines, or fittings to ensure the ability of gradual supply). Thus, compared to a single anoxic zone, potential cost savings can be expected when implementing the CNR process.

*Simultaneous denitrification*

Nitrification and denitrification processes can take place simultaneously in one bioreactor with spatial or time segregation (level of macro segregation level) of oxic and anoxic conditions. Simultaneous denitrification takes place in oxidation trenches and circulation systems with activated sludge in one bioreactor. The creation of oxic and anoxic zones is ensured by the operation of aeration devices. At the same time, the operation of the aerators ensures a sufficient speed of internal recirculation and thus the maintenance of the activated sludge in suspension.

The simultaneous process of nitrification and denitrification (SND) can also take place simultaneously in the aerobic reactor. The conditions for the course of these processes are created inside the structure of activated sludge flocs (micro segregation of oxix/anoxic conditions). The achieved nitrogen removal efficiency without additional carbon is 80 to 96% [5]. The required C:N ratio was 10. The dissolved oxygen (DO) values that need to be ensured are in the range of 0.3 to 0.7 mg/l [33].

The oxidation ditch consists of a ring-shaped channel (Figure) about 1 - 1.5 m deep. Supply of oxygen and maintenance of activated sludge in suspension are provided by aerating rotor consisting of Kessener brush which are placed across the ditch. The mixed liquor circulates at about 0.1 to 0.6 m/s [34]. Arbitrary flow can be expected due to the sequence of the zones with different mixing in the bioreactor and equalization of impact of pollution and hydraulic loads.

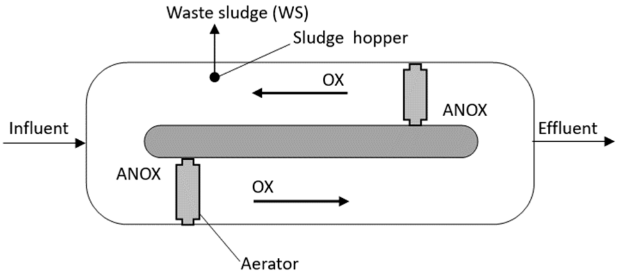

**Figure 3.** Oxidation ditch.

The oxidation ditch was originally developed for small towns in the Netherlands and usually is operated on the principle of a long-term extended aeration process. The endogenous respiration phase of the growth curve is characteristic for this process. Thus, a relatively low organic load (F/M) and long HRT are maintained. High values of SRT and low excess sludge production are also typical for this mode of operation. Wastewater is usually treated without primary sedimentation. Their main disadvantage is the greater demands on the built-up area.

It enables 90 to 95% $BOD_5$ removal efficiency. However, in the case of sufficiently long channels (corridors), it enables the creation of oxic and anoxic zones (macro segregation in space). Controlled aeration can achieve an efficiency of total nitrogen removal in oxidation trenches of around 90% [35].

Oxidation ditches can be operated in the range 8-12 mg/l of effluent TN concentration [25]. The performance of a ditch system cannot, be predicted with as much certainty as other activated sludge systems. Acceptable performance will require field monitoring for optimizing the operation, such as determining the DO profile. The process can then be optimized by adjusting the DO level, which is done by turning aerators on or off or by controlling the oxygen transfer rate by varying the aerator submergence and/or horsepower [22].

*Intermittently aerated CSTR*

For smaller wastewater treatment plants (e.g.**,** oxidation ditch), it is more appropriate to apply systems with time alternation/segregation of oxic and anoxic conditions. Time segregation is an analogy of spatial segregation.

A characteristic feature of systems with time segregation is the high flexibility of changing the length of the cycle, or lengths of the oxic and anoxic periods.

The common of intermittently aerated/operated systems is the possibility of fully automated operation, which creates real conditions for expanding their practical applications.

Intermittent operation regarding oxygen supply can be applied in a batch/semi-continuous mode or a continuous stirred- activated sludge reactor (CSTR). Sedimentation is carried out in a separate sedimentation tank (Figure 4).

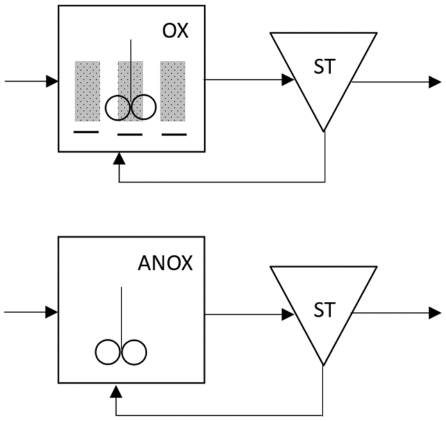

**Figure 4.** Intermittently aerated CSTR. OX- oxic, ANOX- anoxic, ST- sedimentation tank.

In continuously operated systems with time segregation, the activation tank is constantly stirred to prevent sludge settling. Based on a suitable signal (like systems with intermittent operation), air is intermittently supplied to the activated sludge system. During aeration, the concentration of nitrates increases and the concentration of ammonia decreases. In the anoxic period, the course is the opposite [36]. This method of operation is suitable to use e.g.**,** in the intensification of small oxidation trenches for the purpose of nitrogen removal [35], where due to the small lengths of the channels, spatial segregation of oxic and anoxic conditions is not possible. Unlike SBR, this system is close to an ideally mixed system. At low load values, it is necessary to extend the anoxic period to prevent sludge bulking [37].

Intermittent aeration in such a reactor is used intensively for small wastewater treatment plant regarding nitrogen removal [35,37,38].

*Sequential Batch Reactors (SBRs)*

The predecessor of this bioreactor was a batch bioreactor for BOD removal, known as the fill-and-draw process. The reactor was filled with settled wastewater and aerated long enough to oxidize most of the BOD. The contents of the reactor were then allowed to settle, and the treated supernatant was discharged into a receiving water. Part of the settled sludge was withdrawn and the whole process was repeated. However, this reactor fell out of favor due to the amount of operator control required.

With the advent of microprocessor control, a modification of this process, known as SBR, is gaining more and more popularity. It allows to carry out several time-segregated processes such as nitrification and denitrification in the same reactor. Since, in addition to individual biochemical reactions, all other operations (filling WW, removing supernatant, thickening sludge, removing excess sludge) also take place in the same reactor, two or more reactors are needed for wastewater treatment. SBRs are single reactors that cycle through anaerobic, anoxic, and oxic conditions to achieve biological nutrient / nitrogen removal. This technology is particularly suitable where hydraulic and organic loads are highly variable and skilled operating and maintenance personnel are limited [39].

Due to the existing concentration gradient and the controlled alternation of anoxic and oxic conditions, a sufficient decrease in the concentration of organic pollution is usually achieved before the oxic period. Thus, the anoxic period fulfills the function of a selector zone [17].

The SBRs can be operated with less than 8 mg/l TN in an effluent when optimizing DO level [25]. SBRs are convenient for relatively small flow but with high variability.

*Carrousel system*

Challenges to the development of the Carrousel process (Figure 5) were the excessive surface area, and therefore also space requirements, the enormous capital expenditures for surface aerators, and the associated energy costs of designing large capacity oxidation trenches. The Carrousel system has been developed to provide sufficient aeration and maintain adequate flow rates while maximizing surface area utilization [25].

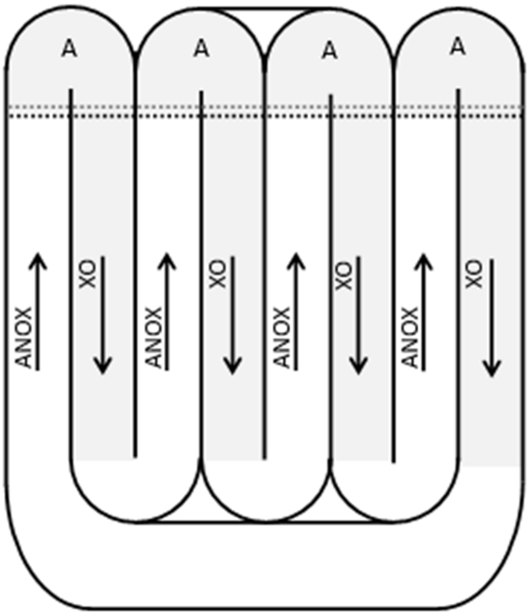

**Figure 5.** Carrousel reactor. ANOX- anoxic, OX- oxic, A- aerator.

This type of activated sludge process represents an innovation of an oxidation ditch with respect to a specific kind of aeration. Vertically mounted mechanical aerators are used to introduce oxygen into this system. At the same time, they provide sufficient horizontal velocity for the activated sludge. Thus, they prevent solids from settling in the reactor channels [40].

Technologically it is an extended activated sludge process, for which high solid retention time (SRT) and hydraulic retention time (HRT), low food to microorganism ratio values and low excess sludge production are representative. From the point of view of the hydrodynamic regime, the Carrousel system corresponds to a series of completely mixing sections with different dissolved oxygen concentrations.

The application of this bioreactor for domestic wastewater treatment (fed with pre-settled sewage and a sludge load ≥ 0.1 kg BOD/kg MLSS.d) relates to severe bulking problems (*Microthrix parvicella*) independently of the type of selector (plug flow or completely mixed) used [41]. This gives another evidence about completely mixed flow in the bioreactor.

The nominal capacity of this bioreactor is significantly higher in comparison to an oxidation ditch. The other advantage of this type of activated sludge process relates to the aerobic stabilization of the activated sludge and the possibility of the exclusion of a primary sedimentation unit in WWTP.

High control flexibility is typical for this type of bioreactor by changing the number of aerators in use, the rotating velocity of aerators and/or the depth of submerging [42].

### 2.1.3. Multi-Anoxic Zones Systems

The current approach should be towards the use of systems with low oxygen and carbon content, which are beneficial for saving energy and preventing environmental pollution.

The four-stage Bardenpho process (Figure 6) has demonstrated its viability in the reconstruction of existing WWTP in Chesapeake, for which the permit was revised with nutrient removal requirements [20].

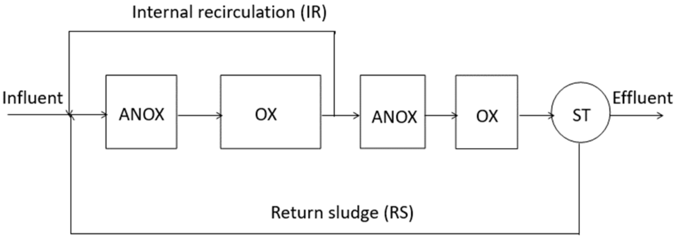

**Figure 6.** Four-stage Bardenpho system.

If sufficient volume of the activation tank is available, its modification to the Bardenpho system may only require the installation of baffles and devices for internal MLSS recirculation. In the case of activation devices in which nitrification was not operated, it will also be necessary to increase the aeration capacity [25,43].

The multi-anoxic zone with step feed arrangement (Figure 7) makes it possible to eliminate some of the disadvantages of pre-denitrification. Recirculation with return sludge is usually sufficient. The efficiency (E, %) of denitrification depends on the number of anoxic-oxic sections n (n = 2 - 4) of the according to the relationship (8):

$$E = 1 - \frac{1}{n(RS+1)} 100\% \tag{8}$$

Figure 6 shows a diagram of the four-stage Bardenpho system.

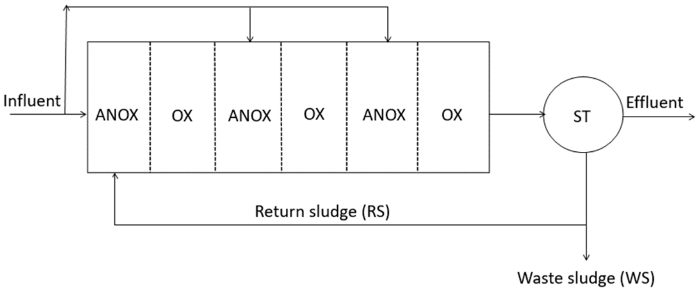

**Figure 7.** Muti-anoxic zone with step feed.

It is obvious from relation (8) that with the return sludge recirculation ratio RS = 1, a 75% denitrification efficiency can be achieved in the four-stage Bardenpho system (Figure 5), resp. in multi-anoxic zone with step feed (Figure 7) 83% efficiency.

For comparison, in the case of a system with upstream denitrification (pre-denitrification MLE, it is necessary to ensure the internal recirculation ratio IR = 2, resp. IR = 5 to achieve the same efficiencies. For higher denitrification requirements, internal recirculation can also be included in 6 stage systems (Figure 7).

In addition to reducing the necessary recirculation, these modifications lead to a gradual equalization of changes in the acid neutralization capacity (total alkalinity) and thus also the elimination of the possible negative impact of decreasing pH on the course of the nitrification process. This is especially advantageous in the case of wastewater with a high nitrogen content, or wastewater with low buffering capacity. Another advantage is the elimination of the influence of internal recirculation on the intensity of axial mixing in individual sections. Due to the cascade arrangement, this system shows a certain concentration gradient. A significant advantage is the smaller total required volume compared to upstream denitrification.

With the required output concentrations of 6 to 8 mg/l TN, it is advantageous to use systems with dual anoxic zones. For example, the Bardenpho system makes it possible to achieve output concentrations of <3 mg/l TN in combination with filtration. For the area where effluent concentrations must be in a range of 6-8 mg/l TN, the application of the Bardenpho system with two anoxic zones without effluent filtration [25] is sufficient.

In systems with separated sludge (separated-sludge denitrification), concentration limits of <3 mg/l TN can also be achieved. If the strict limit for TSS is <10 mg/l, separate-stage denitrification using a downflow filter is required. If effluent filtration is not required, an up flow packed-bed denitrification bioreactor can be used. During the warmer period, it is possible to use cyclic aeration of the nitrification stage to achieve the conditions for denitrification and thereby also reduce the consumption of supplied organic carbon in the denitrification bioreactor, saving operating costs [25].

### 2.1.4. Biofilm-Based Processes

Technologies such as Submerged Aerobic Fixed Film Reactor (SAFF), trickling filters, and biofilters are commonly known as fixed-bed processes in wastewater treatment. A study of nitrogen removal from wastewater plant using a submerged attached growth bioreactor showed 85% average total nitrogen removal [44].

A recent study showed that nitrogen removal of 52–54% was achieved with partial nitritation in sponge trickling filters (STFs) [45]. The report also shows increased nitrogen removal of more than 60% in currently designed trickling plants.

### 2.1.5. Hybrid Systems

In single-reactor hybrid systems with combined suspended and attached growth biomass, a block filling installed in the supporting structure above the aeration elements can be used as a biomass carrier. Another option is the use of submerged modules made of plastic mesh or submerged rotating discs. The growing biomass in the activation tank can also be fixed in suitably shaped plastic particles that move freely in the activation mixture. It is advantageous to use a mesh made of plastic or foam plastic, as the biomass is captured not only on the surface of the carrier particles, but also inside the porous structure. This significantly increased the amount of fixed biomass compared to carriers with a solid surface.

In Germany, the Linpor system was developed for the purpose of intensification of overloaded treatment plants [46]. Polyurethane foam cubes are used as a biomass carrier. The main function of the carrier is to concentrate the biomass in the tank without increasing the material load of the settling tank. The amount of carrier can vary from 10 to 40% depending on the required biomass concentration.

As a result of the operational verification of the Linpor system at the WWTP, where the activated sludge system was fed by WW from the brewery and dairy, a more than two-fold increase in the biomass concentration was determined. Significant decrease in the sludge volume index (SVI) values from 500 ml/g to 85 ml/g was also measured. These changes occurred 30 days after the addition of polyurethane cubes (20% of the tank volume). Nitrification also has started. The nitrogen balance showed that 63% of oxidized forms of nitrogen (created by nitrification or present in WW) were removed by simultaneous denitrification in the anoxic cores of the carrier cubes.

Another system is the Captor system, developed by the University of Manchester [47]. The bioreactor is operated on the principle of a pseudo-fluidized layer. The fluidized state of the biomass carrier made of polyester foam is achieved by compressed air distributed by a system of nozzles at the bottom of the reactor. Part of the carrier is regularly removed from the fluidized layer by a

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mammoth. After straining, the polyester foam particles are mechanically wrung out. The biomass obtained in this way contains up to 6% dry matter. Cleaned particles of the biomass carrier are returned to the reactor by the mammoth. The average concentration of biomass in the reactor is 8 to 12 kg.m$^{-3}$. For municipal wastewater, the retention time in the system is up to 60 minutes.

Intensification of the processes of nitrification and denitrification can also be achieved by applying a hybrid system by combining growth and suspended biomass [4]. The use of growth biomass carriers can achieve a two- to three-fold increase in biomass concentration in the system [46] and, subsequently, a proportional reduction in sludge loading. In this context, it should be emphasized that the above-mentioned increase in biomass concentration will not be manifested by an increase in the loading of the secondary sedimentation tank. The topicality of the application of these systems is also increasing in connection with the need for increased removal of nutrients. Especially with some industrial wastewaters, a negative to toxic effect of higher concentrations of organic pollution on the nitrification process can be expected. On the other hand, by using growth biomass carriers, two biocenoses with different sludge ages are cultivated in the system, which enables the simultaneous course of slower biochemical processes (nitrification, removal of slowly decomposable organic substances) as well as faster processes (removal of easily decomposable organic substances). In addition, depending on the carrier used, there is a certain degree of so-called meso-segregation and subsequently creating conditions for simultaneous nitrification and denitrification processes in the growing biomass.

Drtil et al. [48] measured in the system with 1.5 cm polyurethane cubes a maximum concentration of 8 mg/l $O_2$ at which simultaneous denitrification was still taking place. At the same time, the use of a growth biomass carrier reduces the denitrification volume and thereby also shortens the residence time of nitrifying bacteria outside of oxic conditions. In these systems, an improvement in the sedimentation properties of the sludge compared to the suspended biomass itself is usually observed because of the fixation of fibrous microorganisms in the growing biomass.

### 2.1.6. Advanced Biological Nitrogen Removal Systems

Biological processes Anammox [49], Nitrification and Denitrification [50], Simultaneous Denitrification [51] are among the most effective for the removal of nitrogenous substances from municipal WW, in terms of achieved efficiency (≥ 95%), costs and an environmental friendly nature.

One of the crucial factors in the biological removal of nitrogen from WW is the presence of a sufficient amount of biodegradable organic carbon. However, an external carbon source must be added to increase the effectiveness of the processes [52–54]. If the organic content of wastewater is too low compared to the needed COD/N ratio (analogous to the COD/P ratio for biological enhanced phosphorus removal). In general, the addition of an external carbon source (e.g., methanol, ethanol, acetic acid, glucose), necessary to increase the kinetics of these processes [53,55], together with the costs of disposal of the produced excess sludge, represent the most significant costs in the operation of a WWTP [56,57]. However, as a source of necessary carbon, e.g., WW from the product processes of food, confectionary, milk, and dairy products, thanks to the high content of organic carbon, which is biodegradable can be used [58,59]. while cost savings will be achieved and a lower impact on the environment will be assured. A serious disadvantage could be the insecurity of the qualitative and quantitative characteristics of these WWs, which may be related to the diversity of product cycles [55].

Collivignarelli et al. [60] present results of an innovative biological process applied in sludge line. One of the points of their work was the possibility of recovering the acceptable carbon residue (supernatant) in denitrification processes, replacing bought external carbon sources. Tests on full scale Thermophilic Alternate Membrane Biological Reactor (ThAlMBR) were carried out for 12 months. The ThAlMBR is an advanced biological membrane system used to lysate and oxidize the redundant biological sludge produced by WWTP through thermophilic bacteria, under controlled conditions of temperature and aeration. The ThAlMBR was applied both on thickened (TBSS) and digested sewage sludge (DBSS) with intermittent aeration conditions. Using the respirometric tests, an excellent biological treatability of the supernatant by the mesophilic biomass was observed and the denitrification kinetics reached with the supernatant (4.0 mgN-$NO_3^-$/($g_{VSS}$ h)) and was set up similar to those of methanol (4.4 mgN-$NO_3$−/($g_{VSS}$ h)). Thanks to the analogous results attained on TBSS and DBSS, ThAlMBR proved to be compatible with different sludge line points, maintaining important minimization of sludge production in both cases.

Kim et al. [61], investigated using organic wastes as an alternative to commercial carbon sources. A food waste - recycling wastewater (FWR) was estimated as a convenient alternative to commercial carbon source for natural denitrification. Process stabilization was linked to the acclimatization and function of bacterial populations to the change of carbon source. Stable denitrification performance over a period of seven months was achieved in a full-scale WWTP using FWR as indispensable carbon source. The denitrification performance was stable with a mean nitrate junking effectiveness of 97.2%. These results demonstrate that FWR can be an effective external carbon source that could be beneficial by reducing costs and environmental impacts.

*Nitritation - denitrition*

The process of nitritation - denitritation is also referred to in the literature as a **"**nitrite shunt**"**, which limits the oxidation of nitrites to nitrites and enables the reduction of the produced nitrite to $N_2$ by heterotrophic denitrification. Compared to denitrification, oxygen consumption is 25% lower, organic carbon consumption is 40% lower, and excess sludge production is also lower [62]. The authors of [44] state that by combining the control of the length of aeration and the concentration of dissolved oxygen at a temperature of 15°C, a nitrogen removal efficiency of 95% was achieved in SBR with aerobic granular biomass.

*ANAMMOX system*

A new technology has been developed for the treatment of wastewater with a high nitrogen content, which uses the process of anaerobic ammonium oxidation (ANAMMOX). This process is performed by anaerobic ammonia-oxidizing bacteria (AOB), which oxidize half of the ammonia to nitrite. ANAMMOX bacteria (*Candidatus Brocadia fulgida*) use nitrogen nitrite as the final electron acceptor to oxidize residual ammonia to $N_2$ [48]. In 2014, the authors of [45] from Delft University published on simultaneous partial nitritation and the ANAMMOX process in a suspended biomass system.

Advanced biological nitrogen removal processes (BNR), e.g.**,** nitritation-denitritation and ANAMMOX, therefore, compared to current BNR processes, show a significant reduction in the need for oxygen and organic substrate. The ANAMMOX process further reduces the oxygen demand by 60% without the need for an organic substrate [62].

## *2.2. Phosphorous Removal*

The combination of anaerobic environment (Figure 8), i.e., strictly neither dissolved oxygen nor chemically bounded oxygen is present, with aerobic conditions is necessary in order to remove abundant phosphate from wastewater. Bacteria Acinetobacter spp. and especially the strain Lwoffi were identified as responsible for accumulating excess phosphates in their cells.

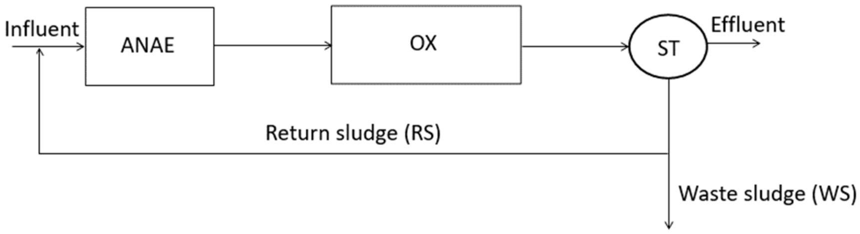

**Figure 8.** Anaerobic/Oxic (AO) system.

Enhanced biological phosphorus removal (EBPR) is provided by microorganisms that are able to accumulate phosphates (PAO). This process requires a sequence of anaerobic and anoxic/oxic conditions [63]. PAOs in anaerobic conditions assimilate volatile fatty acids (VFA) to

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polyhydroxyalkanoates (PHA) inside the cells. They obtain the necessary energy by hydrolysis of intracellular polyphosphate (poly-P). During hydrolysis, orthophosphates are released into the activation mixture. Intracellular glycogen is also transformed into PHA. Thus, the PHA content inside PAO increases and the concentration of glycogen and poly-P decreases. PAOs oxidize PHAs under subsequent aerobic/anoxic conditions, producing energy and organic carbon for the synthesis of new cell mass. The released energy is used by POAs to synthesize poly-P bonds in cells [64]. This released energy is used to synthesize poly-P bonds inside the cell. PAOs usually store more poly-P in the aerobic/anoxic phase than was released in the anaerobic phase [65]. The accumulated poly-P is removed from the biological in the withdrawn excess sludge under oxic conditions. In recent decades, several technological arrangements have been developed to create population of microorganisms enable to remove either nitrogen and phosphorus, or both simultaneously *[66,67*]. The simplest system with increased biological removal of phosphorus in the main stream includes a technological arrangement in which an anaerobic zone precedes the aerobic stage. Return sludge is recirculated before the anaerobic stage. This system was named by its originator Barnard as Phoredox. With such an arrangement, bacteria that are able to accumulate polyphosphates (Poly-P bacteria) have the opportunity to synthesize storage substances. In oxic conditions, these are used preferentially for the release of energy, exceeding their needs. Excess energy is used in the formation of polyphosphates and their accumulation in the cells of microorganisms.

The same configuration was patented by Air Products and Chemicals, Inc. under the label 'Anaerobic/Oxic', or A/O [17].

### *2.3. Nitrogen and Phosphorous Removal*

#### 2.3.1. Current Methods for Nutrient Removal

Biological nutrient removal is a term used to describe the use of microorganisms to take up nitrogen and phosphorus during WW treatment [5]. Nitrogen removal occurs in a two-step process that begins with the use of autotrophs to oxidize $NH_4^+$ to $NO_3^-$ (nitrification), followed by the reduction of $NO_3$ to $N_2$ by heterotrophic microorganisms (denitrification). The successful course of this process is conditioned by the presence of a sufficient amount of organic carbon. Increased biological removal of phosphorus is carried out by a group of microorganisms whose metabolic functions are conditioned by phosphates and thus are able to accumulate them in the form of polyphosphates *[5,68,69*].

After pilot and full operational trials with the Phoredox system, Barnard combined this configuration with upstream denitrification [70]. The result of this combination is a three-stage nitrogen and phosphorus removal system (Figure 9). This system was also later patented by the aforementioned American company under the designation 'Anaerobic/Anoxic/Oxic', or A2/O [70].

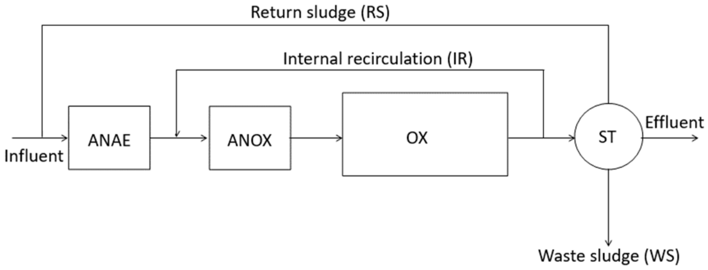

**Figure 9.** Anaerobic/Anoxic/Oxic (A2/O) system.

Another configuration is a five-stage process referred to as "modified Bardenpho" (Figure 10), which was created by combining the Phoredox system (phosphorus removal) and the Bardenpho system (nitrogen removal). Biological stage consists of the sequence: Anaerobic-Anoxic-Oxic-Anoxic-Oxic. Return sludge is fed to Anaerobic zone. There is one internal recycle of nitrates in the system from the outlet of the first oxic zone to the inlet of the first anoxic zone (i.e., from the third to the second tank in the order indicated).

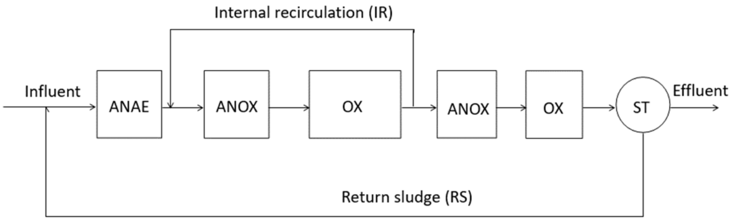

**Figure 10.** Five-stage Bardenpho system.

In the five-stage Bardenpho system, there should be the possibility of an anaerobic zone in case of more sudden higher hydraulic loads or in the case of loads with higher concentrations of dissolved oxygen. An alternative option to ensure operation and control of the process can be a pre-fermentation tank and the division of the anaerobic zone into sections.

The effluent concentrations less than 8 mg/L cannot be consistently obtained using single anoxic zone processes without an additional attached growth filter or external organic carbon supplement TN effluent concentrations less 6 mg/L can be practically attained in a suspended growth system without external organic carbon addition by placing an endogenous anoxic zone in series after the aerobic zone. While the A2/O process does use two unaerated zones, the first (anaerobic) zone is not used for enhanced nitrogen removal but is provided for phosphorous removal or as an anaerobic selector. The first documented case of a second anoxic zone for denitrification was credited to Barnard, depicted schematically in Figure 9. This process served as a precursor to the process he later patented as the Bardenpho process. Phosphorous removal was later accommodated in the Bardenpho process by placing an anaerobic reactor at the head of the treatment train, resulting in a five-stage process also illustrated in Figure 9. The anaerobic fifth stage can be included in facilities that are not required to remove P as an anaerobic selector to suppress the growth of filamentous organisms.

During the 1980s, other alternatives for the simultaneous biological removal of nitrogen and phosphorus were developed, mainly initiated by the knowledge of the negative impact of nitrates on the biological removal of phosphorus due to the use of an easily decomposable substrate by nitrifying bacteria. On this basis, the three-stage Phoredox process was modified at the University of Cape Town to eliminate the nitrate recycle to the anaerobic tank in the return sludge. This process is known as University of Cape Town (UCT) and is schematically shown in Figure 11.

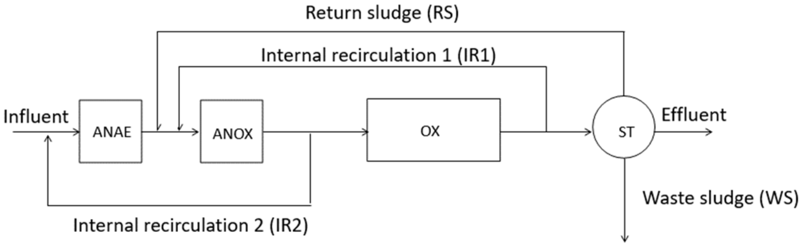

**Figure 11.** UCT process.

The UCT process was also further modified by providing two anoxic zones (instead of one as in the original UCT) and two separate internal recycle lines. The purpose of this modification was to control the return sludge (RS shown in Figure 12) and the nitrate recycle separately and to reduce the $NO_3$ load to anaerobic reactor. Although the Modified UCT process uses dual anoxic zones, the second anoxic zone is not an endogenous denitrification reactor as was described for Bardenpho process. Instead, the second anoxic zone in the Modified UCT is used only to denitrify recycled nitrates from the aerobic zone, and the first anoxic zone is exclusively used asan exogeneous denitrification reactor to denitrify the RS before recycling to the anaerobic zone. This allows increased recycle rates to the second anoxic zone for denitrification and reduces nitrate interference of P removal in the anaerobic reactor.

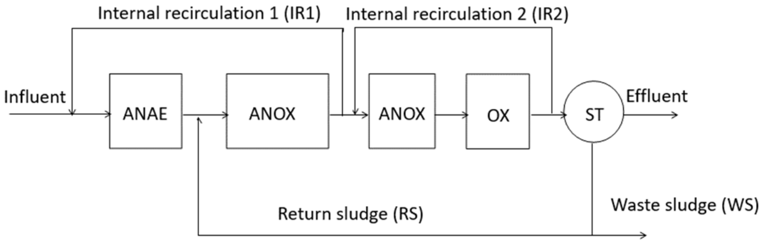

**Figure 12.** Modified UCT process.

The Virginia Plant Initiative (VIP) process was developed and patented by Hampton Roads Sanitation District and CH2M Hill [69]. In VIP method (Figure 13), bioreactors are the same as in the A2/O system.

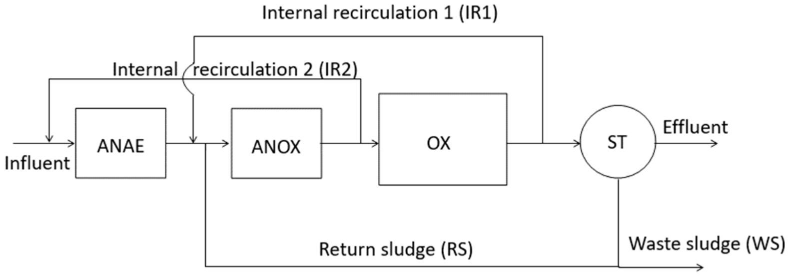

**Figure 13.** VIP process.

However, the returned activated sludge is recycled to the anoxic zone, and the MLSS is recycled from the anoxic zone to the anaerobic tank (IR2). Nitrate is recycled from the oxic zone to the anoxic tank for denitrification (IR1), thus reducing the amount of nitrate fed to the anaerobic zone in return activated sludge (RS).

Heidari [71] used BioWin software to design, upgrade, and optimize wastewater treatment plants by comparing nutrition removal efficiency between A2O and VIP process. Results of simulations shown that even though both the VIP and A2/O achieved excellent nutrition removal, the VIP could remove more nutrition than A2/O.

Usually, the effluent concentration of 8 mg/l TN is presented as the performance of single anoxic zone systems A2/O, UCT, and VIP systems. However, the concentrations in the range 6 to 8 mg/l TN can be maintained with flexible operational abilities. The decrease of limit TN value must be compensated with enhancement of the internal mixed liquor recirculation from oxic to anoxic zone. Internal recirculation ration 1 to 4 should be provided [25]. An additional denitrification zone using an external carbon source can be used to remove low nitrate concentrations.

With the required output concentrations of 6 to 8 mg/L TN, it is advantageous to use systems with dual anoxic zones. For example, the Bardenpho system makes it possible to achieve output concentrations of <3 mg/l TN in combination with filtration. For the area of output concentrations of 6-8 mg/l TN, the application of the Bardenpho system with two anoxic zones without effluent filtration [25] is sufficient.

In systems with separated sludge (separated-sludge denitrification), concentration limits of <3 mg/l TN can be achieved. If the strict limit for TSS is <10 mg/l, separate-stage denitrification using a downflow filter is required. If effluent filtration is not required, an upflow packed-bed denitrification bioreactor can be used. During the warmer period, it is possible to use cyclic aeration of the nitrification stage to achieve the conditions for denitrification and thereby also reduce the consumption of supplied organic carbon in the denitrification bioreactor, saving operating costs [25].

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If moderate levels of TN removal are required, single sludge, single anoxic zone systems such as the A2/O, VIP and UCT processes can be selected for phosphorous removal. These systems can maintain effluent WW with less than 1 mg/l TP. However, TP concentrations in the effluent are variable. The choice between the A2/O, VIP and UCT processes is mainly influenced by the characteristics of the WW.

Reliable performance to less or equal 1 mg/L TP will require chemical precipitation in addition to biological P removal. As discussed previously, dual anoxic zone systems are typically applied where low levels of TN are also required. In this case, the modification or five-stage Bardenpho system would be considered since it includes an anaerobic zone at the beginning of the process line. Effluent TP levels of less than 3 mg/L have been reported with these systems [25].

To obtain TP limit less to 2 mg/L and consistently maintain an effluent with less than 2 mg/L TP, chemical precipitation should be included in the treatment line. Biological nutrient removal systems for both N and P removal systems present some unique operating circumstances since the two processes have conflicting requirements. The mechanisms for biological phosphorous removal in a mainstream technology such as UCT, A2/O, VIP, or modified Bardenpho system is through the assimilation of enhanced levels of P in the bacteria selected by anaerobic zone and subsequent elimination of P through waste sludge. Thus, biological P removal is optimized at a shorter solid retention time (SRT) than is typically applied for nitrification. Sludge production and subsequent P removal is promoted at shorter SRTs. Generally, the compromise is to operate the systems at the shortest SRT required to achieve effluent nitrogen limits [25].

Nitrates also present a problem for biological phosphorous removal processes. For enhanced P removal to proceed, the bacteria must first assimilate organic substrate in the anaerobic zone and store them within their cells. Nitrates that may be present in the RAS allow facultative heterotrophic bacteria to compete for soluble substrate required by the P-removing bacteria. This can degrade the phosphorous removal performance of systems as the A2/O, or modified Bardenpho system with recycle RAS directly to the anaerobic zone. The UCT and VIP processes avoid this problem by returning the RAS o the anoxic zone where the nitrates are reduced by denitrification. Mixed liquor is than recycled from downstream of the anoxic zone to the anaerobic zone [25].

A key criterion for BNR systems to remove both N and P is TBOD5: TP ratio. If this ratio is larger than 20:1, than the recycle of nitrates to the anaerobic zone may be not a problem because there is an adequate supply of excess organic substrate. In this case, either the A2/O, or modified Bardenpho system should be considered for stricter limits on TN. These systems do not have the costs associated with additional recycle from the anoxic to the anaerobic zone. If the TBOD5: TP ratio is less than 20:1, then the VIP or UCT process should be considered. For the same reasons, the TKN: BOD is important. A high TKN: BOD may indicate there is insufficient carbon to denitrify completely, resulting in a carryover of nitrates to anaerobic zone [25].

For WW treatment plants that must remove P and are required to nitrify but no denitrify, it may still be advantageous to include denitrification as a component of single-sludge system. This will reduce the interference of nitrates with the biological phosphorous removal mechanisms and will al3so provide the cost savings associated with the return of alkalinity and reduction in aeration requirements. Mainstream biological phosphorous-removal system will require effluent filtration to achieve lower levels of TP than 0.5 mg/L. This is because the solids that are carried by the clarified effluent will contain an enhanced level of phosphorous, which can have a significant impact on the effluent TP content. Typically activated sludge solids will contain approximately 2% of P, whereas the solids from a biological P removal system will contain typically 4-8% of phosphorous [25].

In the UCT, A2/O, and VIP systems the concentratios of TN in the range 8-12 mg/l can be meet [25]. The UCT or VIP technologies would be convenient in cases, when high phosphorous removal is required and the BOD5:P ratio <20:1. Both these systems are high-rate with low SRT and short HRT [25].

Usually, the effluent concentration of 8 mg/l TN is presented as the performance of single anoxic zone systems A2/O, UCT, and VIP systems. However, the concentrations in the range of 6 to 8 mg/l TN can be maintained with flexible operational abilities. The decrease of limit TN value must be compesated with enhancement of the internal MLSS recirculation from oxic to anoxic zone. Internal recirculation ration 1 to 4 should be provided [25]. An additional denitrification zone using an external carbon source can be used to remove low nitrate concentrations.

### 2.3.2. Advanced Method for Nutrient Removal

The authors of [5] analyse information on current and advanced technologies used to remove N and P from domestic WW. They discuss the mechanisms, advantages, disadvantages and limitations of these technologies. They recommend an integrated approach to nitrogen and phosphorus removal. They consider the process of simultaneous nitrification and denitrification (SND) to be an advantageous method for removing nutrients from domestic WW due to its operational simplicity, cost-effectiveness, low energy demand, efficiency and minimum demand for pH regulation.

Phosphorus denitrification removal appears to be a promising process that provides remarkable energy savings, lower sludge production, increased efficiency, and reduced carbon source conditions. It can thus effectively use the limited carbon resources present in domestic WW.

The integration of DPAO into the SND process is recommended to realize DPAO-mediated denitrification in a simultaneous nitrification-denitrification-phosphorus removal (SNDPR) system. It is also recommended that a combination of archaeal ammonium and anammox modified denitrification phosphorus removal processes to consolidate nitrogen removal and phosphorus recovery from low carbon/nitrogen wastewaters in normal operation without adding an external carbon source.

*Denitrification removal of phosphates*

Phosphorus-accumulating organisms (PAOs) that mediate enhanced biological phosphorus removal (EBPR) require alternating anaerobic and aerobic/anoxic conditions for optimal phosphate uptake. The effectiveness of such approaches requires separate unit operations and control experimental conditions, increasing operational difficulties and technology costs.

However, in the literature, one can find many claims about the existence of some heterotrophic denitrifiers that are able to remove phosphorus under anoxic/aerobic conditions, i.e., without the need to alternate anaerobic and aerobic/anoxic conditions [72–76]. Heterotrophic microbes are characterized as denitrifying phosphorus-accumulating organisms (DPAO).

These are facultative anaerobic bacteria that can use nitrate and/or nitrite as the final electron acceptor instead of oxygen. The process of phosphorus removal under anoxic/aerobic conditions is also known as denitrification phosphorus removal. Phosphorus removal and denitrification occur simultaneously [74].

According to the literature, both PAO and DPAO have enzymes for aerobic metabolism. However, only DPAOs are capable of denitrifying phosphorus uptake. On the other hand, PAOs lack the essential enzymes to utilize nitrate as the final electron acceptor [77]. DPAOs are able to synthesize polyphosphate by the action of the enzyme polyphosphate kinase (PPK) in the presence of external carbon sources under aerobic or anoxic conditions, i.e., without alternating anaerobic and aerobic/anoxic conditions [78].

The advantages of denitrifying absorbed phosphorus include some improvements over EBPR methods. These are energy savings, less sludge production [79] and less requirements for the carbon source. DPAOs use the same carbon source for both denitrification and phosphorus removal, resulting in approximately 50% savings in COD demand [80]. The simultaneous removal of nitrogen and phosphorus can effectively utilize the limited carbon resources present in domestic wastewater with a moderately low carbon to nitrogen ratio [75].

Recently developed advanced biological nutrient removal (BNR) technologies create conditions for stable and high-performance N and P removal. They differ from current BNR technologies in several ways [80,81]. They are aimed exclusively at improving the efficiency of BNR, saving energy and operating costs. These technologies include e.g., simultaneous nitrification/denitrification (SND), bio augmentation batch enhanced treatment (BABE), anaerobic ammonium oxidation (anammox), denitrifying phosphorus removal, single reactor for high activity ammonia removal over nitrite (SHARON), etc. [82–85].

*Heterotrophic nitrification and aerobic denitrification*

Heterotrophic nitrification and aerobic denitrification (HNAD) solves the shortcomings of current biological nitrogen removal systems. It is known that these systems are implemented discretely in two separate anoxic and oxic bioreactors due to the different requirements of the specifics of the individual processes and the microorganisms used (slower speed of autotrophic bacteria, their sensitivity to high organic load, sensitivity of denitrifying bacteria to the presence of oxygen). Such a solution requires strict compliance with the necessary conditions, which translates into high costs [12]. Recent research works have demonstrated the existence of some heterotrophic nitrifiers that are able to perform nitrification using organic carbon *[86–88]*. Many of these heterotrophic nitrifiers are able to denitrify their nitrification products, i.e., nitrates and/or nitrites, to nitrogen gas [89]. These microorganisms can thus use dissolved oxygen and nitrates at the same time and enable nitrification and denitrification processes to be carried out in one reactor under oxic conditions. They use nitrates/nitrites as the final electron acceptor. This process is referred to as the SND process [90].

SND through HNAD provides many advantages compared to current technologies by removing N. It is cost-effective, as evidenced by a 22–40% reduction in carbon source consumption and about a 30% reduction in sludge production [72]. The implementation of nitrogen removal processes in one reactor is much simpler. Higher efficiency due to higher growth rates of heterotrophic microorganisms and their ability to use organic carbon as an energy source as well as the ability to use the products of the nitrification process as reactants for denitrification are reflected in the higher efficiency of this bioreactor. The low DO requirement due to the reduced intensity of aeration and the minimization of recirculation translate into lower energy consumption minimal need for pH control. The alkalinity generated during denitrification partially balances the acidity generated during nitrification [91]. The variety of substrates and products of heterotrophic nitrification allows to expand the application possibilities [92]. HNAD technology also represents an alternative for domestic WW treatment with a low C/N ratio and/or CO2 limitation for autotrophic nitrifying bacteria [93].

This technology overcomes the limitations of current nitrogen removal technologies and provide a viable technology for biological nitrogen removal.

*Integrative approaches for simultaneous nutrient removal and recovery*

It is well known from the literature that SND can offer a one-step treatment process design for nitrogen removal *[65,94,95]*.

As previously mentioned, denitrification phosphorus removal can simultaneously remove nitrogen and phosphorus under either aerobic or anoxic conditions in a single reactor without alternation of aerobic/anaerobic (anoxic) conditions [96]. Thus, under these conditions, DPAOs can be incorporated into the SND process to develop a simultaneous nitrification-denitrification-phosphorus removal (SNDPR) system. In this process, there is a difference from the conventional enrichment of biomass with phosphorus by the EBPR mechanism to the enrichment of the liquid phase with phosphorus mediated by DPAO. Phosphorus from the enriched liquid phase can be easily recovered by induced crystallization, which is significantly simpler than, as was the case for phosphorus removal and recovery in the EBPR system [97]. Connecting DPAO to systems for simultaneous nutrient removal and recovery will certainly be the most ecological and efficient approach. The application of a promising energy- and resource-efficient alternative to anammox in the mainstream would certainly be successful. However, the selective enrichment of AOB and the simultaneous inhibition of NOB due to a stable supply of nitrites remains a fundamental barrier that prevents its practical implementation [85]. Ammonia-oxidizing archaea (AOA) with high efficiency of ammonia oxidation under oxygen-limited conditions are potentially a more suitable microorganism for supplying nitrite and could be an option to link with mainstream anammox [98]. The review [76] proposes another integrated energy-efficient approach by combining archaeal oxidation of ammonium-anammox with modified denitrification. Due to the reduced carbon source requirement and the fact that DPAOs use the same carbon source for both denitrification and phosphorus removal, tsa saves approximately 50% of COD requirements [80]. The simultaneous

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removal of N and P can effectively use limited carbon sources in domestic WW with a moderately low carbon to nitrogen ratio [82].

## 3. Nutrient Recovery

Nutrients are essential elements for sustaining life, including nitrogen and phosphorus compounds that are crucial for various biological processes [99]. Nitrogen and phosphorus play a key role in the structure and function of living organisms. Nitrogen is an essential component of cellular DNA, amino acids and chlorophyll, while phosphorus serves as the cells' "energy currency" in the form of adenosine triphosphate (ATP) [100,101]. In agricultural systems, nitrogen and phosphorus play a critical role in promoting plant growth and ensuring the food supply [100].

Despite the abundance of nitrogen in the atmosphere, its availability in the soil is limited, requiring fixation into reactive forms such as amino acids, nitrates, and ammonia to increase its availability for plant uptake [100]. However, natural fixation processes are not sufficient to meet the demands of the growing global population, leading to an increasing trend of anthropogenic production of reactive nitrogen [102]. Conversely, phosphorus faces depletion problems as a non-renewable resource that is primarily obtained from igneous and sedimentary deposits through mining [100]. Current extraction rates indicate that easily exploitable phosphorus resources will be depleted within the next few decades [103].

The importance of nitrogen and phosphorus in agricultural productivity cannot be overestimated. Synthetic fertilizers, composed mostly of nitrogen and phosphorus compounds, have played a key role in bridging the gap between malnutrition and adequate nutrition for a significant portion of humanity [100,104]. However, inefficient use of nutrients leads to significant losses, with only a fraction of the reactive nitrogen used accumulating in crops, while the rest escapes to the environment through air and water. These losses not only affect human health and biodiversity, but also contribute to environmental problems such as global warming and eutrophication of water bodies [105].

To address these challenges, increasing emphasis is being placed on nutrient recovery as a means of reducing the load on reactive nitrogen and phosphorus production and at the same time mitigating environmental degradation. Nutrient recovery works on the principles of reduce, reuse, and recycle to reintroduce recovered nitrogen and phosphorus as fertilizers to increase agricultural productivity while minimizing adverse environmental impacts. By effectively managing nutrient cycles and reducing nutrient losses, nutrient recovery holds promise for ensuring sustainable resource management, reducing food production costs, and promoting environmental protection [100].

### *3.1. Ion Exchange*

Reactive nitrogen, primarily existing as $NH_4^+$, is effectively recovered using ion exchange and adsorption-based processes. Zeolite, a popular ion exchanger and adsorbent for nitrogen recovery, offers a tetrahedral framework where aluminum and silicon atoms form interconnected cages and channels [100,106]. Substitution of silicon with aluminum creates a net negative charge within the zeolite framework, balanced by exchangeable cations such as $Na^+$, $K^+$, $Ca^{2+,}$ and $Mg^{2+}$ on the zeolite surface.

Performance of zeolites in nitrogen recovery is influenced by various factors such as pH, influent $NH_4^+$ concentration, temperature, and ionic strength. Modification techniques, including treatment with acids, alkalis, and salts of alkaline metals, enhance the adsorption capacity and purity of zeolites.

Once zeolites are exhausted, nitrogen recovery and reuse opportunities are explored. Regeneration techniques, such as NaCl solution regeneration, acid regeneration, regeneration with heath, or biological regeneration, provide concentrated streams of $NH_4Cl$ or $NaNO_3$ for further processing. In these concentrated regenerant solutions, after raising their pH above 9.3, $NH_3$ can be air stripped off, with $NH_3$ gas further processed for fertilizer production [100].

Ion exchange processes are also effective for removing and recovering phosphorus from wastewater, typically through adsorption onto phosphorus-selective media. These media can exchange anions like $Cl^-$ with $PO_4^{3-}$ ions, producing phosphorus-depleted effluent while maintaining charge neutrality in the solid media [107].

The advantages of ion exchange for phosphorus removal include its ability to work in varying wastewater compositions, flow rates, and temperatures. However, drawbacks include poor selectivity towards $PO_4^{3-}$-P, inefficient regeneration, and loss of loading capacity [106,108].

To maintain the efficiency of ion exchange media, periodic regeneration is necessary, adding to the operational costs of water treatment. Ion exchange treatment is suitable for removing phosphorus from effluents with concentrations below 2000 mg/l. Despite the promising potential of using zeolites for nutrient recovery, the process remains costly, particularly for large-scale wastewater treatment plants [106].

### *3.2. Amonnia Stripping*

Ammonia stripping, a pH-dependent process, is widely used for nitrogen removal in wastewater treatment. At a pH of around 9.3, the ammonium nitrogen in the solution is converted to ammonia gas, which is facilitated by the addition of lime or caustic soda to maintain the pH around 10.8-11.5. This conversion allows efficient removal of ammonia gas by air stripping, especially in packed towers that provide a large surface area for mass transfer.

Ammonia stripping efficiency is affected by a variety of factors, including ammonia concentration, hydraulic loading, air flow, pH, and temperature. Studies have shown that increasing pH increases the efficiency up to 92% of ammonia recovery, with the optimum pH range typically between 8 and 11. Another study report that ammonia removal efficiencies range from 80% to 96.6% at optimal pH and aeration times, highlighting the effectiveness of the process in recovering nutrients from wastewater streams.

Similarly, air stripping time also affects ammonia removal, with optimum removal being achieved within specific time frames [100].

Optimization studies by Guo et al. [109] showed a significant increase in ammonia removal efficiency with pH values up to 11. Furthermore, the increase in removal was not substantial. Air stripping time optimization studies showed that an 18-hour duration resulted in optimal ammonia removal efficiency [109].

Recent advances in ammonia removal include the development of innovative gas-liquid contactors, such as water jet aerocyclones (WSA), which exhibit higher stripping efficiency and superior mass transfer efficiency compared to traditional methods such as stripping tanks and packed towers. These improvements help reduce operating costs and improve overall processing efficiency.

Ammonia stripping has been used successfully in large-scale treatment in municipal wastewater treatment plants, offering an effective solution for nitrogen removal. However, issues such as pollution of loaded towers and scaling issues need to be addressed to ensure continuous and efficient operation [110].

### *3.3. Membrane Separation*

Membrane separation processes have emerged as key technologies in wastewater operations, offering efficient and selective removal of nitrogen and phosphorus compounds from various wastewater and liquid waste streams.

Nunes and Peimmann [111] demonstrated the regeneration of gaseous ammonia in an acidic solution using a gas-permeable membrane operated under vacuum pressure. Hasanoglu et al. [112] achieved the recovery of ammonia as $(NH_4)_2SO_4$ on the permeate side of a macroporous hydrophobic membrane. Kurama et al. [113] reported a substantial 96.9% recovery of ammonium ions using reverse osmosis (RO). Gerardo et al. [114] used crossflow microfiltration (MF) to recover nitrogen and phosphorus from dairy farm sludge. Mondor et al. [115] produced a concentrated nitrogen fertilizer from liquid pig manure using electrodialysis and reverse osmosis, albeit with significant ammonia volatilization.

Membrane nitrogen recovery processes offer significant advantages including independence from gas or liquid flow rates, absence of secondary pollutants in the ammonia permeate, and consistent efficiency unaffected by changes in ammonia concentration [100].

In scenarios where phosphorus concentrations approach regulated effluent levels, rapid sand filtration provides a simple solution to reduce tertiary phosphorus, primarily associated with solid organic matter [116]. However, rapid sand filtration lags behind in dissolving dissolved phosphorus and does not provide an easily renewable source of phosphorus. In contrast, membrane filtration has gained prominence due to improved performance and cost-effectiveness.

Tertiary membrane filters, reverse osmosis (RO) systems, and membrane bioreactors (MBRs) have become common in phosphorus recovery, allowing for the collection of both dissolved and solid phosphorus, thereby increasing phosphorus removal efficiency. These technologies have demonstrated effluent concentrations as low as 0.008 mg P/l for RO systems [117].

Forward osmosis (FO), membrane distillation (MD), and electrodialysis (ED) are three key membrane technologies used for nutrient recovery, each with its own unique advantages. FO uses a semipermeable membrane to enrich nutrients on the feed side with the potential for larger phosphate enrichment due to electrostatic repulsion. In MD, heated feed solutions facilitate the conversion of ammonium ions into volatile ammonia, which can be obtained by drawing solution. ED uses cathexis and anion exchange membranes to separate ammonium and phosphate ions, thereby controlling their concentration and regeneration in different chambers [107].

Membrane filtration technologies, such as those based on RO membranes, forward osmosis (FO) membranes, microfiltration (MF) membranes, and nanofiltration (NF) membranes, offer practical potential for the recovery of nutrients, especially phosphorus concentrations. These technologies have been investigated in various combinations, such as hybrid FO-MD systems and hybrid membrane bioreactors (MBR), which show high efficiency in phosphorus recovery.

Despite promising results at the laboratory scale, challenges such as membrane fouling, high cost, and the need for further investigation of salinity control methods remain an obstacle to the widespread practical application of membrane technologies for nutrient recovery [118].

### *3.4. Chemical Precipitation*

Removal of phosphorus from wastewater is often achieved by chemical precipitation using salts of divalent or trivalent metals such as calcium, aluminum or iron [66,119]. Common precipitants such as lime ($Ca(OH)_2$), alum ($Al_2(SO_4)_3·18H_2O$) and ferric chloride ($FeCl_3$) are often used in wastewater treatment plants [100]. The addition of these chemicals initiates the formation of insoluble phosphorus compounds, primarily iron or aluminum phosphate complexes, which are then separated from the liquid phase by settling or filtration [66].

pH adjustments play a critical role in the effectiveness of chemical precipitation. Ferric phosphate precipitates show minimum solubility in the pH range of 4.0–6.0, while aluminum phosphate precipitates achieve minimum solubility at pH 5.0–7.0. However, achieving optimal pH levels may require alkalinity adjustment, especially when using calcium-based precipitation methods such as calcium apatite ($Ca_{10}(PO_4)_6(OH)_2$), which requires a high pH (>9.0) for precipitation [100].

Despite the effectiveness of chemical precipitation in removing dissolved phosphorus, the recovery of phosphates from the resulting sludge remains impractical without a tertiary infrastructure for chemical reduction [118]. In addition, the use of chemicals such as lime or alum significantly increases sludge production during primary and secondary treatment, with estimates of up to a 35% increase in sludge volume after precipitation [100].

Struvite, a crystalline compound composed of magnesium, ammonium and phosphate ions, spontaneously precipitates in wastewater treatment plants under specific conditions, especially at elevated pH values (>8.0) and turbulent flow. While struvite formation presents operational problems such as pipe clogging, it also offers an opportunity for effective nutrient recovery, particularly phosphorus and nitrogen, by acting as a slow-release fertilizer [118].

Struvite restoration involves dosing the influent with magnesium compounds to induce supersaturation and promote precipitation. Maintaining a stoichiometric molar ratio of magnesium to phosphate is crucial, typically exceeding a ratio of 1:1. However, achieving optimal pH values for struvite precipitation may require aeration or the addition of alkaline agents, thus balancing cost effectiveness and operational efficiency [100].

### *3.5. Biological Processes*

Microorganisms, including bacteria and fungi, play a key role in nutrient cycling and can be used to recover N and P from waste streams. Microbial fuel cells (MFCs) represent a promising avenue for simultaneous energy production and nutrient recovery. In these systems, microorganisms oxidize organic matter at the anode, releasing electrons that flow through an external circuit to the cathode, where they drive the reduction of nitrate and phosphate ions, thereby facilitating N and P regeneration. In addition, some bacteria, called nitrogen-fixing bacteria, can convert atmospheric nitrogen gas (N2) to ammonia ($NH_3$) through nitrogen fixation, thus offering a sustainable source of nitrogen for agricultural applications [100].

*Algae-based N and P recovery*

Algae, including microalgae and macroalgae, are capable of assimilating and accumulating N and P from wastewater. Microalgae such as *Chlorella* and *Scenedesmus* show high rates of nutrient uptake and are often used in wastewater treatment systems. These microalgae can remove $NH_4^+$ and $NO_3^-$ from wastewater by biological uptake and simultaneously release oxygen through photosynthesis [118]. In addition, the uptake of luxury phosphorus by microalgae, where phosphorus is stored as polyphosphate in biomass, represents a promising route for obtaining phosphorus. Macroalgae, including water including animal feed and biofuel production [119].

Microalgal growth and furthet microalgae-based nutrient recovery during wastewater treatment can be affected either positively or negatively by several biotic and abiotic factors. Biotic factors include the presence of bacteria, fungi, viruses, and other microalgae, while abiotic factors include light, temperature, pH, salinity, nutrient, dissolved oxygen concentration, and the presence of toxic compounds [119–121,123].

*Recovery of N and P on the basis of macrophytes*

Macrophytes, such as aquatic plants including duckweed and water hyacinth, play an important role in the removal and recycling of nutrients in aquatic ecosystems. These plants have the ability to absorb and accumulate N and P from water through their root systems, thus providing a natural and sustainable way to renew nutrients. Duckweed has proven effective in removing $NH_4^+$ and $NO_3^-$ from wastewater, while the collected biomass serves as a potential raw material for the production of bioethanol. In addition, emergent macrophytes planted in constructed wetlands can enhance nutrient removal through physical uptake, precipitation, and biological assimilation processes [118].

## 5. Conclusion

Nitrogen and phosphorous are inevitable for the growth and vitality of living beings. However, the discharge of nutrient-rich wastewater into aquatic ecosystems leads to eutrophication. Thus, nutrient N and P effective removal from WW is way to meeting strict nutrient discharge standards. In addition, nutrient recovery from WW is equally important for realizing a circular economy by preventing the depletion of limited resources.

The creation of this article was motivated by a draft revision of the currently valid “Urban Wastewater Treatment Directive”.

The choice of nutrient removal technology is marked not only by the characteristics of wastewater, the required quality of treated water and investment options, but also by the stage of decision-making and selection.

The previous period in the treatment practice of the EU Member States was aimed at ensuring a sufficient reduction of discharged organic pollution (secondary treatment) as well as at the intensification and reconstruction of tertiary treatment with the aim of ensuring the required quality of discharged wastewater and thus also ensuring the sustainable protection of water resources.

The evaluation of the previous period showed that positive results were achieved. At the same time, however, new ambitious goals emerged in a comprehensive approach to the protection of not only the water ecosystem but also human health.

The aim of this paper is to provide operators and designers with a brief overview of selected traditional and advanced processes, reactors and technologies for nutrient removal from municipal wastewater. Another goal is to discuss the possibilities and limitations to comply with more strict effluent standards.

From the evaluation of the published papers, it follows that the currently applied traditional methods of nutrients removal have the potential to convey also with the expected stricter limits.

There are more options to intensify the currently operating treatment plants. It is well known that the efficiency of the nitrification process can be significantly affected by the SRT value. Another possibility of intensifying the process is by applying hybrid growth and attached biomass with the use of biomass carriers.

The denitrification rate and efficiency can be affected by the internal recirculation rate, hydraulic retention time and addition external organic carbon.

ln some cases it may be possible to improve the performance of a process by modifying its operation or by adding relatively minor equipment or process improvements. For example, the nitrogen removal performance of a system with a single anoxic zone may be improved-within limits by increasing the mixed liquor recycle rate.

Cyclical nutrient removal (CNR) processes can be advantageously applied in existing activated sludge systems for which the permits for nitrogen removal have also been revised. It only requires small process adjustments, e.g., installation of partitions or timers for the cycling of aeration equipment, or pumps to ensure internal recirculation, pipelines, or fittings to ensure the ability of gradual supply). Thus, compared to a single anoxic zone, potential cost savings can be expected when implementing the CNR process.

Usually, the effluent concentration of 8 mg/l TN is presented as the performance of single anoxic zone systems A2/O, UCT, and VIP systems. However, the concentrations in the range 6 to 8 mg/l TN can be maintained with flexible operational abilities. The decrease of limit TN value must be compensated with enhancement of the internal mixed liquor recirculation from oxic to anoxic zone. Internal recirculation ration 1 to 4 should be provided. An additional denitrification zone using an external carbon source can be used to remove low nitrate concentrations.

If sufficient volume of the activation tank is available, its modification to the Bardenpho system may only require the installation of baffles and devices for internal MLSS recirculation. In the case of activated sludge process in which nitrification was not operated, it will also be necessary to increase the aeration capacity.

New microorganisms that can remove nitrogen through new metabolic pathways were discovered. These include ammonia oxidizing archaea (AOA), complete ammonia oxidation bacteria (COMAMMOX), anaerobic ammonium oxidation associated with iron reduction bacteria (FEAMMOX), anaerobic ammonium oxidation bacteria (ANAMMOX) and denitrifying anaerobic methane oxidation microorganisms (DAMO).

Compared to traditional nitrifying or denitrifying bacteria, these new microbial groups have better physicochemical tolerance and/or lower greenhouse gas emissions. They represent promising microbial communities that could provide new technologies for high-throughput and energy-efficient nitrogen removal from wastewater.

To obtain TP limit less to 2 mg/l and consistently maintain an effluent with less than 2 mg/l TP, chemical precipitation should be included in the treatment line.

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**Abbreviations**

| | |
|---|---|
| ACZ | anoxic contact zone |
| ANOX | anoxic zone |
| ANAMMOX | anaerobic ammonium oxidation bacteria |
| AO | anaerobic/oxic |
| AOA | ammonia oxidizing archea |
| AOB | ammonium oxidizing bacteria |
| ATP | adenosine triphosphate |
| BNR | biological nutrient removal |
| BOD(s) | biological oxygen demand(s) |
| C | carbon |
| CAS | conventional activated sludge |
| CNR | cyclical nitrogen removal |
| COD | chemical oxygen demand |
| COMAMMOX | complete ammonia oxidation bacteria |
| CSTR | continuous stirred-tank reactor |
| DAMO | anaerobic methane oxidation denitrifying microorganisms |
| DBSS | digested biological sewage sludge |
| DO | dissolved oxygen |
| DPAO(s) | denitrifying polyphosphate-accumulating microorganism(s) |
| E | efficiency |
| EBPR | enhanced biological phosphorous removal |
| ED | electrodialysis |
| EU | European Union |
| FEAMMOX | iron reduction bacteria |
| FWR | food waste-recycling wastewater |
| HNAD | heterotrophic nitrification and aerobic denitrification |
| HRT | hydraulic retention time |
| IR | internal recirculation |
| LE | Ludzack- Ettinger system |
| MBR | membrane bioreactor |
| MD | membrane distillation |
| MF | microfiltration |
| MFCs | microbial fuel cells |
| MLE | modified Ludzack-Ettinger (system) |
| MLSS | mixed liquor suspended solids |
| N | nitrogen |
| O | oxic zone |
| P | phosphorous |
| PAO (s) | polyphosphate-accumulating microorganism(s) |
| PE | population equivalent |
| PHA | polyhydroxyalkanoates |
| PPK | polyphosphate kinase |
| R | recirculated liquid |
| R | regenerator |
| RO | reverse osmosis |
| RS | return sludge |
| SAFF | Submerged Aerobic Fixed Film Reactor |
| SBR | sequential batch reactor |
| SHARON | single reactor high activity ammonia removal over nitrite |
| SND | simultaneous nitrification and denitrification process |

| | |
|---|---|
| ST | sedimentation tank |
| SRT | solids retention time |
| SVI | sludge volume index |
| SWO | StormWater Overflows |
| TBSS | thickened biological sewage sludge |
| ThAlMBR | Thermophilic Alternate Membrane Biological Reactor |
| TKN | total kjeldahl nitrogen |
| TN | total nitrogen |
| TP | total phosphorous |
| TSS | total suspended solids |
| UCT | University of Cape Town |
| UWWTD | Urban Wastewater Treatment Directive |
| VFA | volatile fatty acids |
| VIP | the Virginia Plant Initiative |
| WS | waste sludge |
| WW | wastewater |
| WWTP | wastewater treatment plant |

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