BIOLOGICAL PHOSPHORUS REMOVAL: WHERE HAS IT BEEN, WHERE IS IT GOING?

Peter Griffiths
pH Water Consultants Pty Ltd

ABSTRACT

Biological based systems for the removal of phosphorus removal have been implemented within Australia since the early 1980's. Despite significant international and local research, the performance of these types of plants has been, and continues to be, somewhat variable. With the looming "phosphorus crisis" and the nutrient deficient nature of Australian soils, we cannot continue to accept poor performance of these plants that are potentially a significant source of reusable phosphorus compared to chemical phosphorus removal systems. Action is required to provide more effective and lower cost systems. Potential areas of optimisation of current process design procedures are identified.

INTRODUCTION

Australia is faced with somewhat unique municipal sewage characteristics in that influent nitrogen loads per capita are generally higher than those experienced globally in developed countries. Similarly, the per capita phosphorus loads are equal to, if not greater than those for other similar developed countries. Technically, this makes the provision of biological systems achieving the required low effluent nitrogen and phosphorus concentrations more challenging.

Biological based systems for the reduction of effluent phosphorus concentrations have been implemented within Australia since the early 1980's. Despite significant international and local research, the performance of these types of plants has been, and continues to be, somewhat variable.

The impediments to the provision of effective biological nutrient reduction systems include both technical and non-technical factors. The technical factors include such issues as the variety of enhanced biological nutrient removal processes available (more than half a dozen "public domain" processes and many more proprietary systems), plus the impact of the temperature ranges experienced in Australia. In addition, not all aspects of the process design are definitive with "engineering judgement" applied to some components such as anaerobic zones and internal recirculation rates. The outcome of all of these factors is that, despite some very successful plants being implemented within Australia, the full potential of the biological nutrient removal process has not yet been fully realised in terms of maximum cost effectiveness and optimum performance.

In order for Enhanced Biological Phosphorus reduction (EBPR) to be economically competitive compared to alternative chemical phosphorus reduction processes, the increase in treatment volume associated with the provision of additional facilities such as anaerobic zones must be minimised. The intention of this paper is to highlight those processes and features that have proven critical to successful plants. More importantly, the essential design data required to better optimise design and operation to minimise capital and operating costs are identified. Based on this, potential research areas are presented that should provide significant economic benefits to the Australian water industry and, potentially, permit Australia to be a source of expertise for the international water industry.

DISCUSSION

Considerations in Design of Enhanced Biological Phosphorus Removal Plants

There are numerous factors impacting on the cost and performance of an enhanced biological phosphorus reduction facilities (EPBR). These include obvious factors such as temperature and design influent loads. However, once these are established, the key factors become;

- Process configuration
- Sludge Age or Solids Retention Time

The range of process configurations available are briefly reviewed. The factors contributing to the determination of the minimum sludge age are then reviewed in detail to demonstrate the impact on costs and performance. Areas of potential further investigation or research are highlighted to identify where improved performance and/or reduced capital and operating costs would be a benefit.

Process Configuration Optimisation

A broad range of process configurations exist to achieved EBPR. Almost all of these process configurations have, at their core, the Modified Ludzack Ettinger (MLE) process (Figure 1) for substantial nitrogen removal.

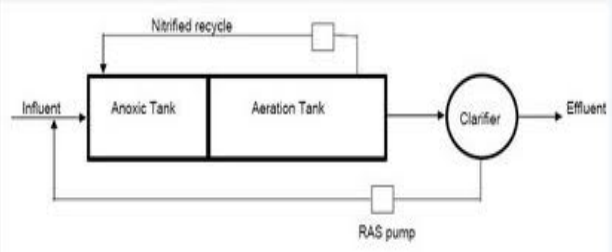

Figure 1: Schematic of MLE process

Utilising this core process, an anaerobic zone was added to achieve EBPR. In its most basic format, this was referred to as the 3 Stage Bardenpho process (also known as the A2O process and Phoredox Process) comprising an anaerobic zone, an anoxic zone and an aerobic zone. This was the "original" EBPR process developed globally by Dr James Barnard.

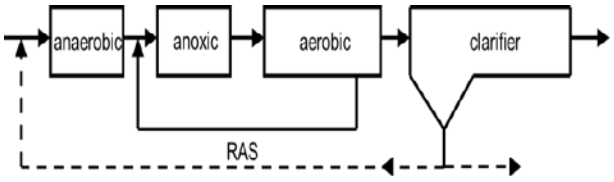

Figure 2: Schematic of 3 Stage Bardenpho process

Initial application of this process was somewhat poor in terms of nitrogen removal and subsequent leakage of nitrate to the anaerobic zone then impacted on the extent of phosphorus reduction achieved.

To better protect the anaerobic zone from nitrate leakage, the basic 3 Stage Bardenpho process was further refined by Dr Barnard. The 5 Stage Bardenpho process was proposed to increase nitrogen removal in order to better protect the anaerobic zone from nitrate leakage. This process configuration involved the addition of a secondary anoxic zone and a small secondary aerobic zone after the main anoxic and aeration zones. The size and cost of the secondary treatment facilities was considerably increased due to the inclusion of these two additional zones.

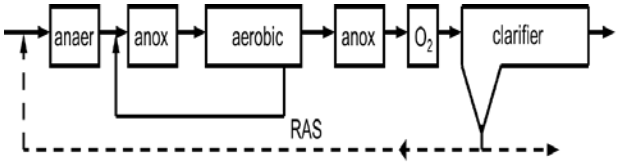

Figure 3: Schematic of 5 Stage Bardenpho process

The 5 Stage Bardenpho process suffered a number of short comings including the need to supply supplemental substrate to the secondary anoxic zone therefore increasing operating costs. The configuration also experienced episodes of secondary phosphorus release in the secondary anoxic zone, negating any benefits from the enhanced biological phosphorus removal.

Alternative process configurations to the initial Bardenpho configurations were also developed. The intention of all of these process configurations developed was to positively protect the anaerobic zone from nitrate leakge.

The UCT process was developed by the University of Cape Town and incorporated an additional pre-anoxic zone and additional internal recirculation (R Recycle) of approximately one times ADWF to reduce or eliminate the discharge of nitrate to the anaerobic zone.

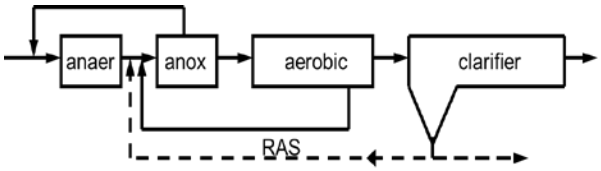

Figure 4: Schematic of UCT process

The UCT process did not, under all circumstances, prevent nitrate leakage to the anaerobic zone. This resulted in the process being modified (Modified UCT process) by dividing the anoxic zone into two sections with the second recycle (R Recycle) being taken from the end of the initial anoxic zone and being returned to the anaerobic zone. The main internal recirculation stream is returned to the second section of the anoxic zone for the bulk denitrification.

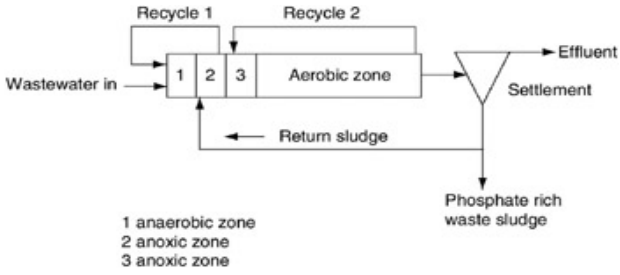

Figure 5: Schematic of Modified UCT process

The Johannesburg process was another alternative process developed, however, at full scale. This configuration required a large anoxic zone to remove all of the residual nitrate from the return activated sludge stream.

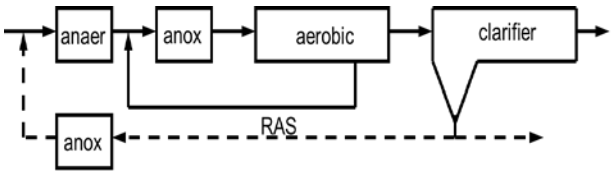

Figure 5: Schematic of Johannesburg process

This process was not completely effective in preventing nitrate leakage and this resulted in the development of the Modified Johannesburg

Configuration. In this configuration, a small stream was provided from the end of the anaerobic zone to the RAS denitrification (anoxic) zone. This stream provided biodegradable substrate from the raw sewage for far more rapid denitrification in the RAS anoxic zone. This drastically reduced the size of the RAS anoxic zone required.

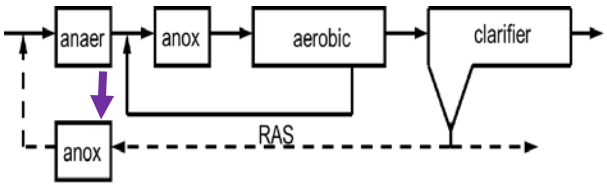

Figure 6: Schematic of Modified Johannesburg process

A further process configuration; the West Bank Configuration was developed by Dr Barnard. This process was similar to the Modified Johannesburg configuration however, rather than providing the small recirculation stream from the end of the anaerobic zone, a small fraction of the raw sewage was bypassed to the RAS anoxic zone.

It should be noted that that are several variations on these processes published in addition to other configurations in the public domain such as step feed processes. However, the purpose of this evaluation is to nominate a technology that can cost effectively achieve low effluent nutrient concentrations whilst being simple and reliable in operation.

Within Australia, many of the initial plants utilised either the Modified UCT configuration (examples include Bendigo 1989, Merrimac 1994, Toowoomba 1996) or Bardenpho based processes usually incorporating pre-fermentation in an attempt to compensate for nitrate leakage to the anaerobic zone (examples include Thorneside, Burpengary East in it is original format). From the processes reviewed, the Modified UCT and the Modified Johannesburg configurations are able to positively exclude nitrate from the anaerobic zone. The Westbank configuration is also able to achieve this objective however, in terms of layout and sizing, it is nearly identical to the Modified Johannesburg configuration. The Modified Johannesburg Configuration is used for comparison as it is simpler in operation and maximises phosphorus removal.

Comparing the MUCT and Modified Johannesburg configurations, it is readily demonstrated that the MUCT process must be larger volumetrically by whatever anaerobic mass fraction is applied. Thus, if a 14% anaerobic mass fraction is used for the biological phosphorus reduction process, given the same operating conditions, a Modified UCT process will have a design volume 14% larger than the equivalent Modified Johannesburg process.

Being a more recent process development, there are fewer examples of the Modified Johannesburg process in Australia. The Morpeth BNR plant in Maitland. NSW, originally designed and commissioned as a Modified UCT type plant, had flexibility to be operated in a number of modes. This reflected the knowledge and surety of the BNR process design at the time (1999). The operating mode was changed from the Modified UCT process configuration to the Modified Johannesburg configuration in 2007 with improved performance (Shaun Clews, personal communication 2008). More recently the Modified Johannesburg process has been successfully employed at the upgraded Burpengary East WWTP achieving very low effluent nitrogen and phosphorus concentrations.

It can be concluded that, of the two public domain processes that are available to protect the anaerobic zone from nitrate leakage and therefore provide effective and efficient EBPR, the Modified Johannesburg configuration offers the more compact layout compared to the alternative Modified UCT configuration. The Modified Johannesburg Configuration can therefore be confidently selected as the lowest capital cost option for future biological phosphorus reduction plants. .

Optimising the Anaerobic Zone

The above evaluation demonstrates that the configuration and sizing of the anaerobic zone is clearly a differentiator in selecting the process configuration that achieves the minimum treatment volume however, how is the optimum size of the anaerobic zone determined?

There has been minimal investigation into the optimum sizing of the anaerobic zone required for optimum phosphorus removal and what research and modelling that has been undertaken could potentially be based on a false premise. Modelling is based on a first order reaction in relation to the concentration of Readily Biodegradable COD (RBCOD) present. However, there are a number of potential models that could fit the available data. Furthermore, the existing models assume phosphorus release in direct proportion to RBCOD taken up whereas early investigations (WRC, 1984) suggest that an initial minimum consumption of RBCOD was required before phosphorus release was observed.

Thus the sizing of the anaerobic zone has been somewhat empirically and experienced based. A number of successfully operating plants have utilised anaerobic mass fractions in the range of 10% to 14% irrespective of the sludge age employed and the operating temperature range experienced. Furthermore, it appears that some plants have been able to achieve a degree of enhanced biological phosphorus removal with

anaerobic mass fractions as low as 6%. The performance of the plants with a 6% anaerobic mass fraction may have been impacted by other factors such as aeration control in the aerobic zone. High performance at these low (or lower) anaerobic mass fractions may therefore be possible.

The impact of reducing the anaerobic mass fraction from say 14% to 6% does not directly equate as an 8% reduction in volume as the implications are far more subtle. It has been calculated that, at 20^0C, a reduction in the anaerobic mass fraction from 0.14 to 0.06 would result in a reduction in the bioreactor volume of 13%. This is clearly a considerable reduction in the bioreactor size required and would represent a considerable capital cost saving. This suggests that further investigation into the optimum sizing of the anaerobic zone in EBPR plants is required.

Further Optimisation of Phosphorus Removal

During the early implementation of EBPR, it was thought that effluent phosphorus concentrations of 1 to 2 mg/L as phosphorus could be achieved. This level of performance was reflected in the establishment of effluent discharge limits designed to encourage the application and demonstration of the biological phosphorus reduction process. Full scale commissioning, optimisation and operating experience has demonstrated that effluent phosphorus concentrations of less than 0.3 mg/L Total Phosphorus can be reliably and consistently achieved without supplement chemical dosing or tertiary filtration. This level of performance has been mandated at a number of locations and the application of biological phosphorus reduction therefore offers a very low operating cost alternative compared to the costs associated with the very high chemical dosing required by purely chemical means (Hamilton and Griffiths 1997).

It has been demonstrated that, to achieve low effluent phosphorus concentrations at reasonable sludge ages, an elevated dissolved oxygen concentration in excess of 2 mg/L is required at the end of aeration (Griffiths and Daigger 2009). This elevated dissolved oxygen concentration permits the extent of phosphorus uptake to be maximised resulting in soluble effluent phosphorus concentrations of less than 0.1 mg/L. The need for elevated dissolved oxygen concentrations towards the end of aeration have been variously reported as early as 1967 and through the early 1970's (Sedlak Ed 1991).

The use of an elevated dissolved concentration at the end of the main aerobic zone can potentially result in considerable dissolved oxygen carryover to the main anoxic zone to the detriment of the denitrification process. To minimise recirculated dissolved to the anoxic zone, a post aerobic zone can be provided after the main aerobic zone and downstream from where the A recycle is taken off as shown in the following figure.

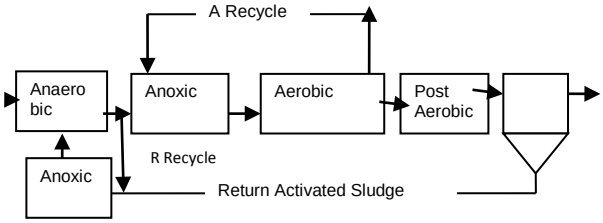

Figure 6: Schematic of Modified Johannesburg process with post aerobic zone.

In this configuration, the post aerobic zone can be operated at high dissolved oxygen concentrations without impacting on the denitrification process. Dissolved oxygen concentrations of 2 to 3 mg/L have been used in the post aerobic zone to achieve very low effluent phosphorus concentrations (Griffiths and Daigger 2009).

The impact of dissolved oxygen concentrations on the extent of biological phosphorus reduction achievable has only limited coverage in the literature (Johnson et al 2006, Griffiths and Daigger 2009) and is not covered at all in the models. It is possible that some of the early 5 Stage Bardenpho systems may have operated with elevated dissolved oxygen concentrations in the secondary aerobic zone thus contributing to improved biological phosphorus reduction.

The application of a post aerobic zone was first implemented in 1996 at the Wetalla WWTP in Toowoomba. A post aerobic mass fraction of approximately 10% was applied and this was duplicated in the 2007 plant augmentation and optimisation. The influent phosphorus concentration is approximately 12 mg/L with a soluble phosphorus concentration in the main aerobic zone of 3 mg/L. After the post aerobic zone, the effluent total phosphorus concentration was reduced to less than 0.3 mg/L (Griffiths and Daigger, 2009).

Clearly, a mass fraction of 10% is adequate for the post aerobic zone when targeting very low effluent phosphorus concentrations. However, it may well be possible that similar, if not identical, performance could be achieved with a smaller post aerobic mass fraction. Reducing the post aerobic mass fraction from 10% to 6%; a 4% reduction; should permit the overall bioreactor volume to be reduced by nearly 7% with considerable capital cost savings.

Additional research and development is required to permit identification of the optimum post aerobic mass fraction. Preferably, revised or refined models would be developed that provide increased understanding of the biochemical processes occurring.

Evaluation of Nitrogen Removal Issues

One would expect that the established nitrogen removal processes of nitrification and denitrification would be well researched and understood permitting accurate design and sizing of at least aspect of the treatment facilities. Unfortunately, this is not the case.

Nitrification Issues

The requirements for nitrification of ammonia within the activated sludge process are well established. The requirements are summarised by the following relationship;

$$f_{aer} * \mu_{A,T} > PF * (b_{A,T} + 1/R_s)$$

where;

f_{aer}	aerobic mass fraction
$\mu_{A,T}$	growth rate of the nitrifiers at T^0C
PF	ammonia load diurnal peaking factor
$b_{A,T}$	death rate of nitrifiers at T^0C
R_s	sludge age

The equation can be literally translated as “growth in the aerobic zone must be greater than death throughout the entire system plus the fraction wasted daily”.

Knowing the growth rate and death rate permits the aerobic mass fraction and sludge age requirements to be determined. Correction factors for the growth rate due to temperature variations, pH and dissolved oxygen concentrations are published in the literature.

The death rates of the autotrophic nitrifying bacteria have long been assumed to be very low compared to those measured for the heterotrophic bacteria that consume the raw sewage organic material. This is attributed to the nitrifying bacteria growing in large, tight “clusters” that are too large and strongly bound to be predated by the protozoan population. A value of 0.03 to 0.04 d^{-1} at 20^0C has generally been adopted (WRC 1984). This is similar to the death rate of the phosphorus accumulating organisms that also grow in large, tight clusters that are resistant to predation (Wentzel et al.,1989).

The growth rates of the autotrophic nitrifying bacteria have been published in the literature. The growth rate measurements identified a range of values between 0.3 and 1.08 d^{-1} at 20^0C (Winkler 1981). The maximum rate measured is also consistent with pure culture studies (Winkler 1981). The broad range of growth rates identified is attributed to the many factors that can impact on the nitrification rates experienced during activated sludge treatment of municipal sewage including the operating dissolved oxygen concentration, the presence of heavy metals or other toxic compounds and pH variations. Modelling of the Merrimac WWTP on the Gold Coast demonstrated that the growth rate of the nitrifying bacteria had to be at least 0.65 d^{-1} at 20^0C assuming a death rate of the nitrifying bacteria of 0.04 d^{-1} at 20^0C. The effluent from the Merrimac WWTP was consistently less than 0.1 mg/L as ammonia nitrogen for daily composite samples.

Recent investigations have suggested that the death predation rate is considerably higher than first thought and that it varies under aerobic, anoxic and anaerobic conditions. Without presenting a detailed review of the literature, the majority of the recent investigations involve either extensive statistical analysis, placing the autotrophs in an abnormal situation (prolonged starvation, high feed, prolonged periods of anoxia or aeration) or calibration without due consideration of potential external factors such as accuracy of aeration control or toxicity impacts. In some of the studies, other potential explanations such as enzyme depletion or ATP exhaustion could have been explored in more detail. Application of these revised kinetic parameters to a number of fully nitrifying plants within Australia suggests that these plants should not be fully nitrifying!!!

Clearly, a more fundamental approach is required to determining nitrification rates that better aligns with the conditions within the activated sludge plant. A number of pilot plants with varying aerobic mass fractions could be operated and the sludge age progressively decreased until ammonia break through occurs. Using simultaneous equations, the growth rate can be determined and then the death predation rate derived for a typical municipal sewage. This would permit more confident and accurate design for nitrification.

Denitrification Issues

In the original University of Cape Town (UCT) activated sludge models, and indeed in the current International Water Association (IWA) models that serve as the basis for most commercial computer simulation packages, the denitrifying bacteria are modelled as a fraction of the total heterotrophic microbial population. The total heterotrophic microbial population is responsible for the consumption and degradation of the influent organic material as measured by the BOD_5 test.

The simplification adopted by the UCT for modelling of the denitrification process was acceptable at the time as it conformed with the results from the pilot plant studies undertaken by UCT (in fact, the fraction of the heterotrophic biomass capable of denitrifying was determined from the pilot plant studies) and reflected the then understanding of the microbiology of activated sludge.

However, the simplification adopted was subsequently demonstrated to be erroneous

(Clayton et al., 1989). The models were unable to match the dramatically changed denitrification rates observed in EBPR plants nor could they adequately model plants with high internal recirculation rates. Other inconsistencies included the constant rate of denitrification observed with varying sludge ages at constant temperatures for a given system and the wide range of rates reported across differing systems at the same temperature

A revised model was proposed (Griffiths, 1994) where the denitrifying heterotrophic bacteria were a separate group of organisms from the rest of the heterotrophic biomass. This revised model was able to adequately address the various anomalies from the original models. The revised model is further supported by more recent identification of "True Denitrifiers" by microbiologists thus supporting the concept that denitrifiers are a separate microbial group within the activated sludge process.

The revised model identifies that there is a minimum anoxic mass fraction for growth of the denitrifying bacteria at any given sludge age and temperature. The required anoxic mass fraction is determined from the relationship;

$$f_{anox}*\mu_{H,T,anox}>PF*(b_H+1/R_s) \quad \text{(eq 1)}$$

This can be literally translated as "growth in the anoxic zone must be greater than death throughout the entire system plus the fraction wasted daily".

The relationship is almost identical in structure as the requirement for nitrification.

$$f_{aer}*\mu_{A,T}>PF*(b_A+1/R_s) \quad \text{(eq 2)}$$

In terms of denitrification, if the anoxic mass fraction is undersized for the given temperature and sludge age, bleed through of nitrate will occur during peak load periods. Thus correct sizing of the anoxic mass fraction is essential not only for EBPR, it is also essential for nitrifying denitrifying plants in general.

The accurate sizing of the anoxic zone requires accurate knowledge of the growth rates and death-predation rates of the denitrifying bacteria. There is minimal data on the growth rate of the denitrifying bacteria in the literature. This is attributable to the broad range of substrates available for denitrification resulting in a broad range of growth rates being experienced. To determine the growth rates on individual compounds would first require knowledge of all of the compounds in the sewage and this would represent a major task in itself.

The dynamic death predation for the denitrifying bacteria is the same as that for the general heterotrophic bacterial population. This value has been reported in the literature as 0.62 d^{-1} at 20^0C. However, this value is in fact a derived value based on the observed overall heterotrophic death predation rate of 0.24 d^{-1} and requires accurate knowledge of such parameters as the heterotrophic yield factor, Y_H, (a nominal constant that actually varies with different substrates), and the particulate endogenous residue fraction, f_{Ep} that is an approximate value only. The published value of the dynamic death predation rate must therefore also be questioned.

Again, it is clear that accurate sizing of the anoxic zone is difficult and requires adoption of a number of kinetic parameters that may be questionable. Again, a more fundamental approach is required to permit better prediction of the anoxic mass fraction required for successful denitrification.

A number of pilot plants with varying anoxic mass fractions could be operated and the sludge age progressively decreased until nitrate break through at the end of the anoxic zone occurs. Using simultaneous equations, the growth rate can be determined and then the death predation rate derived for a typical municipal sewage. This would permit more confident and accurate design for denitrification.

It is important to note that, when investigating the denitrification kinetics and requirements, the pilot plant should, as closely as possible, mimic the operation of full scale plants. A number of geographical differences in design approaches exist and these should be adequately addressed in the pilot plant investigations.

What appears to be a peculiarity within Australia is the use of high internal recirculation (A Recycle) flow rates (>15 times ADWF). This practice has been referred to as an "Australian thing" however, more particularly, this approach predominantly occurs in Queensland and appears to be substantially ignored by designers in more southern states.

The use of high internal recirculation rates is apparently in conflict with published design guidelines that generally recommend that internal recirculation rates should not exceed 6 to 8 times ADWF. However, these guidelines are outdated as demonstrated by model modifications for True Denitrifiers described previously (Griffiths 1994). The Bio P plant at Merrimac on the Gold Coast in Queensland, has been successfully operating since 1995 utilising internal recirculation rates in excess of 20 times ADWF (Griffiths and Daigger,2009). Numerous other plants utilising these high internal recirculation rates have since been provided in Queensland and successfully achieving low effluent phosphorus and nitrogen concentrations.

The high internal recirculation rates are required to achieve low effluent nitrate nitrogen and total

nitrogen concentrations; particularly with the high influent nitrogen loads experienced within Australia. Again, although the high rates go against published guidelines, the guidelines are based on the false premise that only 40% of the heterotrophic biomass can denitrify. The revised microbiological model developed (Griffiths, 1994) based on True Denitrifiers demonstrated that the fraction of denitrifiers is based on the nitrate recirculated to the anoxic zone. Increasing the supply of nitrate increases the fraction of true denitrifiers. This is only logical as pure (100%) denitrifying biomasses are developed on denitrifying filters.

Any pilot plant investigations to determine denitrification kinetics should incorporate a high internal recirculation rate to reflect likely conditions in full scale plants. The use of the high internal recirculation rates in a pilot plant will require the provision of a de-aeration zone to reduce dissolved oxygen carryover to the anoxic zone. The provision of the de-aeration zone reflects current trends for full scale plant design. It should also be noted that excessively high internal recirculation rates should be avoided as these can contribute to significant sludge bulking and interfere with the accurate monitoring and interpretation of pilot plant data.

Features of current successful plants

The review undertaken suggests that most aspects of sizing the various zones for a biological phosphorus removal plant are questionable and require further investigation, revision and refinement. However, despite these uncertainties, there are a number of very successful Biological Phosphorus Reduction plants operating within Australia without any need for any supplemental chemical dosing. How is it that these plants can operate successfully considering the limitations of the design procedures? The simple answer is conservatism and this is best explained using the following example.

A plant is to be designed for a minimum temperature of approximately 20^0C. At this temperature, the required mass fractions for the main aerobic and anoxic zones can be determined as a function of the sludge age using equations 1 and 2 presented earlier. The mass fraction requirements are presented graphically in the following figure assuming a growth rate for the nitrifiers of 0.65 d^{-1} (considered a safe value under Australian conditions).

Optimising for the overall plant design, the minimum aerobic mass fraction required is determined as 0.39 with an anoxic mass fraction of 0.33 at a sludge age of just under 8 days (Figure 7). The remaining 1-0.39-0.33=0.28 mass fraction is used for anaerobic, de-aeration and re-aeration zones (say 0.10, 0.12 and 0.06 respectively).

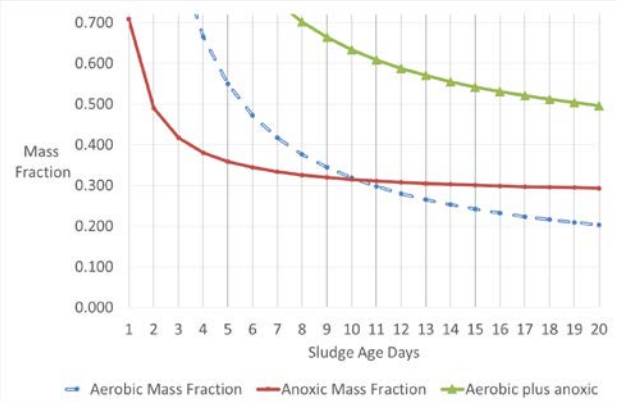

Figure 7: Aerobic and Anoxic Mass fractions (total 0f 0.72) for nitrifier growth rate of 0.65 d^{-1}.

However, if the exercise is repeated with a more conservative growth rate of nitrifiers of 0.45 d^{-1} (as published in WRC 1984), the following graph is derived.

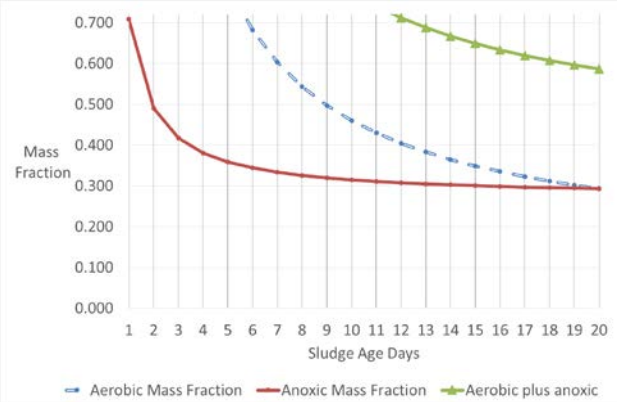

Figure 7: Aerobic and Anoxic Mass fractions (total 0f 0.72) for nitrifier growth rate of 0.45 d^{-1}.

In this example, an aerobic mass fraction of 0.40 and an anoxic mass fraction of 0.32 will be required. This appears a relatively minor change however, inspection of the graph demonstrates that the minimum sludge age required has increased from just under 8 days, to just under 12 days. The volume of the bioreactor required is approximately proportional to the operating sludge age. Effectively, the size of the bioreactor required has been increased by 50% simply by adopting the more conservative growth rate for the nitrifiers.

This situation demonstrates how lack of accurate design information can unnecessarily and significantly add to the size and the cost of the bioreactor. As it is typical to apply a “safety factor” of 125% to the minimum sludge determined (to improve process stability), the additional bioreactor volume required is increased with the design sludge age increased from 10 days to 15 days; an additional 5 days sludge age of bioreactor volume.

Unfortunately, the situation is often worse this with designers adopting a “safe” sludge age; often in the

range of 20 to 25 days resulting in bioreactors nearly twice the actual size required. It is also possible that nitrifier growth rates may be greater than 0.65 d^{-1}, as often suggested in the literature. Should this be the case, shorter sludge ages and smaller bioreactors would be possible and the scale of oversizing through adoption of a 20 to 25 day sludge age would be further increased.

CONCLUSIONS AND SUMMARY

Biological Nutrient removal has been an established technology in Australia for over 25 years and practised globally for considerably longer. Considering the maturity of the technology, many of the kinetic parameters required for accurate design are still questionable if not completely unknown. To address this lack of design information, conservative design approaches are adopted by the industry. This conservatism not only applies to Biological Phosphorus reduction processes, it also applies to the more common and longer established nitrification and denitrification process.

The uncertainty in design is further demonstrated by the lack of a "standardised" process configuration and the frequent provision of "swing" zones and/or flexibility to permit operation of the bioreactor in alternative configurations or flow paths. This flexibility also comes at an increased cost and requires design at very long sludge ages to permit the various operating modes to be employed.

The cost to the industry due to this conservatism is enormous. As an indication, the bioreactor for approximately 20,000 EP would incur direct civil costs of the order of $3.5 million for the bioreactor at a sludge age of 25 days. Reducing the sludge age to 10 days would reduce the direct civil costs to an estimated $2 million. The minimum estimated potential savings of $1.5 million does not include any reductions in associated infrastructure such as roads nor the impact on indirect costs. Savings in excess of $2 million would not be unexpected.

Undertaking pilot plant studies to determine minimum anoxic and aerobic mass fractions, minimum sludge ages and derive the associated kinetic parameters could potentially permit the design and construction of effective and reliable wastewater treatment at significantly reduced capital cost. With Australia's population projected to increase by approximately 200,000 to 250,000 people per year, capital savings of the order of $20 million per year are potentially realisable.

The pilot plant can then also be used to derive the minimum anaerobic and post aerobic mass fractions thus further optimising design and minimising costs for biological phosphorus reduction systems. The post aerobic zone would first be set at a safe value and the anaerobic mass fraction progressively decreased to determine the minimum anaerobic mass fraction acceptable. The post aerobic zone could then be varied with a safe anaerobic zone mass fraction.

It could be argued that any pilot plant results would only be applicable to that particular sewage. However, as the purpose is to determine the kinetics of reaction rather than the extent of reaction, the data derived should be broadly applicable to not only most Australian municipal situations, it should also prove beneficial to many North American, European and other similar location plants.

With the looming phosphorus crisis, the beneficial recovery of phosphorus from sewage in a form where it is bio-available for crops, is considered essential. The biological phosphorus reduction process must therefore form the future basis of wastewater treatment across Australia. Undertaking structured pilot plant investigations should permit lower cost and more effective, high performing biological phosphorus recovery systems to be provided.

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