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Title: **PILOT PLANT STUDY OF ENHANCED BIOLOGICAL REMOVAL OF NITROGEN AND PHOSPHORUS FROM WASTEWATER.**

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1. INTRODUCTION

The nitrogen and phosphorus concentrations in effluents from conventional biological municipal wastewater treatment systems generally greatly exceed the natural concentrations in the receiving waters to which they are discharged. In many cases this does not lead to the environmental impairment of the receiving water, but, where the latter are sensitive to nutrient enrichment, eutrophication may result. Where such is the case, there is a need to reduce effluent nitrogen (N) and phosphorus (P) concentrations to a low level. The pilot plant study reported in this paper evaluated a reactor configuration designed for enhanced biological removal of nitrogen and phosphorus using a suspended floc activated sludge process. The pilot plant was located at Dublin Corporation's Ringsend sewage treatment plant (STP), where its feed wastewater was clarified effluent from the STP's primary clarifiers.

2. WASTEWATER N and P

The extent of removal of N&P by conventional wastewater treatment processes is dependent on wastewater composition, in particular the relative proportions of contained biodegradable carbon, N and P. The biodegradable organic carbon content of wastewaters is conventionally quantified in terms of parameters such as 5-day biochemical demand (BOD_5), chemical oxygen demand (COD) and in units of person equivalent (PE). In this presentation the PE unit is taken as a BOD_5 production of 0.06 kg/d. The nitrogen content of wastewater is usually expressed as total Kjeldahl nitrogen (TKN), which includes organic nitrogen and ammonia, but not nitrate. The phosphorus content of wastewater is expressed as total phosphorus (TP), which includes all phosphorus moieties.

The BOD_5:TKN:TP ratio in municipal wastewaters varies from municipality to municipality, being influenced by a number of factors, including the industrial wastewater component which may dilute or enrich the domestic wastewater fraction. In domestic wastewater, human wastes account for about 50% of the BOD_5, about 85% of the TKN and about 50% of the TP; food residues, laundry and bathroom discharges account for the remainder. Irish municipal wastewater's typically carry a TKN load in the range 10-14 g/PE.d and a TP load in the range 2-3 g/PE.d. Thus, a typical average Irish municipal wastewater has a BOD_5:TKN:TP ratio of 100:20:4.

The limiting nutrient requirements for microbial growth are generally considered to be satisfied (Pipes, 1979) by the following nutrient threshold ratios:

	BOD_5	**:**	**N**	**:**	**P**
aerobic processes	100	:	5	:	1
anaerobic processes	100	:	0.5	:	0.1

It is therefore clear that N and P concentrations in a typical Irish municipal wastewater exceed the basic microbial growth requirements of conventional aerobic biological treatment processes by a factor of about 4, resulting in typical removals in the range 20%-30% by such processes.

3. ENHANCED BIOLOGICAL REMOVAL OF N

The enhanced biological removal of nitrogen from wastewaters is carried out in two distinct process stages. The first stage is the process of **nitrification** or the conversion of ammonia to nitrate and the second stage is **denitrification** or the reduction of nitrate to gaseous nitrogen endproducts.

3.1 **Nitrification**

Microbial nitrification is a two-step process, the first step being the conversion of ammonia to nitrite, which is accomplished by Nitrosomonas bacteria, while the second step involves the conversion of nitrite to nitrate by Nitrobacter bacteria:

The overall chemical oxidation reaction is:

$$NH_4^+ + 2\,O_2 \rightarrow NO_3^- + 2H^+ + H_2O \qquad (1)$$

Taking account of the incorporation of nutrients in the process of cell synthesis, using yields of 0.08 g VSS/g NH_4^+-N and 0.05 g VSS/g NO_2^--N, for Nitrosomonas and Nitrobacter, respectively (USEPA, 1993), the overall reaction describing the complete nitrification process becomes:

$$1.00\,NH_4^+ + 1.89O_2 + 0.0805\,CO_2 \rightarrow 0.0161\,C_5H_7O_2N + 0.952\,H_2O + 0.984\,NO_3 + 1.98\,H^+ \qquad (2)$$

where $C_5H_7NO_2$ is taken as representing the bacterial cell composition. Thus the conversion of 1mg of ammonium-nitrogen is estimated to result in the consumption of 4.32 mg oxygen, the production of 0.13 mg of nitrifying organisms and the destruction of 7.07 mg of alkalinity (as $CaCO_3$).

Nitrification kinetics

The nitrifying bacteria are chemoautotrophs; their growth energy is derived from the oxidation of inorganic N and their carbon source is carbon dioxide.

The growth rate of nitrifiers is estimated to be some 10 to 20 times slower than the growth rate of the heterotrophs which are responsible for carbonaceous BOD removal. Of the two species responsible for nitrification, Nitrobacter has a higher growth rate than Nitrosomonas. The growth rate of the latter, which is responsible for the conversion of ammonia to nitrite, is thus normally rate-limiting for the nitrification process. It also follows from this that nitrite is not usually found in high concentrations in nitrifying processes operating under steady state conditions. The growth of Nitrosomonas can be expressed according to the Monod growth model as follows:

$$\mu_N = \hat{\mu}_N \frac{N}{K_N + N} \qquad (3)$$

where:

μ_N = specific growth rate of Nitrosomonas (d^{-1})
$\hat{\mu}_N$ = maximum specific growth of Nitrosomonas (d^{-1})
K_N = half-saturation coefficient for Nitrosomonas (mg l^{-1} NH_4^+-N)
N = NH_4^+-N concentration (mg l^{-1})

For design purposes, the value of K_N may be taken as 1 mg NH_4^+-N l^{-1}, while the value of the maximum specific growth rate constant is dependent on temperature and may be represented by the following empirical Arrhenius-type expression (USEPA, 1993):

$$\hat{\mu}_N = 0.47\,\theta^{(T-15)} \qquad (4)$$

where θ is generally taken to have a value of about 1.1.

Because of the relatively low value of K_N, the nitrification process proceeds, under typical operating wastewater treatment conditions, at the maximum growth rate for the Nitrosomonas bacteria, i.e. it is a zero order process, independent of the ammonia concentration. If, however, the ammonia nitrogen concentration drops down towards the half saturation concentration level of 1 mg l^{-1}, then the process becomes rate-limited by the reduced concentration according to equation 3.

According to the Monod model of microbial growth, the effluent substrate (in this instance NH_4^+-N) in a completely suspended floc mixed reactor, operating at steady state, can be expressed in terms of the sludge age θ_s and the Monod kinetic parameters, as follows:

$$N = \frac{K_N(1 + k_d\theta_s)}{\theta_s(\hat{\mu}_N - k_d) - 1} \qquad (5)$$

This relationship is plotted in Fig 1 for temperatures of 10 and 20 ^oC, illustrating the marked influence of temperature on nitrification performance. For example, Fig 1 indicates that a sludge age of about 10 days is necessary to reduce the reactor ammonia nitrogen concentration to 1 mg l^{-1} at an operating temperature of 10 ^oC, while at a temperature of 20 ^oC the ammonia nitrogen concentration is reduced to 0.5 mg l^{-1} at an operating sludge age of about 5 days.

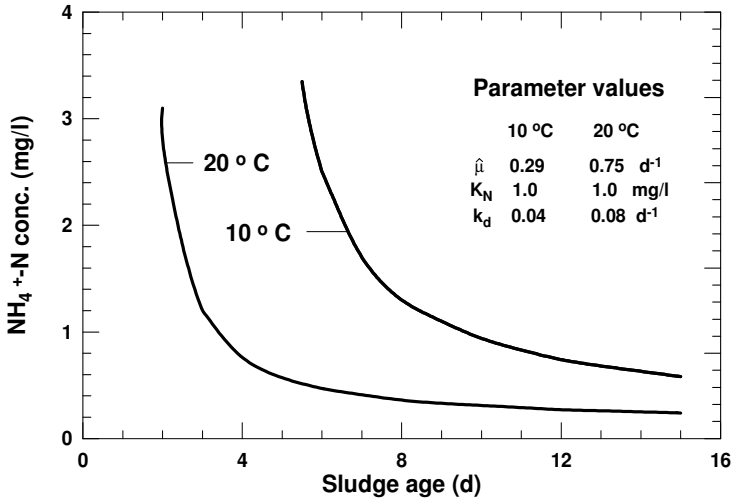

Fig 1 **Steady state ammonia nitrogen concentration in a completely mixed nitrification reactor**

It has been observed (Stenstrom & Song, 1991) that the **dissolved oxygen** concentration (DO) has a significant influence on the nitrification process. It would appear that the growth rate of Nitrosomonas may be slowed down at DO concentrations less than 1 mg l^{-1}. The achievement of this limit value throughout the mixed liquor in a suspended floc activated sludge process, however, may require an operating DO level of at least 2 mg l^{-1}, depending on the intensity of mixing and the associated spatial DO gradients in the microbial floc. A still higher DO concentration (> 3 mg/l) is required in attached biofilm processes to ensure penetration of dissolved oxygen into the biofilm.

As noted above, the nitrification process exerts a substantial alkalinity demand (7.1 mg alkalinity as $CaCO_3$/mg NH_4^+-N). This inevitably reduces pH, particularly where there is an insufficient buffering capacity in the wastewater being treated. The optimum pH range for nitrification would appear to be 7.0 - 8.5. For design purposes, this range may be extended to 6.5 - 9.0 (USEPA, 1993).

Nitrifying organisms are susceptible to inhibition by many organic substances such as solvents and inorganic substances such as heavy metals (Hockenbury & Grady, 1977; Benmoussa et al., 1986). It has also been reported (Gujer, 1977) that the recycle of anaerobic digester supernatant may have an inhibitory effect on Nitrosomonas growth rate.

3.2 **Denitrification**

The microbial reduction of nitrate is brought about by a variety of oxygen-utilising heterotrophic bacteria which, in the absence of oxygen, are capable of using nitrate in place of oxygen as a terminal electron acceptor. Research has shown that anoxic respiration of this kind also takes place at low DO concentration; the DO concentration at which denitrification stops has been reported to be 0.2 mg l^{-1} in pure cultures (Focht & Chang, 1975) and in activated sludge systems to be in the range 0.3 - 1.5 mg l^{-1} (Burdick et al., 1982). In the denitrification process, nitrate and its reduced forms act as electron acceptors, resulting in a stage-wise reduction of nitrate to gaseous nitrogen:

Redox state of N:	+5	+3	+2	+1	0
	NO_3^-	NO_2^-	NO	N_2O	N_2
	Nitrate	nitrite	nitric oxide*	nitrous oxide*	nitrogen*

* gaseous end products

A variety of heterotrophic bacteria, commonly present in activated sludges, participate in the denitrification process including Alcaligenes, Achromobacter, Micrococcus and Pseudomonas. These organisms have the remarkable metabolic capability of using either oxygen or nitrate as electron acceptor in their energy generation process. If oxygen is present, it is preferentially used over nitrate.

As heterotrophs, these bacteria use organic matter as their carbon source and hence remove BOD in conjunction with denitrification. The overall stoichiometry, neglecting cell synthesis, may be approximated as follows:

$$2NO_3^- + 2H^+ + 2.5C \rightarrow N_2 + 2.5CO_2 + H_2O \qquad (6)$$

Based on electron acceptor capacity, 1 g of nitrate nitrogen is equivalent to 2.86 g of oxygen. Thus, the combination of nitrification and denitrification processes can significantly reduce the overall oxygen demand relative to nitrification on its own.

The organic carbon requirement for the denitrification process, expressed in BOD terms, corresponds to a BOD_5/NO_3^--N ratio ≥ 3. Thus, a typical fully nitrified effluent will not have sufficient residual biodegradable carbon to act as carbon source for the denitrification process. In practice, either the wastewater influent or a supplemental source such as methanol, is used to provide the necessary organic carbon for the denitrification process.

The denitrification process increases the bicarbonate alkalinity, theoretically creating 3.57 mg alkalinity as $CaCO_3$ per mg of nitrate nitrogen reduced to nitrogen gas. This recovery of alkalinity partially reverses the drop in alkalinity and pH associated with the preceding nitrification process. This compensatory effect can be a significant process design consideration for wastewaters that are low in alkalinity and may be sufficient to prevent an inhibitory drop in pH in the nitrification step.

Denitrification process kinetics

The rate of growth of denitrifying organisms can be expressed in a Monod-type expression, using nitrate as the rate-limiting nutrient:

$$\mu_D = \hat{\mu}_D \frac{D}{K_D + D} \qquad (7)$$

where:

μ_D = specific growth rate (d^{-1})
$\hat{\mu}_D$ = maximum specific denitrfier growth rate
D = concentration of nitrate nitrogen (mg l^{-1})
K_D = half saturation coefficient (mg l^{-1})

As the value of K_D is very low (0.3 - 0.6 mg/l NO_3^--N, refer Table 11.3), the process is effectively zero order with respect to nitrate concentration.

The other material environmental influences are the concentrations of biodegradable carbon and of DO, the latter having an inhibiting effect, as discussed above. These influences can be combined with equation 7 to give the following overall expression for denitrifier growth rate:

$$\mu_D = \hat{\mu}_D \left(\frac{D}{K_D + D} \right) \left(\frac{S}{K_S + S} \right) \left(\frac{K_o}{K_o + S_o} \right) \qquad (8)$$

where S and K_s refer to the biodegradable organic substrate and S_o and K_o refer to DO. The Ks value may be taken as that which applies to heterotrophic growth under aerobic conditions.

The term $K_O/(K_O+S_O)$ acts as a switching function, turning the denitrification process on and off. A value of K_O of 0.1 mg l^{-1} has been suggested (IAWPRC model, 1994) for denitrification, implying a halving of the growth rate at a DO concentration of 0.1 mg l^{-1} relative to its value at zero DO.

As with all biological processes, denitrification is significantly influenced by temperature. The magnitude of this influence can be expressed by an Arrenhius-type function of the form:

$$\mu = \mu_{20}\, \theta^{(T-20)} \qquad (9)$$

where values of θ have been reported (USEPA, 1993) in the range 1.02 - 1.08 . At a θ value of 1.05, the mean of this range, the growth rate at 10 ^oC is calculated to be 61% of its rate at 20 ^oC.

In general, the denitrification process is much less sensitive to inhibitory substances than is the nitrification process. Experimental findings (USEPA, 1993) indicate that denitrification rates may be depressed below pH 6.0 and above pH 8.0.

Design considerations

It is clear from the foregoing discussion that biological nitrification and denitrification processes have conflicting environmental requirements for optimal operation. Nitrification requires a highly aerobic environment with a sufficiently long microbial residence time to allow the development of a sufficiently high concentration of the slow-growing nitrifying bacteria, Nitrosomonas and Nitrobacter. These conditions result in a very low biodegradable carbon substrate level in nitrifying reactors. Denitrification, on the other hand, requires an anoxic environment and the availability of an ample biodegradable carbon substrate concentration (BOD_5**:** NO_3^--N ≥ 3).

The nitrification and denitrification processes can be efficiently combined to achieve enhanced nitrogen removal using the reactor layout shown in Fig 2. The denitrification (anoxic) and nitrification (aerobic) reactors are operated in series, with the former in the upstream position. While this reactor configuration provides the required environmental conditions in each reactor, the nitrate formed in the aerobic reactor has to be delivered to the anoxic reactor by a large recycle flow.

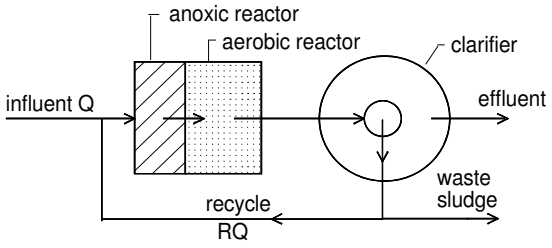

Fig 2 **Single anoxic/aerobic reactor pair**

By arranging a number of reactor pairs in series, as illustrated in Fig 3, an increased denitrification efficiency can be achieved (Miyaji et al., 1980; Schlegel, 1987). The potential denitrification efficiency of such multiple reactor pairs is given by the expression:

$$\eta = 1 - \frac{1}{n(1+R)} \qquad (10)$$

where η is the ratio of nitrate removed to nitrate produced, n is the number of reactor pairs and R is the recycle ratio i.e. recycle flow/inflow. The influences of n and R on denitrification efficiency are illustrated in Fig 4. For example, a set of three reactor pairs, as illustrated in Fig 3, operating without recycle, has the same denitrification efficiency as a single reactor pair with a recycle ratio of ca. 2.

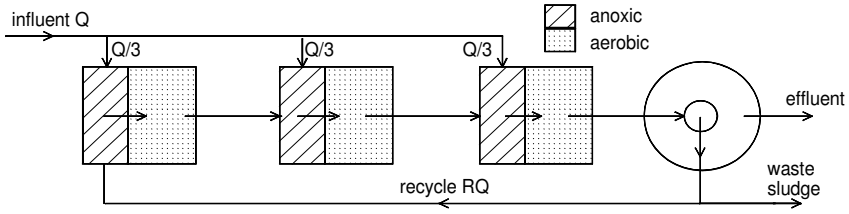

Fig 3 Illustration of three anoxic/aerobic reactor pairs in series

As discussed in an earlier section, denitrification is inhibited in the presence of dissolved oxygen. Hence, it is desirable that the recycle flow does not carry a high DO concentration into the anoxic reactor. For this reason and also in the interest of energy conservation the aerobic reactors should be operated at the minimum DO level that provides efficient nitrification.

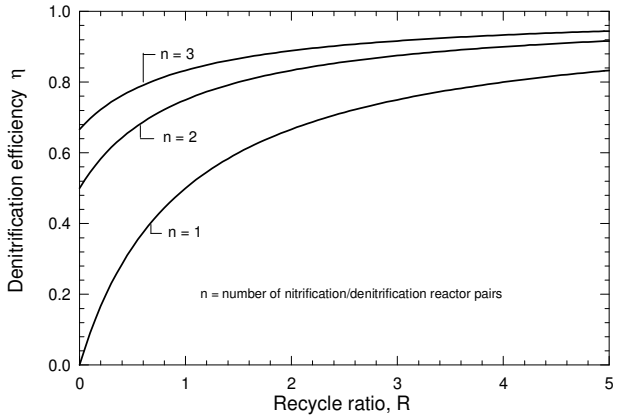

Fig 4 Denitrification efficiency of multiple reactor pairs

4. ENHANCED BIOLOGICAL REMOVAL OF P

The activated sludge process can be manipulated to enhance phosphorus removal through the creation of environmental conditions that produce the so-called ‘luxury’ uptake of phosphorus, i.e. an uptake in excess of the normal metabolic fixation of P by bacterial cells. This enhanced uptake has been found to result in an increase in the P-content of the biomass from a typical 1-2% on a dry weight basis in a conventional activated sludge process to the region of 3-6% in a biological phosphorus removal process (Sedlak, 1991).

The enhanced uptake of P is achieved by subjecting the mixed liquor to an anaerobic/aerobic cycle. In the anaerobic phase there is an uptake of fermentation products (volatile fatty acids (VFAs) such as acetic acid and propionic acid), which accumulate as storage products within the microbial cells. This uptake of VFAs is accompanied by a corresponding release of P into solution. In the following aerobic stage the stored products are oxidised, resulting in a simultaneous enhanced uptake of P which is stored as polyphosphate within the cells.

The enhanced removal of P is considered to be due to a specific genus of bacteria, *Acetinobacter calcoaceticus.* The amount of phosphorus incorporated into the microbial biomass would appear to be not greatly influenced by temperature in the range 5-20^{o}C, with some research evidence to indicate a higher incorporation at lower temperatures (Sedlak, 1991).

The two main factors that are known to influence biological P-removal are the VFA:P ratio in the anaerobic reactor and the sludge age in the aerobic reactor. The VFA availability is dependent on the prior fermentation conditions and carbon substrate availability. Fukase at al. (1982) found that the BOD:P removal ratio increased from 19 to 26 as the sludge age increased from 4.3 to 8 days. At the same time the P-content of the activated sludge decreased from 5.4% to 3.7%. In practice, it may be difficult to achieve an effluent TP level in the range 1-2 mg/l, where the BOD_5:P ratio is less than 20.

The anaerobic reactor is usually sized to provide an hydraulic residence time of 1-2h, based on process influent flow (Sedlak, 1991). The shorter residence time is adequate for septic wastewaters having a relatively high BOD:P ratio, while the longer residence time may be necessary to allow some breakdown of particulate BOD in wastewaters with a low soluble BOD content. As far as possible nitrate should be excluded from the anaerobic reactor, as its presence may inhibit the fermentation process that produces VFAs.

The aerobic reactor is designed as a conventional activated sludge process. Where enhanced nitrogen is also required, as is the case in the present context, it is important that as little nitrate as possible is carried over into the anaerobic reactor for the reasons already outlined. As noted above, the incorporated P-content of the activated sludge decreases with increasing sludge age as does the process sludge yield. Hence, the required BOD:P ratio for a given residual P concentration increases with the aerobic reactor sludge age.

In addition to the foregoing process design considerations, two operational considerations, in particular, must be taken into account to achieve a low residual effluent P concentartion (a) the effluent suspended solids must be maintained at a low level and (b) the excess sludge must be maintained in an aerobic state to avoid P loss.

5. **PILOT PLANT SETUP**

The pilot plant comprised an anaerobic inlet chamber in series with a set of three aerobic/anoxic reactor pairs and a final clarifier. These component process zones were fitted into a single rectangular segmental GRP tank of nominal plan dimensions 2m x 1.5m by 2m high, as shown in Fig 5. The overall biological process volume was 4.4 m^3, subdivided into aerobic, anoxic and anaerobic zones, the aerobic:anoxic:anaerobic volume ratios being 6:3:1. Each aerobic chamber was fitted with an internal vertical baffle wall to prevent short-circuiting and create a 'round-the-end' flow path. The anaerobic and anoxic zones were mixed by a downdraught propellor mixers discharging through draught tubes with a common overhead propellor drive system. The aerobic zones were aerated by a fine bubble diffused air system using tubular membrane diffusers. The air was supplied by a variable speed oil-less vane-type air blower. Initially an air-displacement sludge recycle system was used but this was subsequently replaced by a small pump fitted with a trimmed impeller to reduce its output and improve its solids transmission characteristics.

As shown in Fig 5, the inflow and sludge return streams were mixed in the anaerobic chamber, whence the flow was split into three equal streams discharging into the three anoxic chambers. The discharge from each anoxic zone to its paired aerobic zone was located at floor level. In addition to the external sludge recycle, the airlift effect of the aeration system was used to generate an additional internal recycle in each of the three aerobic/anoxic reactor pairs. This was facilitated by lowering the division

wall between the anoxic and aerobic chambers to create a weir overflow as shown in Fig 5. This had the effect of creating a significantly increased recycle without using any additional pumping equipment.

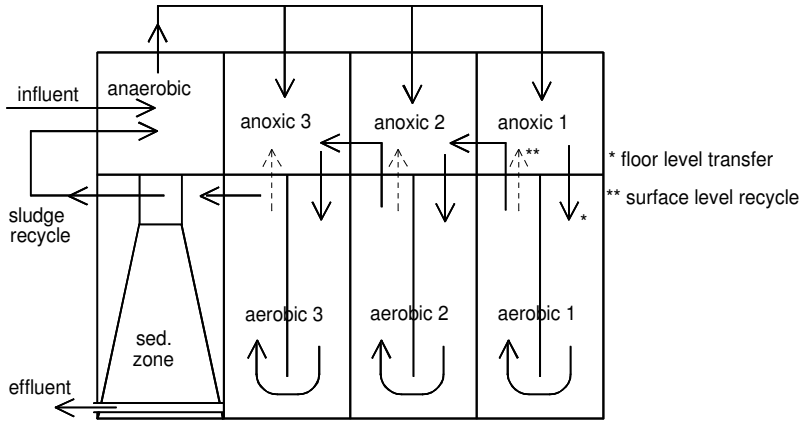

Fig 5 Schematic plan view of pilot plant tank

6. PROCESS CONTROL AND ON-LINE MONITORING

A PC-based process monitoring and control system was developed for the pilot plant study, using software written in the LabVIEW graphical programming language and using data acquisition cards supplied by National Instruments.

The following process variables were automatically controlled from the PC:

- process DO through speed control of the air blower
- mixer speed in the anoxic and anaerobic zones
- sludge age through regulating mixed liquor wastage

On-line monitoring was also provided for the following parameters:

- temperature
- pH
- ORP
- mixed liquor suspended solids

Rather than install monitoring probes in each chamber of the pilot plant, it was decided instead to use a common set of probes for the plant as a whole. This set comprised a dissolved oxygen (DO) probe, a redox (ORP) probe, a temperature probe and a pH probe. The probes were located in a flow-through cell located in a site hut adjacent to the pilot plant. An automated pump-based sampling system sampled each chamber in turn at a frequency set by the operator. Prior to each sampling event, the sample lines were flushed for a short period. The computer monitor displayed current values in graphical form and also allowed the operator to view the already recorded stored data from all the on-line probes.

Ammonia and nitrate were also measured by probes but not on-line as the probes used were not suited to on-line application.

Phosphorus concentrations were measured by conventional chemical analysis.

It was found necessary to devote a considerable effort to the calibration and maintenance of the monitoring probes during the course of the test period.

7. PROCESS PERFORMANCE

7.1 Commissioning

Following its construction in the workshop of the Department of Civil Engineering, University College Dublin the pilot plant and its associated equipment were transported to Dublin Corporation's Ringsend treatment works where the system was located adjacent to the outflow from the plant's primary clarifiers. A site hut which housed a temporary laboratory, the computer control system and the sampling system, was set up adjacent to the pilot plant. The wastewater feed to the pilot plant was delivered by a submersible pump through a constant head tank, which was designed to regulate the inflow rate.

The plant was seeded with a non-nitrifying activated sludge from the Swords wastewater treatment plant. A pure culture of nitrifying organisms (Nitrosomonas) was added to accelerate the development of nitrification. Initially the inflow rate to the plant during the commissioning phase was set at 3.6 l/min corresponding to a mean overall biological reactor residence time of about 20 hours.

While process commissioning was in progress the control system development and testing was proceeding in parallel. The initial design of sludge-recycle, which was based on a batch-wise displacement of settled sludge from the sedimentation zone into the anaerobic zone, was replaced by a continuously pumped system; the intermittent batch-displacement was found to have an adverse effect on the stability of the sludge blanket in the clarifier.

A further operational problem encountered during the commissioning phase was the development of a biological foam on the surface of the aerated zones, which was unsightly and adversely affected control and operation. The precise causes of biological foaming, which results in the formation of a floating layer of activated sludge of mousse-like consistency, containing entrapped air in a mainly filamentous floc structure, are not well understood (Foot, 1992, Tipping, 1995), but would appear to be associated mostly with low loading conditions. The foam problem was eventually brought under control by a combination of hypochlorite spray and physical removal of the foam layer.

7.2 Sampling and analysis

The pilot plant influent and effluent streams were sampled at hourly intervals by automatic samplers, generating daily composite samples for both.

TKN, nitrate, mixed liquor suspended solids and temperature measurements were carried out using probe methods with frequent checking against conventional manual procedures. The measurements of the remaining parameters monitored, including BOD_5, COD, TOC, suspended solids and TP, were made by conventional laboratory procedures, in accordance with Standard Methods for the Examination of Water and Wastewater (APHA, 1989).

7.3 Influent wastewater characteristics

The pilot plant was in operation for a 10-month period; during the first 6 months it was operated in a developmental mode, hence the results presented here are confined to the final 4 months of operation. The pilot plant influent characteristics (outflow from main plant primary clarifiers) are illustrated on Fig 6, which plots the variation in BOD, TKN and TP parameters over the test period. The corresponding mean and range values for these parameters are presented in Table 1.
These values are based on the analysis of daily composite samples.

The estimated average quality parameter values for the Ringsend raw wastewater over the test period indicate a BOD_5:TKN:TP ratio 100:14.3:3.2, which, when compared with the typical domestic wastewater ratios for these parameters, discussed earlier, would indicate a significant carbon enrichment relative to a typical domestic wastewater. The mean quality parameter values also place the wastewater in the relatively strong municipal wastewater category.

Table 1
Pilot plant influent characteristics
(effluent from main plant primary clarifiers)

Parameter	Mean	Range
BOD_5 (mg/l)	242 (403)*	77 - 341
TKN (mg/l)	31.6 (37.2)	12.4 - 43.8
TP (mg/l)	11.7 (13.0)	2.5 - 18.2

* estimated raw wastewater values based on the assumption that primary clarification removed 40% of the BOD_5, 15% of the TKN and 10% of the TP.

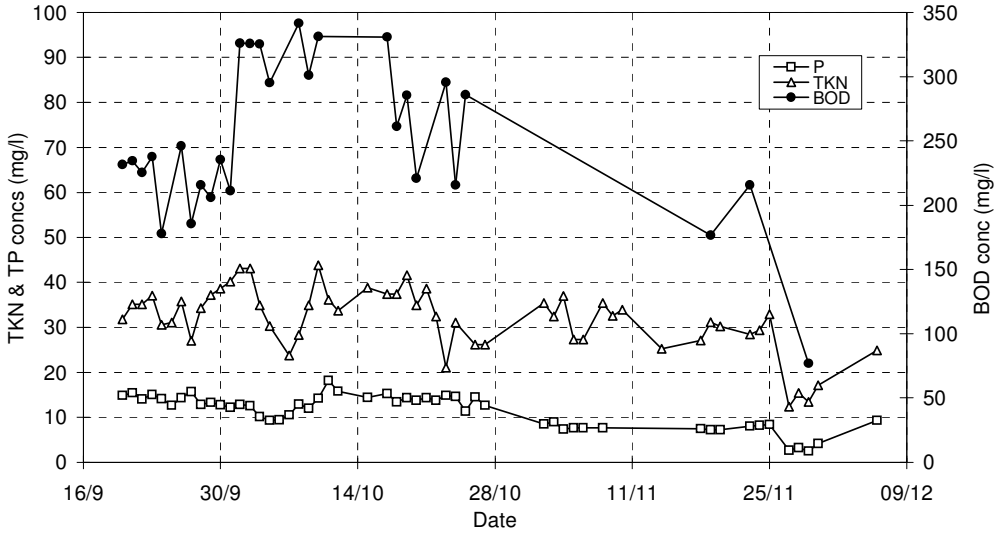

Fig 6 **Pilot plant influent characteristics**

7.4 Pilot Plant Results

The results presented here relate to the final 4-month operating period, September to December 1995, during which the influent flow rate was set at 4.8 l/min, corresponding to a mean overall hydraulic retention time in the biological process zones of 15.3 hours. During this period the MLSS was maintained at a constant concentration of 3000 mg/l.

The performance over this period is summarised in graphical format on Figs 7, 8, 9 , which present results for BOD_5, N & P, respectively. A corresponding overall summary of the plant performance is given in Table 2.

As discussed earlier, enhanced biological P-removal is dependent on the availability of a readily biodegradable substrate in the anaerobic fermentor. At the end of the test period the influent biodegradable organic carbon was artificially increased for a short period by the addition of 10 mg/l of acetic acid. The resulting effect on the effluent TP concentration is plotted in Fig 10.

The anaerobic environment, essential to P-release and VFA uptake in the anaerobic chamber is characterised by a negative ORP value. The measured ORP variation in the anaerobic zone over the test period is plotted in Fig 11.

The variation in mixed liquor temperature over the test period is plotted in Fig 12.

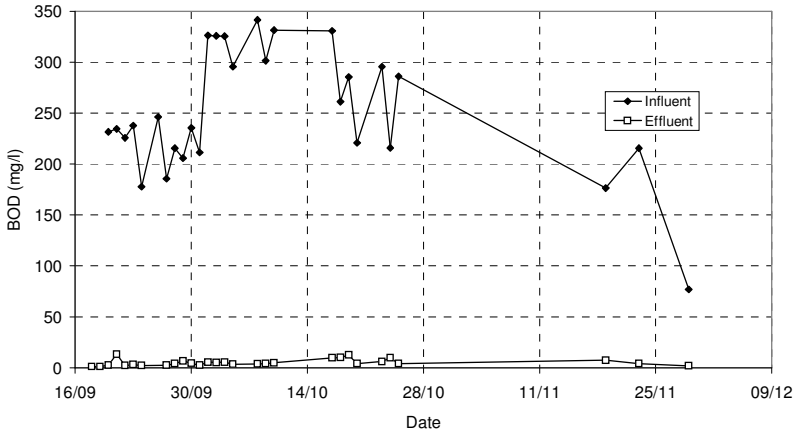

Fig 7 **Pilot plant BOD removal performance**

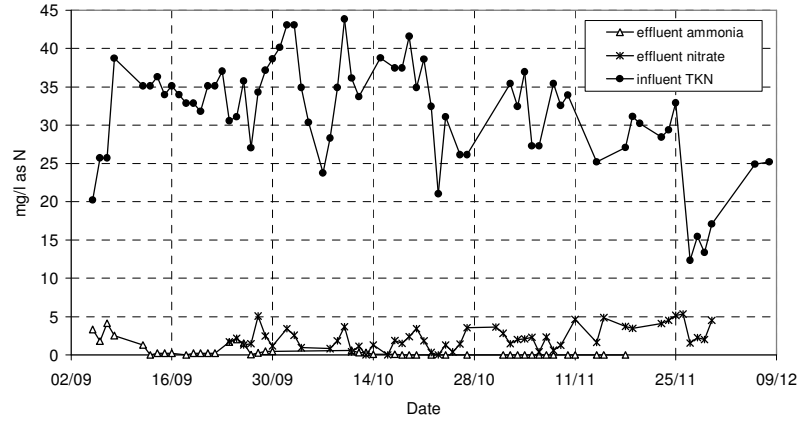

Fig 8 **Pilot plant N-removal performance**

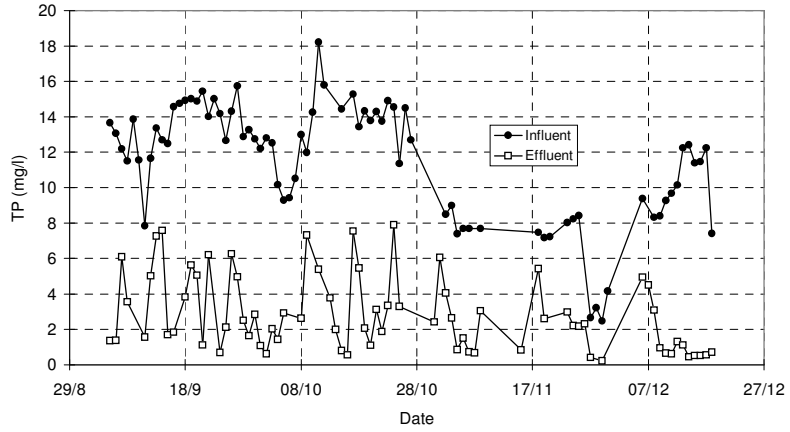

Fig 9 **Pilot plant TP-removal performance**

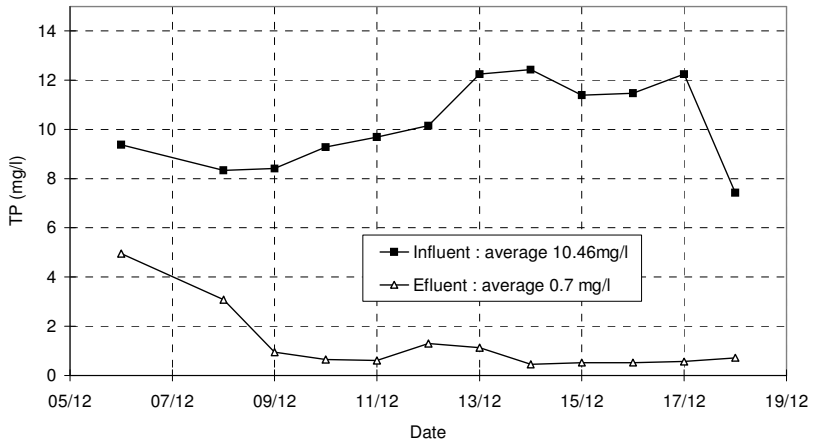

Fig 10 **Influence of added VFA on P-removal performance**
(10 mg/l acetic acid added from 7/12 onwards)

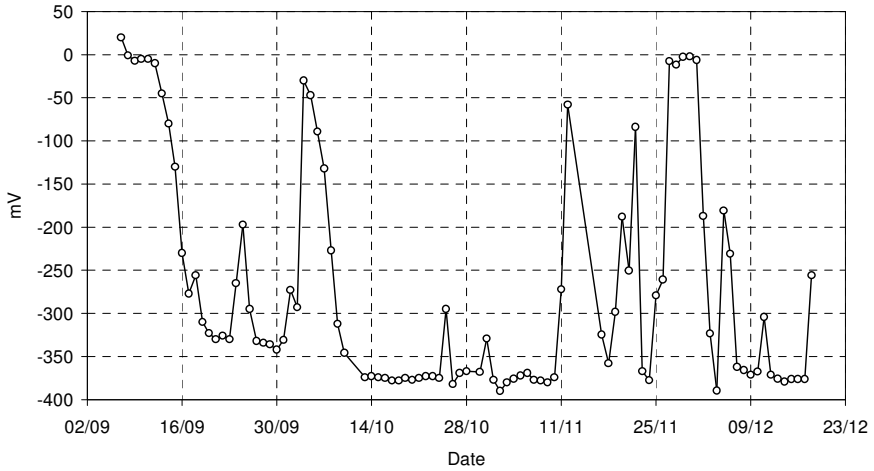

Fig 11 **Measured ORP in the anaerobic zone**

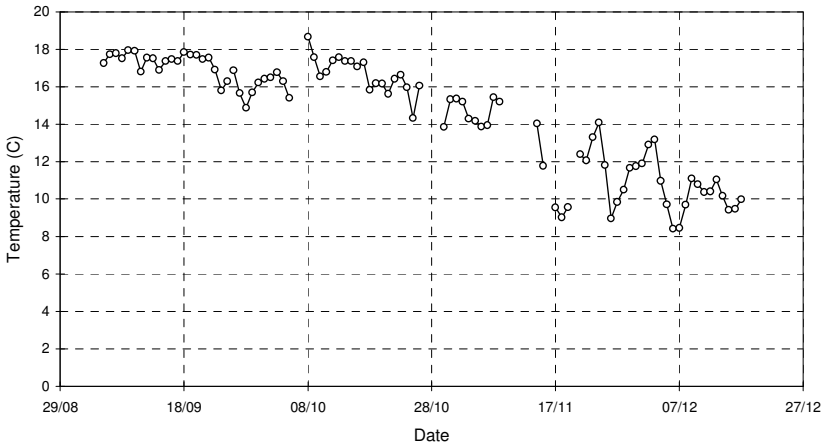

Fig 12 **Mixed liquor temperature**

Table 2
Pilot plant performance summary

Parameter	In/out	Mean	St. dev.	Max.	Min.
BOD_5	Influent (mg/l)	242	68	341	77
	Effluent (mg/l)	5.2	3.2	13.4	1.3
	% removed	97.8			
TOC	Influent (mg/l)	77.3	28.4	163.9	34.5
	Effluent (mg/l)	14.3	7.4	51.4	6.3
	% removed	81.5			
N	Influent TKN (mg/l)	31.7	6.8	43.8	12.4
	Effluent NH_3 (mg/l)	0.5	0.9	4.1	0
	Effluent nitrate-N (mg/l)	2.2	1.5	5.3	0
P	Influent TP (mg/l)	11.7	3.4	7.9	0.2
	Effluent TP (mg/l)	3.1	2.1	7.9	0.2
	% removal	75			

7.5 **Discussion of results**

(a) **Process loading**

The mean sludge loading rates and hydraulic retention times in the three bioreactor zones during the test period, based on the influent wastewater flow, were:

	Sludge loading rate	Hydraulic retention time
	(kg BOD_5/kg.MLSS.d)	(h)
Aerobic zone	0.21	9.2
Anoxic zone	0.42	4.6
Anaerobic zone	1.26	1.5

The foregoing results show that the mean pilot plant organic loading rate was about twice that typically applied in the design of nitrifying activated sludge processes treating normal municipal wastewaters. As may be seen from Fig 12 the mixed liquor temperature dropped from about 18 ^oC at the start of the test period to about 10 ^oC at the end of the period.

(b) **BOD removal**

As is to be expected from an activated sludge process operating at a loading close to the upper bound of what is conventionally designated the extended aeration loading range, the pilot plant achieved the very low mean effluent BOD_5 concentration of 5.2 mg/l, despite the substantial variation in the influent BOD_5 concentration, as reflected by the standard deviation of 68% for the set of mean daily values.

As noted in section 7.3, the mean influent BOD_5 concentration was quite high for a clarified municipal wastewater, suggesting a significant industrial component and very little dilution by rainwater.

(c) **Nitrogen removal**

As the effluent nitrogen data plotted on Fig 5 show, the pilot plant achieved, on average, over 90% nitrogen removal, with a mean effluent ammonia concentration of 0.5 mg N/l and a mean effluent nitrate concentration of 2.2 mg N/l.

These results represents a very good nitrification performance, particularly in view of (i) the relatively high prevailing average sludge loading rate of 0.21 kg BOD_5/kg MLSS and (ii) the decrease in mixed liquor temperature from 18 ^oC to 10 ^oC over the test period. Although there was no evidence of a reduction in nitrification performance in response to the reducing temperature over the test period, a more extended test run at the lower temperature would be necessary to confirm the influence of temperature on nitrification rate. It seems clear that the automated DO control played a significant role in sustaining the very high degree of nitrification achieved by the pilot plant.

It will be note that the average nitrate-N concentration significantly exceeded the average ammonia-N concentration in the effluent. This is in line with the expected system performance as discussed in the earlier section dealing with design considerations relating to combined nitrification/denitrification processes. It is also noteworthy that the effluent nitrate-N concentration, while low in absolute terms, exhibited a significant degree of scatter.

(c) **Phosphorus removal**

The pilot plant removed, on average, 73.5% of the influent TP. However, as may be seen from Fig 9 and as reflected by the high standard deviation of the effluent TP data (68% of the mean value), there was a considerable fluctuation in the percentage removal and in the effluent TP concentration. The use of chlorination to control the biological foaming problem may have contributed to the fluctuation in P-removal performance. Despite the fluctuation in performance, the plant clearly exhibited a significantly enhanced biological P-removal, the mean removal of 73.5% being nearly three times the expected removal for a conventional activated sludge process. This is borne out by the elevated P content of the sludge biomass which was found to be 4.66% on a mass dry weight basis, while the comparative value for a conventional activated sludge (Swords STP) was found to be 2.6%. The P-removal rate falls within the performance band of 3.3 - 5 g TP per 100 g BOD_5 removed reported for full scale plants (Peter & Sarfert, 1989).

The Ringsend wastewater was probably well suited to biological P-removal, in that, due to the size of the sewer catchment, the wastewater was in a relatively well fermented state by the time it had passed through the primary clarifiers. This is reflected in the highly negative ORP values for the anaerobic chamber, plotted in Fig 11.

The results presented in Fig 10 show that the addition of 10 mg/l of acetic acid to the influent wastewater over the 10-day period at the end of the test run had a twofold effect on biological P-removal (i) the effluent TP concentration was reduced to less than 1 mg/l and (b) the fluctuation in effluent TP concentration, characteristic of the prior pilot plant performance, was greatly reduced. This finding is in general agreement with other studies (Sedlak, 1991), which ascribe an important influence to the role of VFA in enhanced biological P-removal processes.

(d) **Control system performance**

The monitoring and control software developed for the project proved to be a very useful operational tool in running the pilot plant, providing an online continuous record of key process parameters. A reliable system for DO control is essential for biological nutrient removal systems. This is because of the necessity to maintain three distinct environments in adjacent flow-connected reactor zones. Thus, the DO concentration cannot be allowed to reach too high a level in the aerobic zone as the carryover of oxygen into the anoxic zone would inhibit the denitrification process. Likewise, the carryover of nitrate into the anaerobic zone would inhibit the enhanced biological removal of phosphorus by raising the environmental ORP level.
The aeration system, in which the three aeration zones were fed from a common header supplied from a speed-controlled air blower, was not ideal in that it did not permit the simultaneous control of DO level in the three aerobic zones. Various strategies for controlling DO were tried, including (a) using the average DO in the three zones (b) the DO in any one of the three zones, as the DO control parameter. In the end, the control strategy used was to control the DO level in the final aerobic zone at 1.5 mg/l.

CONCLUSIONS

1. The pilot plant study demonstrated the technical feasibility of achieving an integrated removal of BOD, N and P in a single multi-chamber reactor, provided the component anaerobic, aerobic and anoxic process environments are reliably controlled.
2. Over the 4-month test period, during which the wastewater temperature dropped from 18 ^{o}C at the beginning to 10 ^{o}C at the end, the pilot plant was operated at an average sludge loading rate of 0.223 kg BOD_5/kg.MLSS.d. The following average percentage removals were achieved:

BOD_5	97.8%
N	94.0%
P	75.0%

3. The pilot plant achieved an enhanced biological removal of P as demonstrated by the foregoing reduction in wastewater P-concentration, which is considerably higher than typically achieved in conventional secondary biological treatment (20% - 30%) and as also demonstrated by the raised P concentration in the activated sludge biomass (4.6% of dry weight solids).
4. There was a considerable fluctuation in the extent of P-removal by the pilot plant and hence enhanced biological P-removal may not be adequate on its own where discharge to a sensitive receiving water demands the achievement of a low P concentration with a high degree of reliability. In such cases, enhanced biological removal can be used, in conjunction with chemical precipitation, to effect a substantial reduction in the amount of precipitant chemical required.

5. Good performance from an enhanced biological P-removal process requires the elimination of solids carryover in the effluent and also careful excess sludge management, since the incorporated P may be released if the sludge becomes anaerobic.

6. The pilot plant achieved an excellent removal of N, indicating that this particular configuration of multi-chamber reactor provides an efficient process configuration for the combination of nitrification and denitrification processes. The process was also enhanced by taking advantage of the aeration system to generate additional internal recirculation between the aerobic and anoxic zones at no additional energy cost.

7. Biological foam or scum formation was a persistent problem. Although the technical literature indicates that scum formation is a common problem at activated sludge plants operating at low organic loading rates, it would appear that it is not a well-understood phenomenon. The remedial measures used on the pilot plant were a combination of chlorination and manual scum removal. It is probable that a continuously operating scum removal system would provide the best method of biological foaming control at full scale plants.

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References

Benmoussa, H, Martin, G, Richard, Y and Leprince, A (1986) Inhibition of nitrification by heavy metal cations, Water Res., 20, 1333

Burdick, C R, Refling, D R and Stensel, H D (1982) Advanced biological treatment to achieve nutrient removal, JWPCF, 54, 1078-1086.

Focht, D D and Chang, A C (1975) Nitrification and denitrification processes in wastewater treatment, Adv. Appl. Microbiol. 19, 153-186.

Foot, R (1992) The effects of process control parameters on the composition and stability of activated sludge, J.CIWEM, 6, 2, p215.

Fukase,T, Shibeta, M and Mijayi, X (1982) Studies on the mechanism of biological phosphorus removal, Japan Jour. Water. Poll. Res., 5, 309.

Gujer, W (1977) Discussion on the paper: Dynamic nature of nitrifying biological suspended growth systems, Prog. Water Tech., 9, 279.

Hockenbury, M R and Grady, C P L (1977) Inhibition of nitrification - effects of selected organic compounds, JWPCF, 49, 768.

IAWPRC Task Group on Mathematical Modelling for Design and Operation of Biological Wastewater Treatment Processes, Activated Sludge Model No. 1 (1987), Activated Sludge Model No. 2 (1994), IAWPRC, London.

Miyaji, Y et al. (1980) Biological nitrogen removal by step feed process, Prog. Water Tech., 12(6), pp193-202.

Peter, A and Sarfert, F (1989) Operational experience with the biological removal of phosphorus in the Berlin (West) sewage treatment plants, Korrespondenz Abwasser, 36, pp77-85.

Schlegel, S (1987) Bemessung und Ergebnisse von Belebungsanlagen mit Nitrifikation und Vorgeschalteter Kaskadendenitrifikation, Wasser Abwasser, 128, H8, pp422-431.

Sedlak, R (1991) Phosphorus and Nitrogen Removal from Municipal Wastewater, Principles and Practice, 2nd. ed., Lewis Publishers.

Standard Methods for the Examination of Water and Wastewater (1989) APHA, AWWA and WPCF, 17th. ed., Washington.

Stenstrom, M K and Song, S S (1991) Effects of oxygen transport limitation on nitrification in the activated sludge process, JWPCF, 63, 208.

Tipping, P J (1995) Foaming in the activated sludge process: an operator's overview, J.CIWEM, 9, p281.

USEPA (1993) Nitrogen Control, Technomic Publishing Co., Inc., Lancaster, USA.