THE BIOLOGICAL ASPECTS OF A WASTEWATER TREATMENT PROCESS POWERED BY A RENEWABLE ENERGY SOURCE

by Peter Kenneth Bradley

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DEDICATION

This thesis is dedicated to my fiancée Helen Brannon, who has had to suffer my tired outbreaks throughout the write-up, and has given me all the support that I could have asked for. Now we are one step nearer! It is also dedicated to my parents who have not only supported me throughout my university career, but have always been there for me, thankyou..

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CHAPTER 1 LITERATURE SURVEY

SUMMARY

The aim of this survey is to summarise the current wastewater treatment methods used in industry and outline new treatments that have or are about to come onto the market. Many of these new technologies are based around the more established systems such as the activated sludge process and the trickling filter. However, some have shown novel innovations in treatment protocols, such as combining two proven methods to overcome problems and produce an improved process. Many of the problems encountered centre on nitrification, so a brief outline of this will be given. Also covered here will be the variety of effluents that treatment systems could be asked to treat and their relative challenges. Finally the WATER process, a new treatment methodology on which this thesis is based will be introduced, its aims outlined and the conclusions from the survey relative to this process will be drawn.

1.1. INTRODUCTION

The nature of sewage both in terms of content, flows and loadings has never been entirely fixed, but has never changed as rapidly as at present. Currently, ever-increasing standards are demanded for treating domestic waters in the face of progressively increasing water-intrusive additives and equipment, for example washing machines and dishwashers. Until around the turn of the century, disposal of household waste, like vegetable peelings, was relatively diffuse, and had a relatively small impact on the environment (Metcalf and Eddy,

1992). However, with an accelerating emphasis on factory-produced, pre-packed, processed and often factory-cooked food there is now an ever-increasing quantity of domestic and trade effluent requiring treatment. Add to this other industrial and agricultural effluents including dye-residues, toxic metals, recalcitrants and xenobiotics, and one can indeed see the need for more stringent safeguards and the background leading to it. This has led to pressure for more effective treatment processes, not just large municipal plant but also suitable for small communities of up to 500 people or so, who, at present, rely on cess pits and septic tanks (Racault *et al*, 1990). Even such small communities are not immune from change, restaurants, fast food outlets and cafes all make their impacts on the environment. Terrain is also an important consideration in remote areas, for it can be expensive to install mains sewerage and large centralised treatment plants in mountainous areas or with dispersed populations as in Norway. Here there is an additional challenge of low temperatures, where plants using activated sludge technologies have often given unsatisfactory results because they could not handle seasonal variations in temperatures and treatment demands which caused seasonal overloading (Odegaard *et al*, 1990).

This problem of seasonal overloading is becoming increasingly prevalent in tourist areas, and in farming communities, which are always under pressure to improve treatment systems. Indeed, seasonal tourist activities in the countryside (camp-sites, ski resorts), and seasonal agricultural activities, such as those associated with food and wine production, often now require treatment units that can cope with substantial hydraulic overloads, while having only modest maintenance costs when they are idle. Further problems can arise from intensive

cattle breeding which has given rise to major problems of waste disposal throughout the EC, and such problems are exacerbated for remote farms in regions possessing fragile overloaded ecosystems (Rigby *et al.*, 1996). Here it has been increasingly common practice to dilute slurries to reduce aquatic toxicity connected with runoff of slurry liquors following land spreading (Rigby *et al.*, 1996). Clearly such problems are going to place increasing pressure on the design and operation of appropriate plant now and in the future.

Further pressure has also come from recent E.C directives here, which have required every member country to introduce legislation so as to impose progressively more stringent controls on the treatment and disposal of municipal and industrial effluent. This trend is set to continue. Several countries in the European Community, such as Denmark and the former West Germany, have already introduced appropriate legislation so that they comply with earlier amendments (Imhof, 1990); others have yet to do this. The contrasting levels of compliance within the E.C on directives on wastewater treatment largely reflect prior existence of established infrastructure. Thus Denmark and the former West Germany with relatively compact high density populations already have developed sewerage systems linking most of their inhabitants. The same is true of The Netherlands, where the sewerage network connects 95% of the total housing stock (Voorneburg, 1993) although at this level connection to the central system has now reached its economic limits (Voorneburg, 1993). In the Netherlands also there are additional pressures arising from restrictions to the dumping of municipal sludges on land (Voorneburg, 1993). Here and elsewhere, including

the U.K., there is considerable interest in treatment methods producing minimal amounts of sludge, and then only high quality sludge.

Increasing stringency in this country (England and Wales) will also apply to levels of nitrogen as ammonia, nitrate and nitrite in allowable discharges for population equivalents greater than 2000. Present levels for ammonia of 10-20mg L^{-1} will be reduced to approximately 5mg L^{-1} depending on the sensitivity of the receiving waters (Water Guardian, 1995). These directives, and consequent legislation, form part of the ongoing campaign to improve water quality standards, and more especially to alleviate avoidable occurrence of eutrophication in rural ponds and standing waters. Although, in England and Wales, the Urban Waste Water Treatment (1993) Act applies only to 2000 population equivalents or more, there is little doubt that similar standards will be increasingly applied to much smaller installations which are currently growing in popularity. This is perhaps not surprising if one remembers that the latest amendments to European regulations (Roberts, 1995) are now being targeted at small communities such as villages and farms. The remoteness of these communities has so far exempted them from the requirement to treat their effluents to the discharge levels of towns and cities. The trend towards small-scale works also reflects growing capabilities in control of plants, at least when the influent quality is broadly predictable and stable (Voorneburg, 1993). In this respect, the Act relies on Environmental Agency policy on smaller communities, and this policy, equally, is

becoming increasingly stringent. The present work, in its overall context, aimed at a population equivalent of 500 or fewer, is devised to meet such stringency.

In all of these examples there is obviously a major market opportunity for improved but inexpensive treatment systems, and it is this perspective that provides the context for a literature review below. In section 1.2 we discuss various treatment methods (large and moderate scale) currently in use. Section 1.3 covers enhancement methods. Section 1.4 covers nitrification and the problems associated with it. Section 1.5 covers various effluent types and their relative challenges. Section 1.6 covers the WATER project, and combined with section 1.7 the conclusions and aims of this work which will be covered in the subsequent chapters.

1.2 TREATMENT METHODS

Apart from septic tanks for very small communities (Fastenau *et al,* 1990), there are many varieties of systems potentially suited to meeting the demands mentioned previously. These systems use two main methods; either the activated sludge process (Henze, 1989; Stanbridge, 1976) or the trickling filter (Gousailles, 1989). More recent variants include, the ICI Deep Shaft (Staples, 1978), the Pasveer Ditch (Mandt and Bell, 1982), suspended biofilms (Lessel, 1989), and reed beds (Cooper *et al,* 1990), and more recently, Biological Aerated Filters (BAF's) and Rotating Biological Contactors (Ware *et al.*, 1990) are

mentioned. There may also be interest in systems utilising immobilised bacteria (Tanaka, 1989), and more especially in technologies powered by renewable energy sources such as wind (Janes and Thomas, 1992). All employ mixed microbial populations to reduce the levels of BOD (Biological Oxygen Demand); COD (Chemical Oxygen Demand) suspended solids (SS) and ammonia. Of principal concern here is the reduction of these quantities at least to levels outlined by the National Rivers Authority (NRA) for discharge to water courses. The limits imposed vary with locality but may be approximated to 10mg L^{-1} of ammonia, 20mg L^{-1} suspended solids and 30mg L^{-1} for BOD.

BOD (i.e. non-specific bio-digestible organic material) is degraded (Stainer, 1988) by several classes of bacteria and fungi. Some of the more significant are likely to be *Pseudomonas, Chromobacterium, Sphaerotilus, Ascoidea, Sepedonium, and Oospora*, all of which are involved in biological activities. Bio-degradation involves conversion of complex organic matter into smaller more accessible components before metabolism. Biodegradation of this sort is essential to biological treatment, which is invariably augmented and accelerated by the formation and settling of flocs (Mattock, 1980). Floc formation is associated with the microbial secretion of polymeric adhesives that promote adhesion and accumulation of particulate solids (Degremont, 1989). Flocculation can be accelerated by the addition of certain chemicals such as aluminium sulphate and ferric sulphate. In all suspended growth systems, it is the enhanced settling provided by flocs, that decides the BOD degradation rate.

1.2.1 Activated Sludge (AS) Processes

AS is the most widely used approach used in the developed world (Pitman 1989). The process (Fig 1.1) involves the aeration of screened, settled sewage in the presence of microorganism flocs i.e.: the "activated sludge". Organic wastes are then degraded by the bacteria, either within the flocs following adsorption or directly by free microbes (Metcalf and Eddy, 1992). Such parameters as inlet quality and tank configuration, loading and throughput rate, all contribute to the complexity of the problem. Solutions are available whether the process is operated as a stirred tank or plug flow. Surface aerators or bottom diffusers normally achieve mixing and aeration. Following bio-oxidation, the liquor is transferred to a sedimentation tank where the flocs are settled, before discharge of treated effluent. A proportion (80-90% typically) of the AS is recycled to sustain sludge age and performance, whilst the treated effluent is either discharged to a water course or piped to tertiary treatment, polishing for example (fine settling) and/or disinfection according to local requirements.

The performance of AS plants is usually assessed (Henze, 1989) on the basis of the minimum holding time for reductions in BOD, SS and ammonia levels. Aeration retention times depends on temperature, mixed liquor suspended solids and the influent BOD/ammonia level, but they typically range (Lewin, 1978) from 9-11h for BOD loading of

250mg L^{-1} oxidised to a discharge limit of 30mg L^{-1}. It is clear that the main duty of any treatment plant is to remove this carbon load.

However, a further problem for these plants is the removal of nitrogen, as in most countries, complete nitrification is mandatory. With this in mind an alternative measure of performance is sludge retention time. This is the weight of suspended sludge (SS) retained divided by the weight removed per day; commonly 10 days is acceptable for the removal of ammonia. This approximation is, dependant on temperature as can be demonstrated by the equation below (Taken from Forster, 1995).

$$SA \text{ (minimum sludge age)} = 3.05 \times (1.127)^{20 - \text{temperature}}$$

Both aeration retention time and the sludge retention time are used in comparing the relative performances of AS units. In practice the sludge retention times may exceed nine days in established units (Nakajima, 1989), suggesting that bio-digestion (endogenous metabolism, without production of more cells) improves with time as plants become thoroughly established. Under these conditions, about 90% nitrification may be achieved (Nakajima, 1989), the remaining 10% (i.e. 1-4mg L^{-1}) comprising soluble organic nitrogen (Henze, 1989) which is considered inert. Some principal variations of this basic theme are now surveyed as follows.

With immobilised bacterial populations as those shown in Fig 1.2 (Tanaka, 1989) the AS process can be configured to incorporate some segregated, immobilised bacterial populations. For instance, in one such case (Fig1.2) nitrification is achieved by passing the liquor through a bed of fixed porous pellets supporting nitrifying bacteria as an immobilised film (Tanaka, 1989). There are several claimed advantages over standard floc systems, most notably lower reactor volumes (reduction of 35%) and shorter retention times (reduction of 70%) for prescribed performance in nitrification, usually complete (i.e. up to 95% removed). The immobilisation medium used here (Tanaka, 1989) was cross-polymerised polyethylene glycol (PEG), as porous pellets of diameter 2-3mm with claimed durability and compression resistance. This system delivered nitrification times of 6-8h, compared with 18h for conventional plant under similar operating conditions and similar hydraulic loadings ($36m^3$ day^{-1}). Even better performance was achieved using an airlift reactor, with recycled liquor, which reduced the retention time by a further 30%, equivalent to a nitrification rate of 15mg L^{-1} day^{-1}. This work indicates that immobilisation can provide a system offering reliable, rapid and near complete nitrification, with a very significant performance improvements over conventional AS designs.

Further improvements in performance are also claimed by Cooper *et al* (1995) who reported compartmentalisation (Fig 1.3) as a method for enhancing biological performance. They claimed that a compartmentalised system "incorporating two anoxic zones instead of the standard one, improved overall nitrification within the plant". Favoured layouts are shown

in Fig 1.3, with various sites for sludge incorporation, in relation to zones of anaerobic, aerobic, and anoxia (low oxygen, high ammonia) operation.

In the continuous systems of traditional AS plants aerobic reaction and solids settlement are preferred sequentially in separate units. Normally there should be steady liquor flow and a steady return of AS from the settler to the reactor. In reality, of course, (Barnes *et al.*, 1981) the plant must be designed to accommodate fluctuations of organic and hydraulic load. An alternative strategy is the "continuously-fed intermittently-decanted system" incorporating the same two basic unit operations but within a single tank (Fig 1.4.). Separated in time, then, the single tank functions alternately as an aerobic reactor and as a settlement tank. As conceived here there is no control over the entry to the chamber or equalisation. Such systems are designed as "extended aeration" processes (aeration lasts up to twice as long as expected), and tank sizing is normally determined by the interval required for aerobic biological reaction (Goronszy *et al.*, 1983).

Operated sequentially, it provides carbonaceous oxidation, nitrification, denitrification, solids separation and decanting. The influent is continuous, while the effluent is withdrawn intermittently, and this, of course requires that aeration be intermittent to allow settlement. Initiated and monitored by a time-base process controller these systems are operated with submerged aeration units instead of surface aerators, due to the requirement for anoxic conditions during settlement.

Intermittent aeration here involves several short periods of aeration, rather than the continually maintained aeration of established AS plants (Nakajima, 1989). Investigations showed that when aeration changed from continuous through greater intermittency, BOD reductions varied only slightly from 89-98% of their original values. However, nitrification was much more sensitive with their values varying from as low as 32 up to 93% of their original value, reflecting nitrifiers', susceptibility to anoxia. For a BOD loading rate 0.2kg m^{-3} day^{-1} (200mg L^{-1}), highest nitrification was achieved with 4 x 5h aeration periods daily (each separated by 1h) and corresponding to consumption ratio of 3-4 (kg DO_2/kg BOD), a BOD removal of 98%, and nitrification to 93% of the starting level (50mg L^{-1} ammonia). Using submerged filters, with the nitrifying bacteria grown on packed supports for improved stability, Lessel (1989) recorded reductions of 50% in total nitrogen although the intermittent regime inhibited any worthwhile denitrification (i.e. insufficient anoxia).

Other work by Nakajima (1989) found 6-8% decrease in performance as % overall BOD reduction when initial loading was reduced from 350 to 250mg L^{-1} with a corresponding reduction in nitrification of approximately 16%. In this case it seems that more than 70% total nitrogen was converted via nitrate to gaseous nitrogen (i.e. substantial denitrification). According to Takeuchi (1989) aeration is crucial in achieving satisfactory nitrification, with the temperature also being very important as his results indicated a 45% decrease in nitrification in winter (summer rate 12-16mg L^{-1} day^{-1}, winter rate 7-9mg L^{-1} day^{-1} of oxidised ammonia). This is consistent with the idea that at 10°C, complete nitrification will

take 10 days. The importance of temperature in connection with nitrification is also outlined by the NRA's discharge limits, which vary with season (around 5mg L^{-1} in winter, 1mg L^{-1} in summer). All of this work clearly shows that nitrification in activated sludge plants is very dependent on adequate aeration, and any lapses in this air supply can cause considerable problems.

1.2.2 Trickling Filters

Manifestly distinct from the AS process and variants outlined above, the trickling filter (Christensen, 1989; Stanbridge, 1976), represents the only other long-standing option which has been widely adopted. Trickling filters plants combine a bed of permeable medium on which the microorganisms grow/attach and through which the wastewater is percolated or trickled under gravity. Oxygen is supplied either from upward circulating air via the drainage/collection system (Metcalf and Eddy, 1992), or compressor-driven forced aeration. A stratified bio-film (Fig 1.5) grows on the medium, in which the top layer comprises mainly aerobic heterotrophic species formed by high proximity BOD, while nitrifiers proliferate near the base, which is highly aerobic but where the carbonaceous material has already been largely oxidised. The depth of the unit is designed to meet the demands of hydraulic and organic loading on the filter, 100m^3 day^{-1} and BOD and ammonia in the medium strength range (200-250mg L^{-1} and 30-50mg L^{-1} respectively). An alternative media, i.e. high voidage, low mass plastic beads, compared with gravel can increase performance (Sarner, 1981) by encouraging higher biomass densities and effluent flow rates. The high rate filters

can still have satisfactory nitrification, even at the higher carbon loadings (Metcalf and Eddy, 1992), with increased performance achieved using pre-treatment as a roughing filter to remove most of the BOD. As a control on the biological loading and to sustain wetness of the bio-film, part of the clarified outlet is sometimes recycled (Metcalf and Eddy, 1992).

As with all biological treatment systems, filter efficiency (i.e. activity of the bacteria) is rather sensitive to temperature, and shortfalls here can to some extent be offset by increasing the depth (increases retention time) as well as lowering the biological loading rate. There is now a broad consensus (Christensen, 1989) that trickling filters really need to be augmented with AS so as to increase BOD removal, or to be buffered for increased capacity as water treatment demands increase over the years. Activated sludge certainly oxidises BOD more quickly than trickling filters but on the other hand the trickling filters deliver higher clarification. Typical limits for AS are 10mg L^{-1} ammoniacal nitrogen and 30mg L^{-1} BOD in 16-18h, whereas for a trickling filter the equivalent performance takes 4-6 days. This is not saying that hybrid systems incorporating both activated sludge and trickling filters are the whole answer, but they are an improvement.

1.2.3 Rotating Biological Contactors

As indicated in the schematic Fig 1.6, RBC's comprise a series of discs supported on a central shaft and partially submerged in the liquor. The discs act as bio-film supports and their rotation results in cyclic alternation of submergence and aeration (Pescod *et al.*, 1983).

Typical disc materials are polystyrene and polyvinyl chloride. If properly designed, and there are practical problems here (Ware *et al.*, 1990), RBC's can provide superior performance because of lower organic loading per unit mass of biological solids; also there can be longer retention times exposed to active biology and better control of short circuiting (Metcalf and Eddy, 1992). However, RBC's are more temperature sensitive than standard activated sludge or trickling filters due to exposure of the bio-film to changing air temperatures and they can suffer from drying out and odour problems. High maintenance needs, can cause problems, and it is well documented that this type of treatment system suffers from problems such as shaft failure (Ross *et al.*, 1994).

1.2.4 Oxidation Ditch /Facultative Lagoons

Both methods represent extended versions of AS except that the bacteria are usually largely retained rather than recycled and oxidation ditches typically operate with 1-3% (air/v) whereas AS with forced aeration range from 10-15% up to 25-30% for intense sludge blanket reactors (Bailey and Ollis, 1988). Lagoons, particularly, can vary from natural aerobic ponds where aeration is delivered by the wind and photosynthetic products of algae (Metcalf and Eddy, 1992) to mechanically aerated plants; essentially activated sludge plants but in a pool. The aerobic ponds will, typically, exhibit complex prey-predator system characteristics that are not easily controlled. Further problems include possible poor performance for stronger effluents (> 50mg L^{-1} BOD) and a frequent requirement for large sedimentation tanks because of low solids settling velocities. As a consequence they are

most commonly employed for polishing or for sludge digestion (Degremont, 1989). Other developments here include (Rachwal *et al.*, 1983) incorporation of mechanical or pumped aeration, boosting performance by 25%. One configuration for surface aeration utilises a rotating spiked cylinder, which promotes surface disruption (Rachwal *et al.*, 1982). Aeration is also achieved using driven impellers, either supported on pontoons, or on tethered floating buoys. Rotors, like the TNR, however, suffer from high maintenance problems, and other mechanical disadvantages.

One variation of the oxidation ditch, which has attracted interest over the past two decades, is the "Carousel" system (Rachwal et al, 1983). Here one or more vertical shaft aerators are employed to circulate the liquor around a closed circuit (Fig 1.7). The aerator rotation imparts angular momentum which is partly deflected by the dividing wall resulting in translation around the channel at sufficient speeds (0.2-0.5m s^{-1}) as to cause turbulent flows over the cross section of the channel. The mixed liquor sustains alternating zones of aerobic/anoxic conditions, the former achieving nitrification and the latter (possibly) denitrification. A demonstration plant (Johnstone *et al.*, 1983) monitored over a 3 year period was shown to reduce ammonia by 82.5-94.5% from 20mg L^{-1} at entry and BOD reduction by 95% from 160mg L^{-1} at entry.

1.2.5 Deep Shaft

Whilst the activated sludge principle is employed (Staples 1978) in so far as the process is aerobic with mixed population biology, aeration here is achieved using a high-pressure airlift loop (Fig 1.8). High pressure here derives from the considerable depth of the shaft (150-200m) which exploits the high head to enhance dissolution of oxygen from the bubbles. Concentration of oxygen at the base of the downcomer is typically as much as 6-7mg L^{-1}, ideal for accelerated oxidation. For comparison with other processes the unit has a retention time of 1.5h, a sludge age of 2.6-2.8 days, requiring 1.3kg dO_2/kg BOD and an overall power requirement of 39.3 W/capita. D. This, in turn, means that the claims made for the device rest on small ground area, rather than economy, and there can be problems over rotting of mechanical supports, and of de-aerating the settling sludge.

1.2.6 Septic Tank

The traditional approach (Metcalf and Eddy, 1990) for remote and unsewered areas, septic tanks are prefabricated units that serve for combined settling and skimming (removal of large non-settleable solids i.e. toilet paper) delivering limited anaerobic digestion between emptying. However, their performance in treatment is poor (negligible BOD degradation) and their continuing use in the developed world is justified only because of the cost of replacement (Fastenau *et al,* 1990). Although settled sludge has to be removed from the tank, and tankered for treatment, pierced pipes distribute the effluent into surrounding soil

(Fig 1.9). Whilst the method may be adequate for remote dwellings, it is not suitable for larger communities or farms because of contamination risk to groundwater (Racault *et al*, 1990), and general water company policy in the UK is now to sewer even isolated dwellings where at all possible.

1.2.7 Artificial Wetland Treatment

Planted soil filters or artificial wetlands (Fig 1.10) typically involve plants, soil and hydraulic channels into which the wastewater is introduced, usually as screened or settled sewage (Cooper *et al*, 1990). Plant selection is most often from species of reeds usually propagated either from seeds or German rhizomes. There are several different parameters on which the treatment processes are broadly dependent. The most important of these are bed size and flow rate/distribution (Findlater *et al.*, 1991). Brief outlines of each of these is given overleaf:

1) Bed size: The important factor here is the rate of oxygen transfer into the bed/root zone, typically between 5-30g O_2 day^{-1} m^{-2}, with the lower end of this range more likely (Armstrong *et al*,. 1988).

2) Flow rate: The hydraulic conductivity of the bed needs to be sufficient for the wastewater to make contact with the root zone (Hobson, 1989). To

encourage a high hydraulic conductivity, most U.K. reed beds are constructed with gravel rather than soil.

Both entail elements of physical filtration by the substrate including plant debris and it seems that such deposition in the bed is a major contribution to the treatment achieved (Findlater *et al.*, 1991). Of course, this decomposition does depend on the content of indigestible substrates like heavy metals, their likely rates of decomposition in or on the bed, and the subsequent questions, which are then going to arise about plant efficiencies.

Claimed advantages of reed bed systems are low operating costs, relatively low capital costs of £100 to £300 per population equivalent depending on land availability, low maintenance cost and a good reduction in BOD and suspended solids from low strength waste-water. On the other hand, major disadvantages include large land occupation coupled with substantial capital outlay in preparing the substrate and prolonged period for initial equilibration (3-4 years). Difficulties with nitrification have promoted the use of up-stream high intensity treatment plant (Woodruff *et al.*, 1994) with the reed bed now employed only for final polishing. This requirement for further plant also means that several of the advantages, such as low maintenance and no mechanical equipment no longer apply.

1.3 PERFORMANCE ENHANCEMENT

Many wastewater treatment systems incorporate pre-processing preparations including suspended solids reduction, precipitation and biofilms. The aim of this pre-processing is to enable the main system to reduce the contaminants to discharge levels without using inordinately long residence times.

1.3.1 Precipitation

Pre-processing precipitation usually involves the addition of flocculent chemicals, typically aluminium sulphate but occasionally manganese sulphate (Karlsson and Smith, 1989). Floc formation traps and accumulates carbonaceous particulate matter (BOD), as well as phosphate, and nitrogen-containing compounds. Removing suspended solids in this way prolongs desludging intervals and so increases sludge age in the reactor. This in turn provides higher stability and activity of the bacterial population. Typically the bio-oxidation rate of organic matter is raised by 8-12% (Karlsson and Smith 1989) following pre-treatment by precipitation.

1.3.2 Biological Aerated Filter's

Biological aerated filters accomplish both solids capture and biofiltration, and operate very much like undergravel filters but with the advantage of easy cleaning. They are generally classified as "expandable granular biofilters", and are distinguished by the use of a buoyant granular medium. The packed bed of plastic beads through which the waste water is passed,

captures solids and simultaneously provides a large surface area for cultures of bacteria (Nitrosomonas sp. and Nitrobacter sp) which remove dissolved nitrogenous wastes. Bead filters remove suspended solids by at least four different mechanisms as the water is passed through the bed. Physical straining is probably the most dominant mechanism for the larger particles (> 80 microns), other process include interception and bioabsorption. The other process of note is biofiltration and is dependant on the formation of a stable microbial layer on the beads, to extract dissolved chemicals from the water and convert them to particulate biomass or harmless dissolved compounds (Aquaculture systems; Robinson *et al.*, 1994)

1.3.3 Combined Processes

The purpose here is to incorporate the advantages of simplicity offered by AS and efficiency provided by trickling filters. Since attached growth process are more resistant to departures from equilibrium, especially those of shock loadings, whilst suspended growth systems are more efficient, recent effort has been given to combining both qualities (Newby *et al*, 1988). Although there are numerous variations on detailed specifications, the broad thrust is to use trickling filters for BOD reduction and shock loading, followed by AS for nitrification. Fig 1.12, shows various combined systems (Nakajima, 1989) and how different processes are alternated to achieve the best overall performance. Combined, such systems can tolerate higher BOD loadings than can be achieved individually. However such systems often involve sequential operation via a series of tanks and thus considerably increase costs.

Perhaps the major underlying consideration for any wastewater treatment system using microorganisms is the need for maximal surface area at the cell surface for exchange of oxygen and nutrients. In other words in an ideal system a layer of only one or two cells thick would clearly provide the best exchange conditions. In many cases, when the area is reduced by diffusional limitations, following increasing thickness of bacterial film, the result is anoxic conditions locally (Degremont, 1989). In trickling filters, the surface area is no greater than $200m^2\ m^{-3}$ of reactor whereas with bead supports the specific surface area may be as much as $750m^2\ m^{-3}$. The latter types of reactors can treat sewage in a single stage, but the high bacterial densities demand high oxygen supply, even to the extent of using pure oxygen (Metcalf and Eddy, 1992). On the other hand sludge formation (and therefore land demand) is small although nitrification deficiencies still remain to be resolved (Haddad *et al*, 1991).

In summary these various process pose some interesting questions, including overall cost-effectiveness of many, especially at small scale, the ease of operation and robustness. There is also the question of how sensitive nitrification is, and the effects of shock loading. All of these need to be addressed if a new system is to be accepted.

1.4 NITRIFICATION

Nitrification poses a major challenge for many treatment systems (Fig 1.13). The importance of this aspect of the total process is often recognised by severe regulatory constraints placed on the release of nitrogen-containing compounds to the environment (Roberts, 1995). Domestic sewage (30-70mg L^{-1} ammonia, 100-400mg L^{-1} BOD, 400-2000mg L^{-1} COD) is ammoniacally weak compared with many industrial and agricultural wastewater's (50-200mg L^{-1} ammonia, 500-2000mg L^{-1} BOD) but nevertheless is still strictly controlled. Indeed in some areas of the U.K restrictions are tighter than the normal 40:20:10 with maximum ammonia limits (2-3mg L^{-1}). Biochemically it represents the oxidation of ammonia to nitrite and nitrate (Prosser, 1989) via two sequential stages involving different bacteria (Stenstrom, 1991), where the first stage utilises ammonia and the second nitrite as specific energy sources. Oxygen is the terminal electron acceptor and carbon dioxide or carbonate is the carbon source (Prosser, 1989).

The first stage, oxidation of ammonium ions to nitrite; is called nitrosification (Stryer, 1995), and can be represented as:

$$NH_4^+ + 1.5O_2 \text{ ------> } NO_2 + 2H^+ + H_2O$$

The reaction is catalysed by the genus *Nitrosomonas*, with two species of this genus *Nitrosomonas europea* and *Nitrosomonas monocella* most frequently involved. Other genera have also been identified, including *Nitrosococcus, Nitrosopira, Nitrocystis* and *Nitrosogloea* (Belser, 1979) but are considerably rarer than *Nitrosomonas*.

Following nitrosification, the nitrite is then oxidised to nitrate, via the reaction:

$$NO_2^- + 0.5\ O_2 \text{ ------> } NO_3^-$$

for which the bacterium responsible is *Nitrobacter*, with *Nitrobacter winogradskyi* being the only well defined species (Watson *et al.*, 1981). Other species that have been isolated more recently including *Nitrobacter agilis, Nitrocystis, Nitrococcus* and *Nitrospira,* but their effectiveness for nitrite oxidation is not well documented (Belser, 1979). Lack of nitrite accumulation under steady state conditions with adequate aeration indicates that nitrosification is the rate-limiting stage. In operational terms nitrification can be sustained only below critical sludge loading rates of $0.15 kg^{-1}\ d^{-1}$ corresponding to sludge ages greater than 4 days or so (Stanbridge, 1977). This poses a practical challenge in so far as such sludge ages may be incompatible with those required for optimal BOD reduction (Prosser, 1989).

1.4.3 Importance of Oxygen

With dissolved oxygen acting as terminal electron acceptor for nitrifying organisms, the process is sensitively dependent on dissolved oxygen concentration. In AS processes it is generally accepted (Prosser, 1989) that nitrification does not occur below about 1% oxygen saturation. Similarly, essentially no nitrifying activity occurs when sludge is stored anoxically, even for short periods (Metcalf and Eddy, 1992). Indeed prolonged anoxic exposure (>1day), can lead to bacterial death (Stryer, 1992). Reported dissolved oxygen saturation concentrations (K_s) for pure cultures of *Nitrosomonas* and *Nitrobacter* range over 0.25-2.5mg L^{-1} (Painter, 1986), higher than those for heterotrophs and consistent with nitrifiers being poor competitors at low dissolved oxygen (Prosser, 1989). However, the effect of low dissolved oxygen on nitrifier metabolism is not well understood although ammonia (and nitrite) oxidation is substantially slowed at low concentrations of dissolved oxygen (Laanbroek, 1994). The main inhibitory effect, thus seemingly concerns nitrite oxidation since nitrite accumulation typically results under these conditions.

More detailed studies (Prosser, 1989) suggest that low dissolved oxygen has a complex broad band inhibition of the whole set of processes. Thus, in reality, specific ammonia oxidation rate (i.e. per unit biomass) is also reduced at low dissolved oxygen although population growth is enhanced which compensates accordingly. In other words the *Nitrobacter* seemingly utilise metabolic compensations to compensate for anoxia, resulting

in zero net effect under a wide range of typical conditions. On the other hand, nitrite oxidisers have no such compensating metabolism and nitrite production is significantly reduced at low dissolved oxygen. From a practical point of view, adequate dissolved oxygen provides the key to specification for adequate ammoniacal oxidation.

1.5 CONCLUSIONS

Activated sludge processes in the UK, have largely replaced trickling filters as the method of choice. Energy enhancement methods, including the incorporation of anoxic zones, pre-precipitation and biofilms as performance boosters for AS systems. Capital outlay on multi-tank treatment plant and energy demand during lifetime operation are now of increasing importance in plant specification due to the economic environment. Selection determents also include discharge constraints, sizing and suitability for job, maintenance requirements, infrastructure, climate and land availability (Racault *et al.* 1990).

So far as small plant is considered (say up to 500 population equivalents), AS sludge plants are not considered appropriate due to their sensitivity to hydraulic excesses. Aerated biofilters and rotary Contactors currently are supposedly more appropriate at least as upgrades for the traditional septic tank. Indeed as a broad bottom line, not only septic tanks but also facultative lagoons may require additional aeration to ensure adequate performance.

From these statements, we can draw some conclusions. With the discharge level stipulated it is well known that efficient processes are needed. This means that the future of such systems as lagooning, septic tanks and reed beds is weak. For very small plant, activated sludge systems have to be discarded to the benefit of more overload-resistant processes. A good solution should be a batch plant system with a storage tank. Systems that can work with or without sewer connections are preferred since users do not need to install one. The temperature profile over the year is also vital, due to the temperature sensitivity of some systems. Compact modular installations are often preferred especially if the surrounding area is of natural beauty. Often the need for large volume treatments lasts only part of the year (tourism), therefore units with minimal operating costs are advantageous.

1.6 The WATER Project.

WATER (**W**ind **A**ssisted **T**reatment for **E**nvironmental **R**ecovery) is a bio-oxidation process developed by **FRED** Ltd, a Research Park Company at Birmingham University, in part under sponsorship from DTI through SMART awards in 1990 and 1991. As configured following an extensive experimentation and evaluation exercise, **WATER** is a multi-stage fed-batch bio-oxidation process in which aeration and agitation are provided by drawdown (snorkel) aerators moving in the tank around a rotating central axis, powered by a Vertical Axis Wind Turbine (VAWT). Biological activity and stability have been extensively examined in lab and pilot tests supported by validation of mechanical

effectiveness and robustness in prolonged field tests of the VAWT as power source. At the time of starting the project (October 1992), extreme 1/2m model tests (Janes *et al.* 1992) had shown BOD removal complying with NRA regulations (10-30mg l^{-1}).

Since the purpose of the project as a CASE Award with FRED Ltd was specifically to focus on the biological performance of WATER, it is appropriate here to consider the wider issues before moving on in the following chapters to the specific purpose of the project. This is the practical implementation of wind power and the applicability of WATER is defined by site availability of wind resources, albeit with scope for reassessing the VAWT. At concept level, feasibility had already been described and reported by Janes and Thomas (1992), in wind terms with a mean speed of 6m/s sufficing. Of course remoteness forces unconventional power systems to be considered as the supply of electricity to these areas can cost around 5$/kWh for such things as the cost of installing the necessary land lines etc. The local topography has a considerable influence on the wind speed (Palutihof *et al.* 1987) and regularity of supply via hills and valleys, coastal zones etc. The major challenge here is not so much sufficiency over the longer term but day to day inconsistencies and the biological response to slackish periods. It was this aspect especially that provided the focus for this study as reported in the following chapters.

Chapter 2 covers the effects of oxygen-limitation on the bacterial populations found in the WATER system, especially the nitrifiers, which seem to be particularly affected. The main

problem here is that information on a minimum time for recovery after oxygen limitation is essentially not known, and this must be established before strategies such as detection and utilisation of hardier strains, less susceptible to limitation can be implemented. This work will therefore be done both at shake-flask level and then verified at both half-metre pilot plant and 2m pilot plant scale. Chapter 3 looks at the use of a microbial support matrix, to stabilise and retain microbial flocs in their designated zones, and assesses performance increases from this compartmentalisation. Chapter 4 covers the potential problems of using a support matrix, such as blocking. Chapter 5 will then continue this by assessing performance increases arising from combining both an anaerobic pre-treatment step, as a "buffer" against toxic and shock loads and a final denitrification step, to remove nitrate.

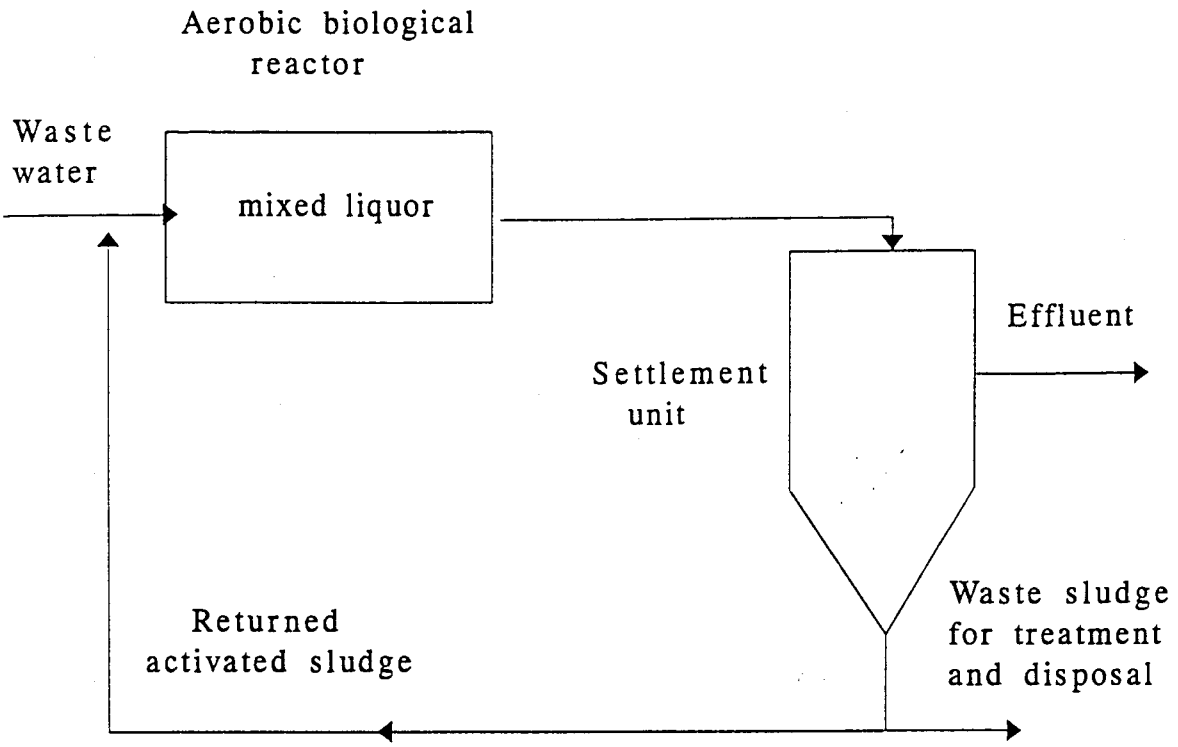


Fig 1.1 Schematic of activated sludge
sludge processes (Metcalf and Eddy, 1992)

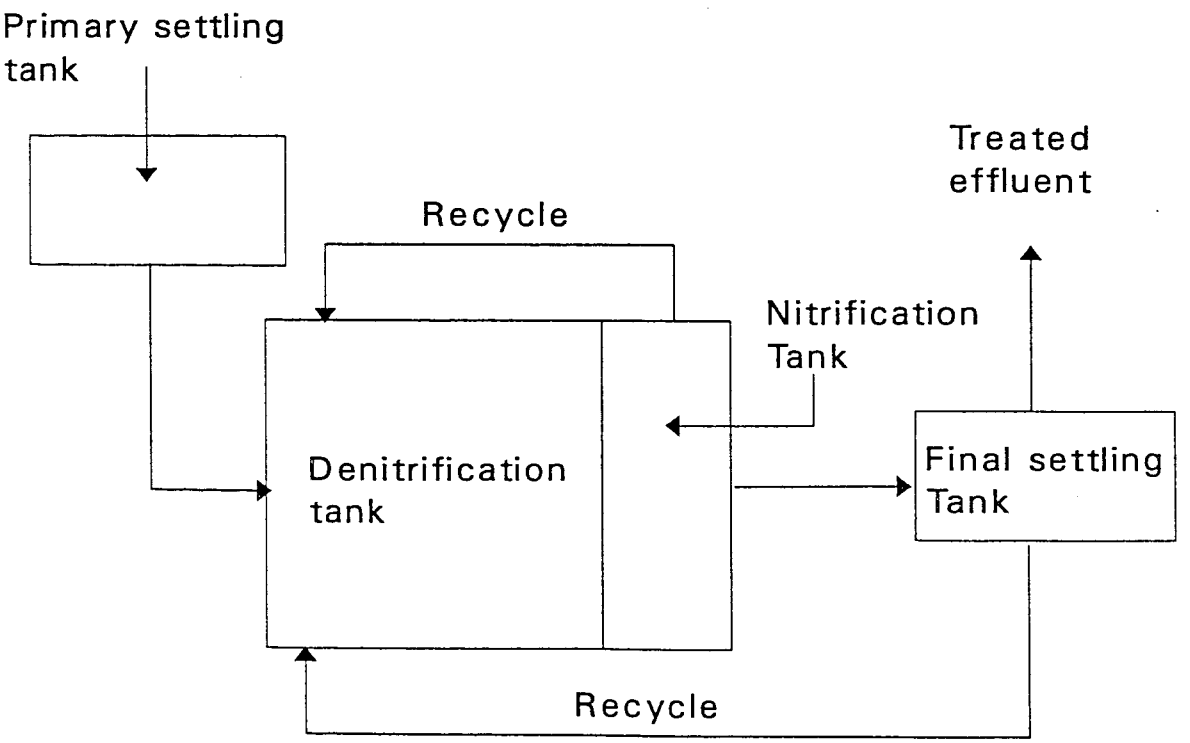
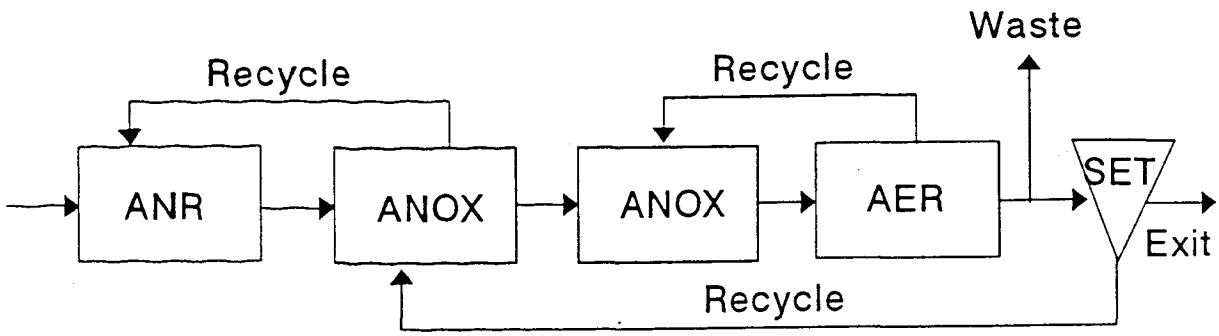
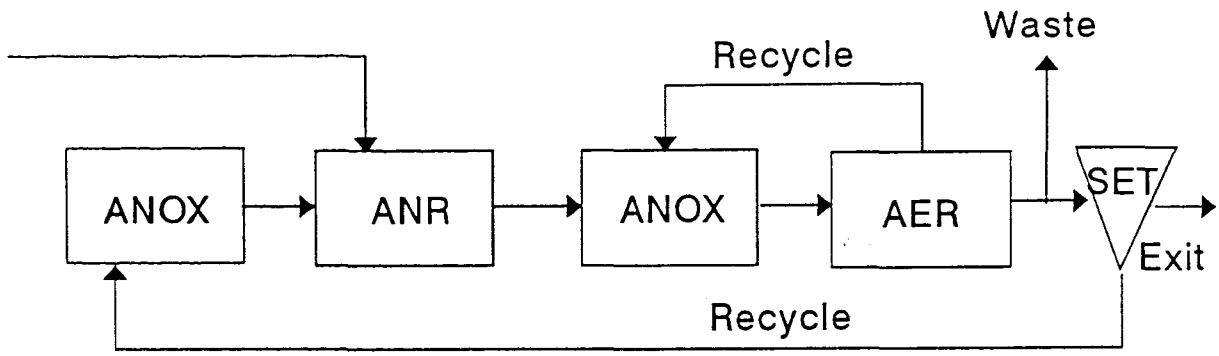


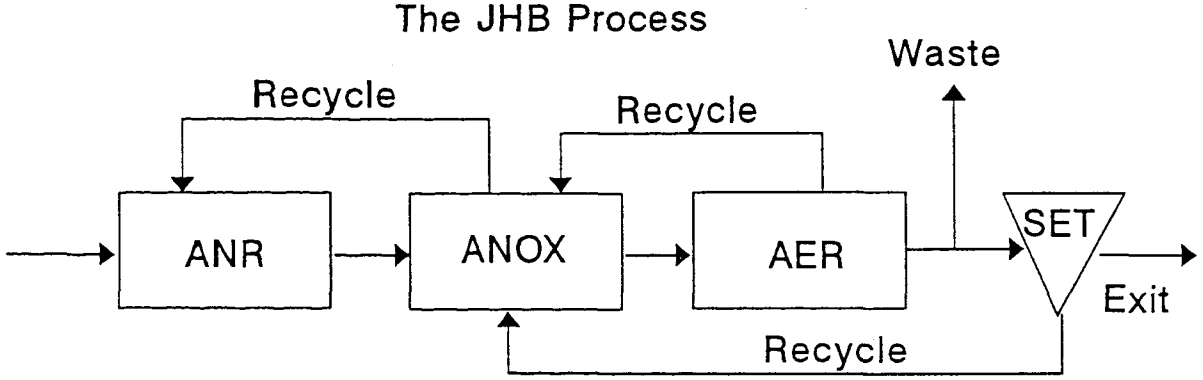
Fig 1.2. Immobilised system, reduction in throughput time is from 16h to 6-8h. (K.Tanaka, 1989).



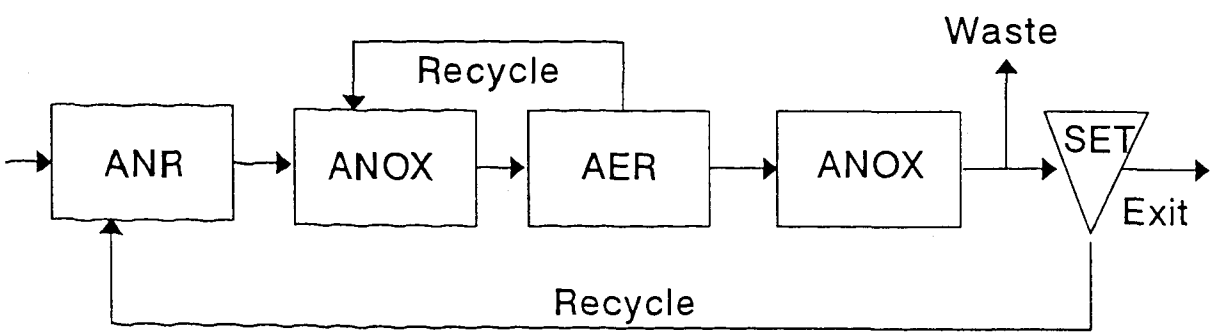
The modified UCT Process



The JHB Process



The UCT Process



Phoredox Process

Fig 1.3. Various compartmentalised systems (Pitman, 1989).
ANOX=Anoxic, ANR= Anaerobic, AER=Aerobic

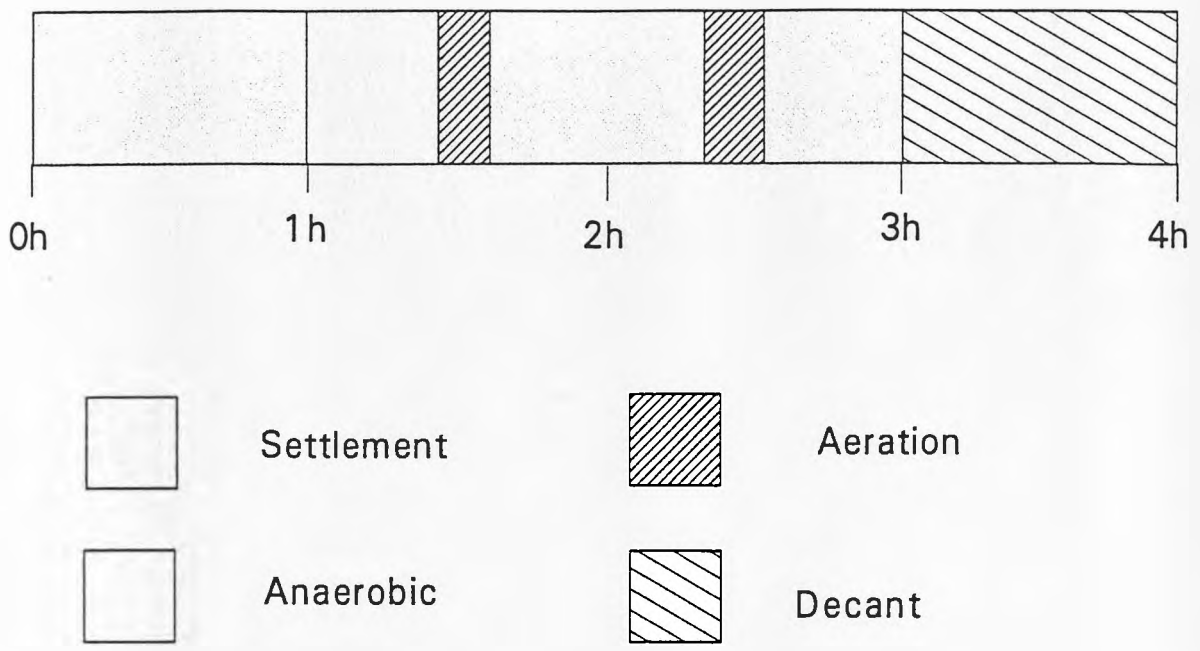


Fig 1.4. Use of intermittent aeration. Two 10min bursts from the aerators. This also provided the mixing (Barnes et al, 1981)

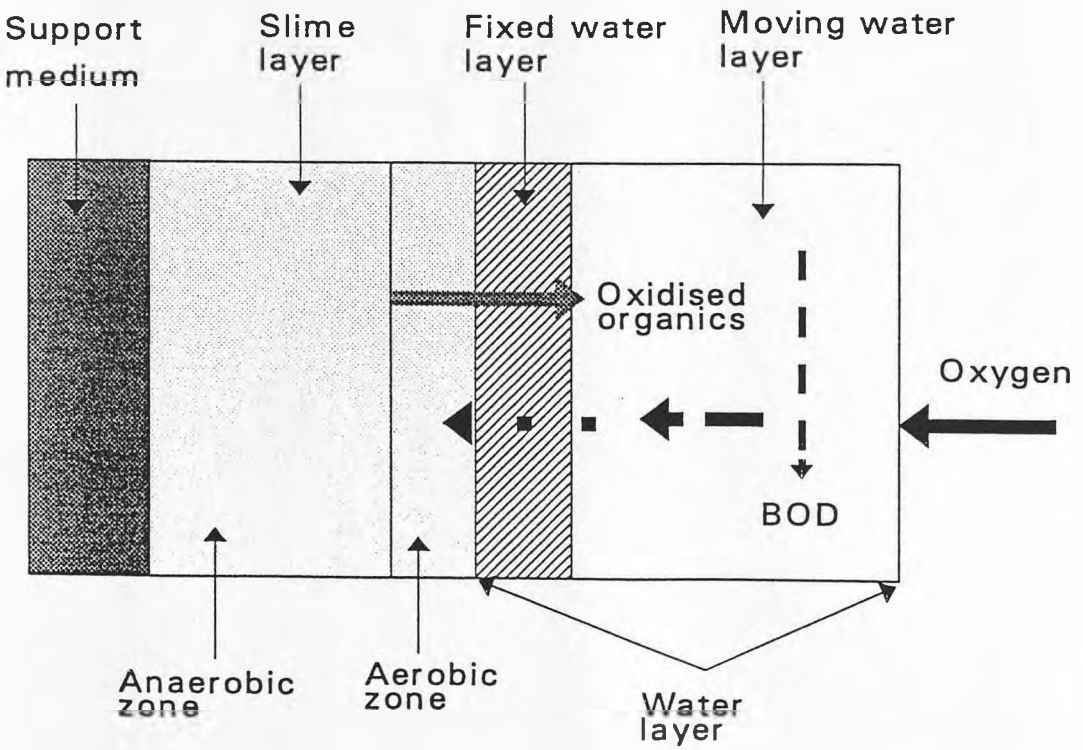


Fig 1.5. Schematic of BOD removal from a percolating filters support surface, From McGhee, (1991).

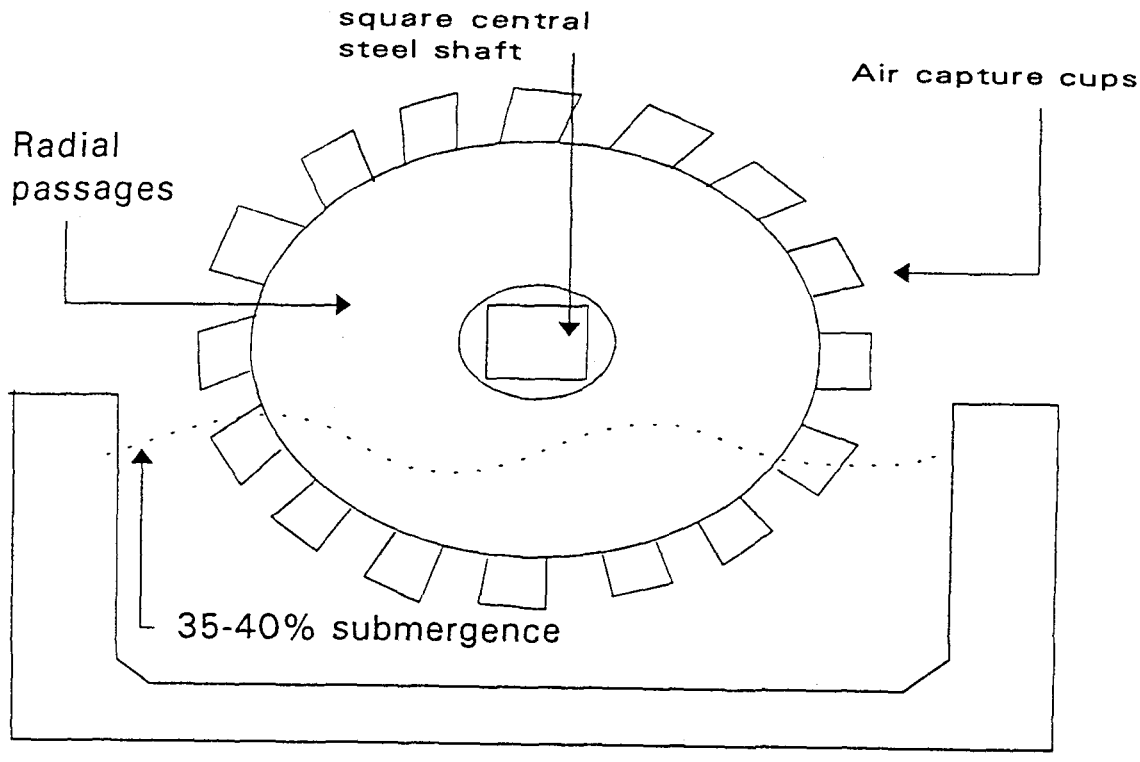


Fig 1.6. A schematic of a rotating Biological contactor (Ware, 1990).

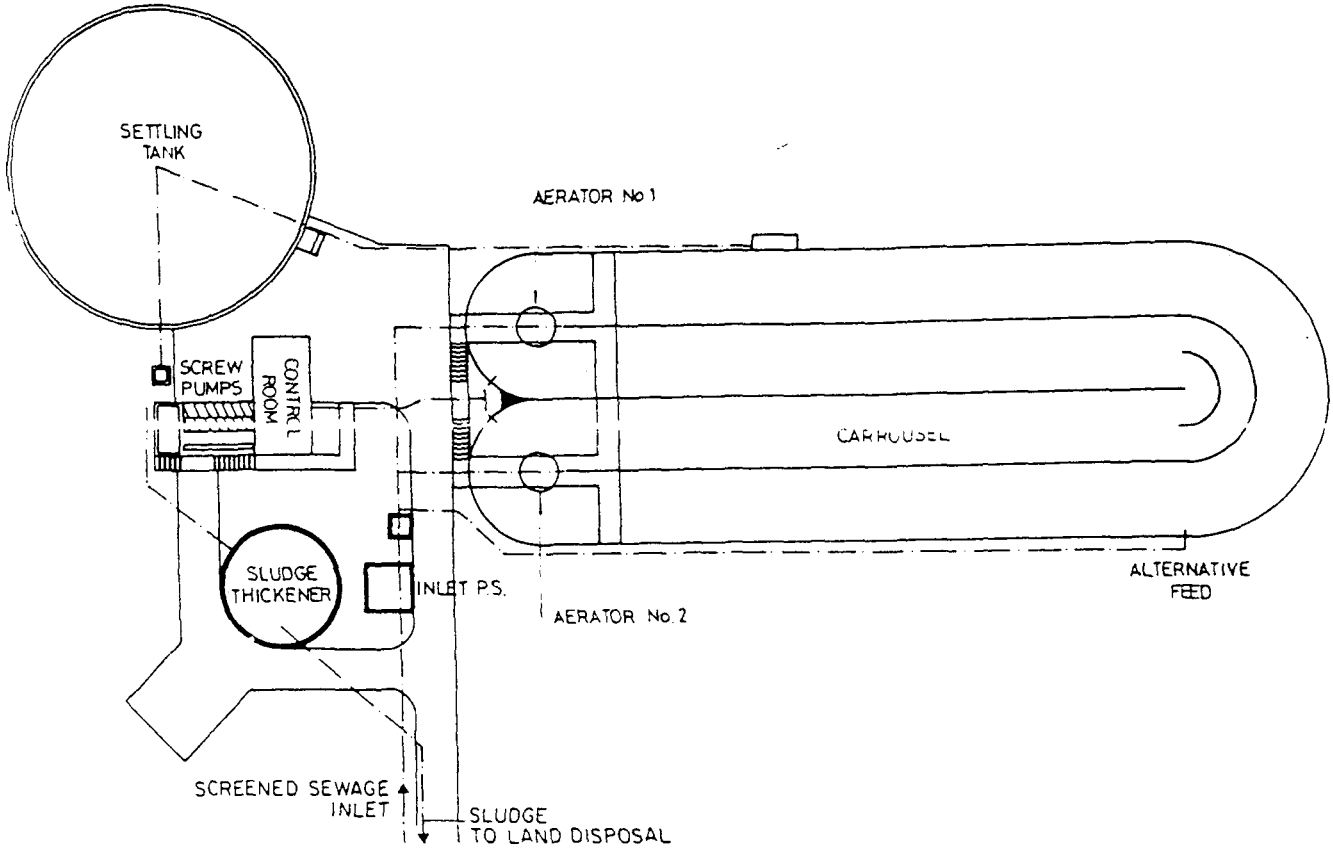


Fig 1.7. Layout of Ash Vale Carrousel treatment system. From Barnes et al, 1981.

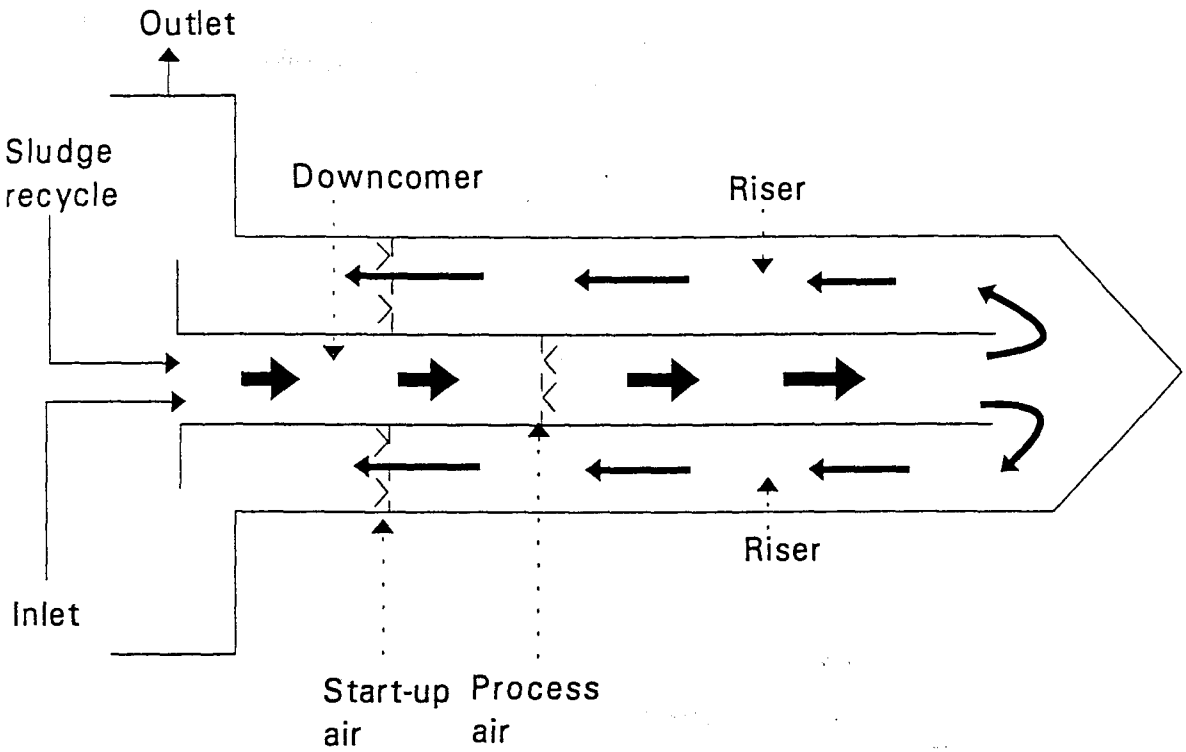


Fig 1.8. Ithe I.C.I Deepshaft process.
From K.D.Staples, 1978.

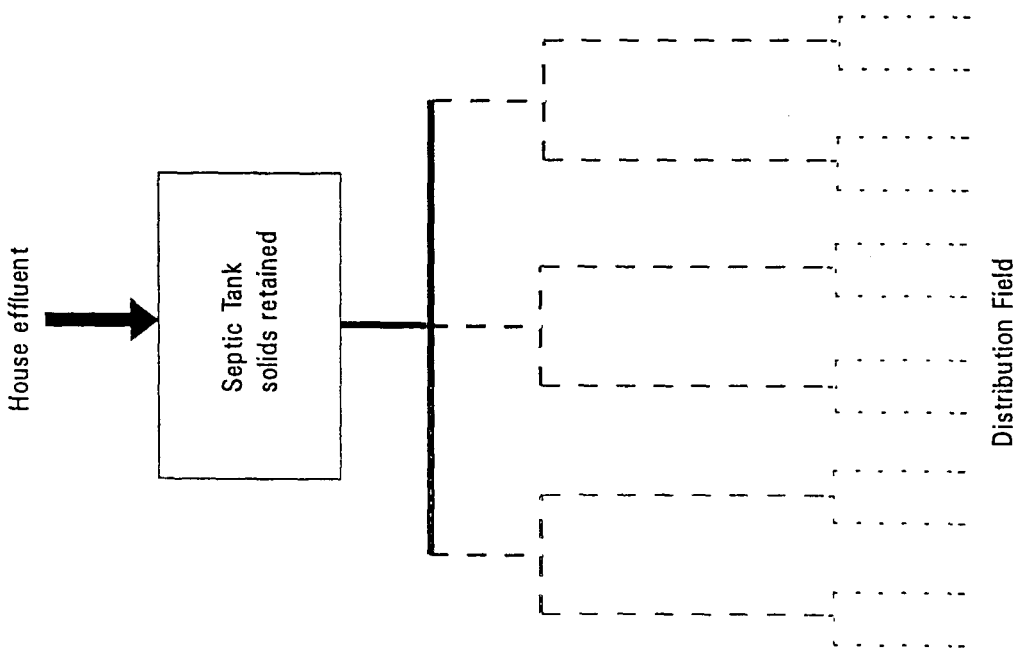


Fig 1.9. A schematic of a septic tank system. The disposal field drainpipe, is used to distribute the treated effluent. Solids are retained in the septic tank.

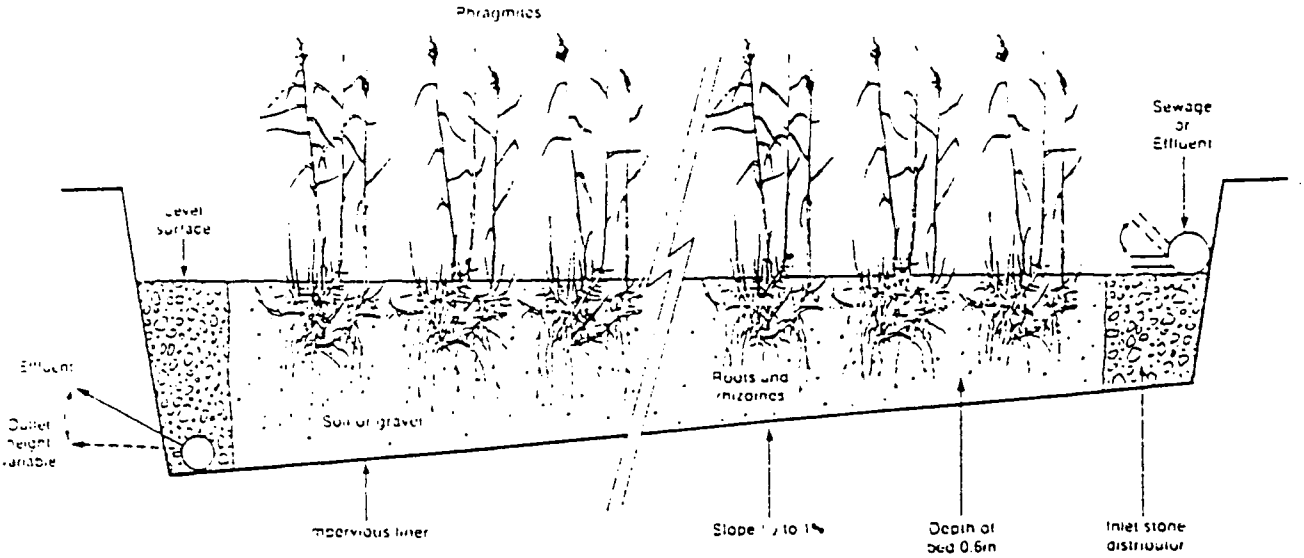


Fig 1.10. A typical arrangement for a reed bed treatment system (Taken from Cooper at al, 1989).

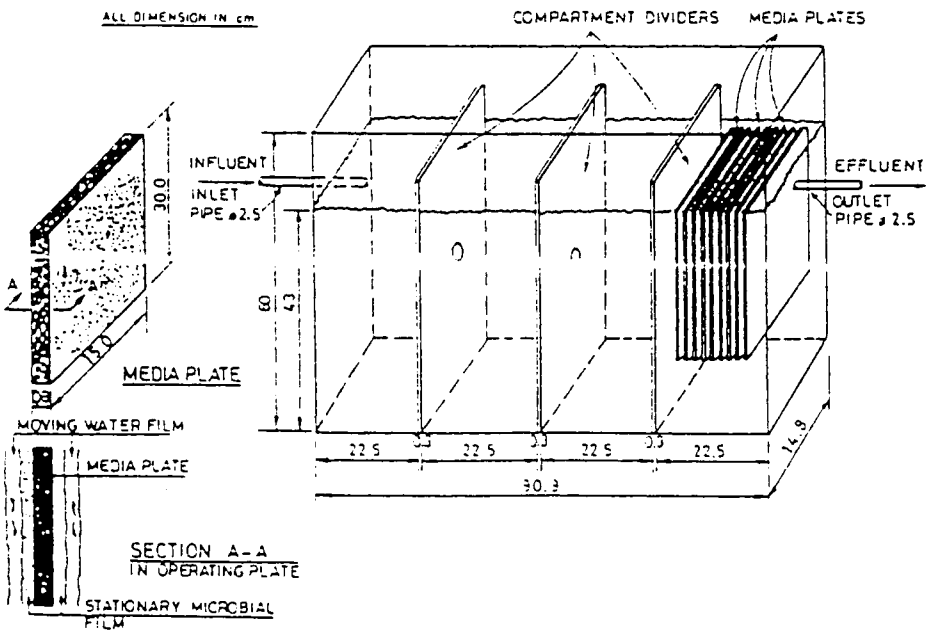


Fig 1.11. Details of an aerated submerged fixed film reactor. From Al Haddad, 1991.

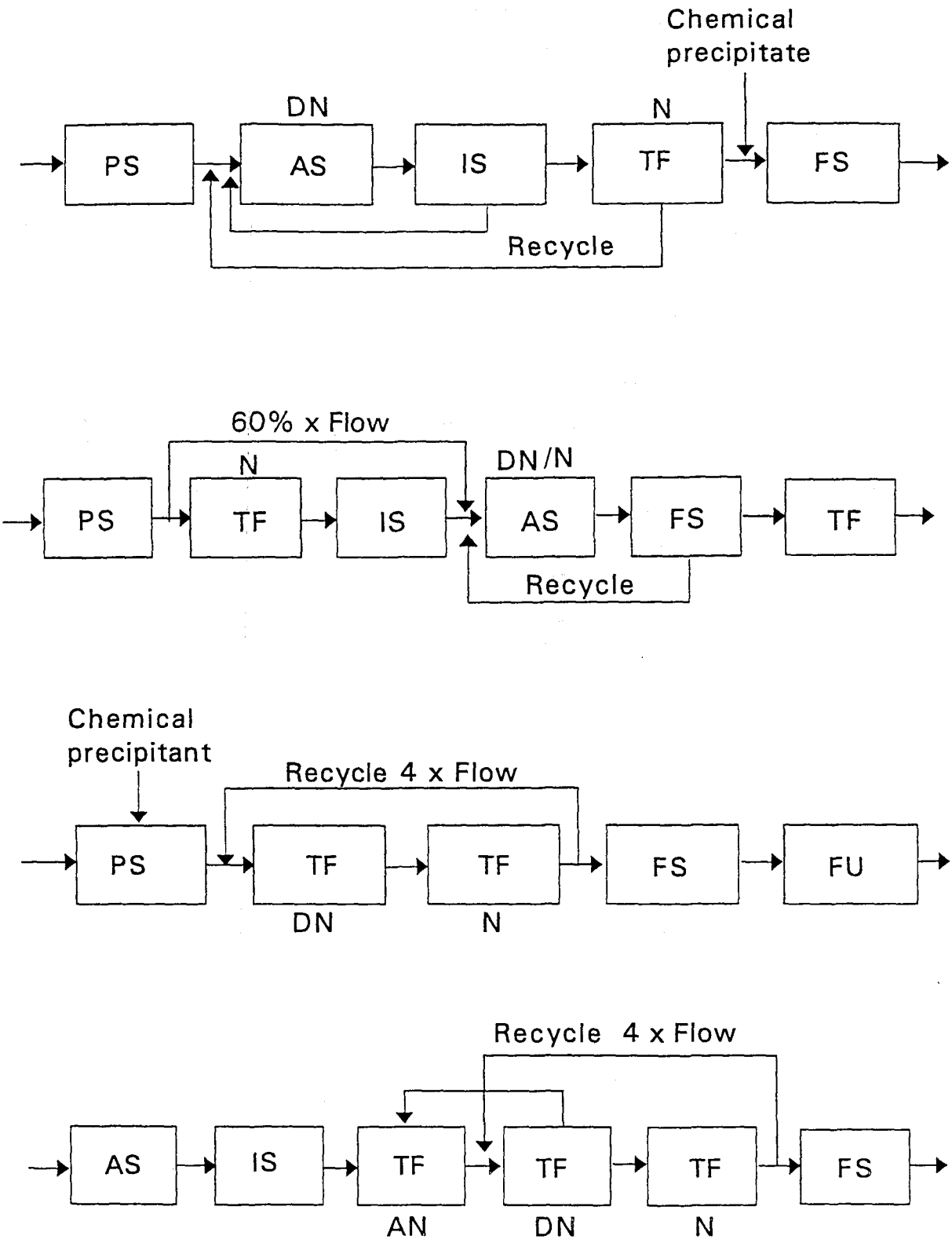


Fig 1.12. Various combined systems, from Nakajima, 1989. PS= Primary settling tank, AS= Activated sludge, IS= Intermediate settling tank, TF= Trickling filter, FS= Final settling tank, FU= Filtration unit, AN= Anaerobic, DN= Denitrification, N= Nitrification.

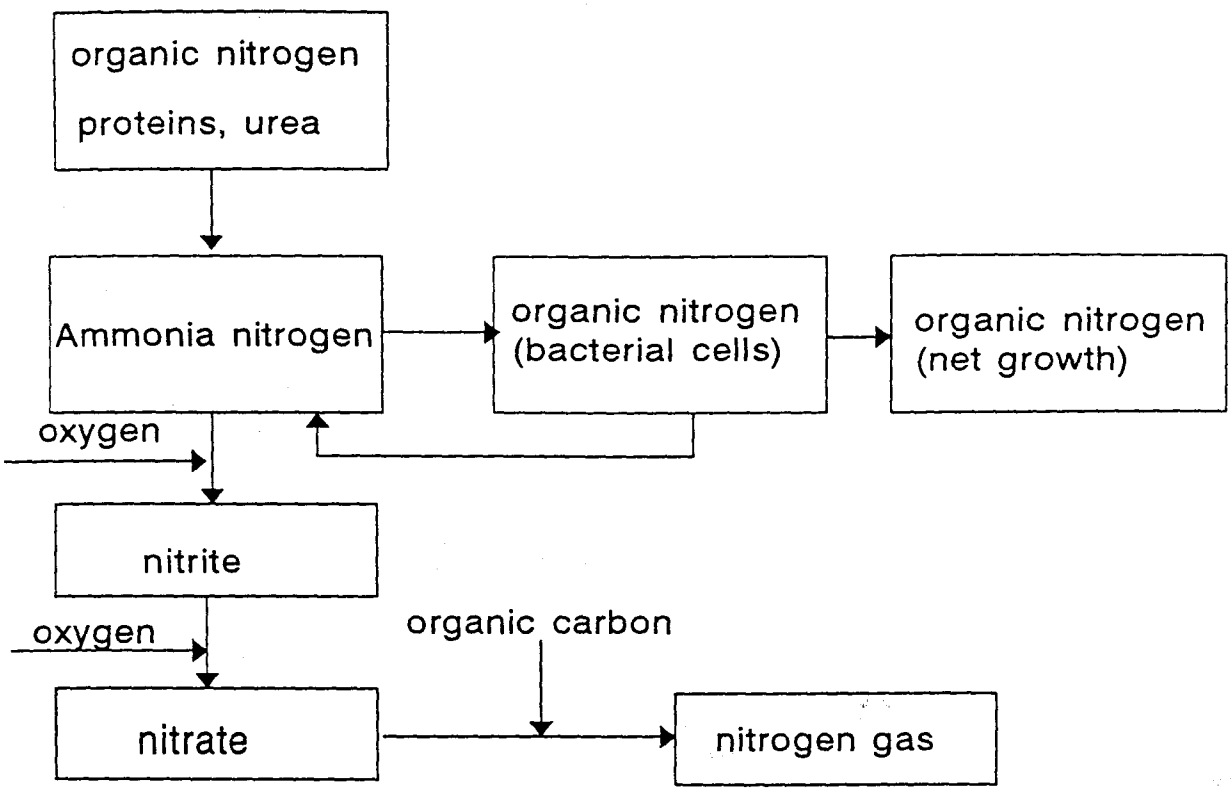


Fig 1.13 nitrogen transformation in biological treatment processes. From phosphorus and nitrogen removal, Sedlak, 1994.

CHAPTER 2. NITRIFIER INHIBITION DUE TO COMPETITIVE ANOXIA.

SUMMARY.

A prototype wind-powered plant (**WATER** Wind Assisted Treatment for Environmental Recovery) incorporating a bio-oxidation process as described by Janes and Thomas (1992) was subjected to field trials. The tests revealed dormancy in the nitrifying bacterial population if wind speeds of less than 3m s^{-1} persisted for periods of several hours (Shrimpton *et al,* 1993), this reduction being attributed to oxygen deficiencies due to inadequate aeration and agitation. As a basic study to assist with the understanding of this behaviour, a series of shake-flask and half-metre motor driven pilot plant tests were conducted to explore the association between nitrification inhibition and the surrounding BOD and dissolved oxygen (DO) concentrations. The shake-flask results detailed here demonstrate that the nitrifiers' dormancy is due to their inability to compete with heterotrophic bacteria under severe anoxic conditions. The findings were subsequently confirmed at 0.5m with segregation of the BOD and nitrification duties.

2.1. INTRODUCTION

Nitrification is a sensitive indicator of dissolved oxygen abundance because oxygen is required as the terminal electron acceptor by nitrifying bacteria. Nitrification can also be affected by a number of environmental factors, which include short bacterial retention time (compared to BOD removal), low temperature (10°C equal 10days for complete nitrification), a wide range of organic compounds (e.g., allythiourea and thiocarbonate),

and mixing intensity. This is clearly shown by Figs. 2.1 and 2.2, (from Stenstrom and Song, 1991) where, as mixing rate increases so does the reaction rate. Nitrifiers are also affected by extremes in pH, as shown in Fig. 2.3 (Prosser, 1989), where the growth rate can be seen to be significantly higher between pH 6.4-7.6 than outside these values. The impact of these factors on the two microbial populations in activated sludge (AS), the nitrifiers and the carbon-consuming heterotrophs, differs, due to the nitrifiers' lack of robustness (Painter, 1977). Nitrification in the AS process is also dependent on the sludge age, due to the lower specific growth rate of the nitrifiers (Metcalf and Eddy, 1994). Hence the nitrifying population needs time after periods of environmental stressing to re-establish itself before nitrification can restart.

A further factor affecting the nitrification rate is the organic loading rate, as organic carbon removal and nitrification often occur within one single reactor. The conditions within such a reactor, therefore, must be conducive to growth and activity of both heterotrophic and nitrifying bacteria. In operational terms nitrification can be expected only below a critical sludge loading rate of 0.15kg d^{-1} or a sludge age of greater than 4d (Goronszy, 1995). This poses a problem, for if sufficient time is provided for adequate nitrification, the sludge age may no longer be optimal for BOD removal. Even when both heterotrophic removal and nitrification do not occur in the same reactor, potential problems still exist. These centre on the requirement of the nitrifiers for carbon dioxide as their carbon source. This means that in a system involving two reactors in series, the amount of dissolved CO_2 present when the effluent enters the nitrification zone is vital. If it is insufficient then nitrification will suffer, due to carbon substrate limitation.

In practice, sensitivity problems with nitrification are more often interpreted as ammonia deficiency, more precisely (Hanaki *et al.*, 1990), that carbon consuming heterotrophs inhibit nitrifiers via competition for nitrogen (ammonia). Now ammonia rather than dissolved oxygen becomes the rate limiting substrate, so long as the dissolved oxygen level does not drop below 1% saturation level (Prosser, 1989). Below this value the dissolved oxygen will once again become the rate limiting substrate. Therefore in wind powered aeration, how the nitrifier population recovers following frequent and possibly extended periods of anoxia is of more importance. As there will be frequent cessation of aeration with wind powered treatment plant. There is seemingly an adequate understanding of re-aeration response of nitrifiers in pure culture (as described below) but not as well established perhaps in competitive mixed populations.

2.1.2 Nitrification..

The role of oxygen in nitrification and assimilation pathways as indicated below (Stryer, 1994) is crucial for *Nitrosomonas* and *Nitrobacter* species, both of whom utilise oxygen in their conversions.

N_2O

Nitrobacter | ***Nitrosomonas***

NO_3^- <----------> NO_2^- <----------> NH_4^+ <---------> AMINO ACIDS

nitrate *nitrite* assimilation

reductase *reductase*

The saturation constants for oxygen (K_s) in pure cultures of *Nitrosomonas* and *Nitrobacter* are about 0.25-2.5mg L^{-1} Dissolved oxygen (Painter, 1986) meaningfully higher (some 20%) than those for carbon consuming heterotrophs. This explains why they are unable to compete with heterotrophs for dissolved oxygen, when the levels are low (below 2mg L^{-1}, Prosser, 1989). Just how anoxia affects nitrifier metabolism is not that well understood (Prosser, 1989) although there is considerable interest in this area, such as that described in Laanbroek (1993; 1994) and Tanaki (1994). However, most of this work centres on the effects of competition between different nitrifier species rather than competition between mixed populations.

2.1.2 Oxygen Sensitivity.

The effects of Dissolved oxygen limitation have of course been documented previously, though, surprisingly this is normally in general terms. In the first place, as long ago as 1964, Downing typically reported "To study the effect of the anaerobic conditions to which sludge may be subjected at the bottom of the final settling tanks, a nitrifying activated sludge was kept anaerobic for 4h and was then re-aerated. Its nitrifying ability was zero for a few minutes after re-aeration began, but rose within 20 min to its original value. In practice the period during which sludge remains anaerobic is unlikely to be as long as 4h on average, and it is not thought that the above effect is likely to be of much importance in practice, except perhaps in very high-rate plants". Evidently earlier workers had noticed the effects of dissolved oxygen limitation on bacteria, but dismissed them.

Later work such as Hanaki *et al.* (1990) showed that nitrite oxidation proceeds much more slowly than ammonia oxidation with low amounts of Dissolved oxygen, suggesting that nitrite should accumulate. However, these "single step" interpretations do not fit with the specific ammonia oxidation rate ie per unit biomass being elevated at low levels of Dissolved oxygen such that the increased population compensates for the reduced ammonium oxidation rate/unit biomass. Whilst *Nitrobacter* species seemingly have this metabolic compensation system (Hanaki *et al.*, 1989) nitrite oxidisers do not, and so nitrite oxidation rates are significantly reduced by low dissolved oxygen (Gee *et al.*, 1990). Specifically their slow growth and poor energy utilisation (Laanbroek, 1993) compared with most other bacteria means that special consideration needs to be given to nitrification.

When the kinetics of ammonia oxidation within various systems are considered, there have been many reports published including those of Dipinto *et al.*, 1990; Wentzel *et al.* 1991. The majority of these use simplified and idealised models, typically taking the aeration vessel as being perfectly mixed, and sludge as being a single species, whose growth kinetics follow simple Monod kinetics (Bailey and Ollis, 1987). Some models do take into consideration such problems as substrate inhibition kinetics and solids recycle (Bertucco *et al.*, 1990) along with the possibility of denitrification occurring (Clayton *et al.*, 1991). However, as there are so many different models, varying in complexity and content, it would be impossible to cover them all here so apart from particular instances where problems arise that can easily be explained using them, the work will be presented without their reference.

The present work was prompted by early field tests of the wind powered treatment plant (reported by Shrimpton *et al.*, 1993) when amongst the difficulties encountered with wind intermittency and interpretation of the data, under performance in nitrification was clearly demonstrated. It was suggested that this sluggish metabolism and failure to compete with more active species during restart, was due to the very low dissolved oxygen. To understand and quantify these limitations the basic studies reported here were conducted, under the conditions of controlled intermittency, using both shake-flasks and small laboratory half metre tank. These studies had the specific aims of determining the recovery time of the nitrifiers in a mixed bacterial population, and the best tank configuration to minimise this oxygen limitation effect. The account is offered in four sections beginning with experimental methods, then results, discussion and conclusions. The results section will be split into two distinct sections, the first covering the identification of nitrification inhibition due to bacterial competition following anoxia (single tank tests), the second describing the results from our proposed solution (two tank tests).

2.2. EQUIPMENT, MATERIALS AND METHODS.

2.2.1 Intermittent Aeration Experiments (Shake-flask scale)

Intermittent aeration experiments were done in 500ml conical flasks containing activated sludge (30ml) to which primary screened sewage (270ml) was added producing a 10:1 ratio which is typical of conditions employed at many British sewage treatment plants

(Stanibridge, 1977). Starting ammonia concentration was then adjusted to approximately 80 mg L^{-1} by the addition of 1 molar $(NH4)_2CO_3$ solution (10ml). All flasks were placed in a rotary incubator (200 rev min^{-1}) and kept at a constant temperature of 20°C. The following protocols were followed:

Series 1 To assess time intervals for dissolved oxygen starvation of the bacterial population to cause dormancy and for renewed aeration to reactivate them.

Series 2 As above but using a pure culture of nitrifying bacteria obtained from International Biochemicals, Slough.

Series 3 As for series 1 but utilised a stressed bacterial population.

Specific protocols used in each series were as follows *Test 1*, one hour alternating aeration on:off; *Test 2* - three hour alternating aeration on:off; *Test 3* - five hours alternating aeration on:off. Oxygen starvation was achieved by flushing the head space with air-free nitrogen for 1min and stoppering. The aerobic control was maintained unstoppered and shaken continuously. At the end of 5 cycles the regime was switched to continuous aeration for 20h, resulting in overall exposure to intermittent aeration. The continuous aeration provided, proved to be sufficient to reduce ammonia to 10mg L^{-1} or less, in line with regulatory constraints.

2.2.2 Recovery Times

The above tests were also used to note the effects of competition between the nitrifiers and heterotrophic organisms were assessed by testing with a range of BOD loadings again using 500ml conical shake-flasks. The flasks were filled with activated sludge (30ml), as the bacterial source, primary screened sewage (270ml) and the ammonia concentration was again adjusted using the addition of 1 molar $(NH_4)_2CO_3$ solution (10ml). Incubation also followed the same method as for the intermittent aeration tests, except that different time intervals were used. In the current tests a starting aeration period of 5h (to allow equilibration of bacteria), followed by two starvation periods of 12h, separated by a 7-8h aeration period to allow the nitrifiers to re-activate. Sampling was done at the end of each period and hourly during the aerated periods.

The tests with AS were conducted in two phases; the first using a sample obtained from Minworth Sewage Works (STW) and the second with ammonia-conditioned AS as described below. Triplicate dry weight measurement of the first AS sample was adopted to standardise sludge solids content (4% v/v) in all studies. In addition to these tests, pure laboratory cultures of *Nitrosomonas* and *Nitrobacter* (Biolyte NC200, International Biochemicals Ltd, Slough) were evaluated as being a 1:1 mixed population at 10:1 dilution corresponding to the nitrogen biomass loading of AS, delivering oxidation rates of between 30-40 mg L^{-1} h^{-1} ammonia conversion to nitrate. The ammonia conditioned AS was prepared using 10L in a 12L tank with exposure to ammonia over several weeks increasing from 35mg L^{-1} to 500mg L^{-1}. Increases were daily and involved 10mg L^{-1}

step increases in ammonia concentration. Primary screened sewage was added (1L, of 180-250mg L^{-1}) daily to the aerated (using air sparger) tank, vis pouring so as to maintain an approximately constant heterotrophic bacterial population (measured via allythiourea inhibited BOD measurements). Our purpose here was to boost the nitrifier population, by using a favourable environment.

2.2.2 Stirred Aerated tank

The configuration and operation of the stirred aerated tank described by Janes and Thomas (1992) is shown schematically in Fig. 2.4a whilst 2.4b shows the two stage variant employed here. The tank is 500mm diameter and 330mm deep. The snorkel tube agitator aerators are 170mm by 10mm diameter with a perforated back face. Two or four of these were used symmetrically and deployed in a radial rotation supported on a cross beam from a central shaft, which in turn was powered by a half metre Vertical Axis Wind Turbine (VAWT, see Janes and Thomas, 1992 for details). This was driven by fan-supplied wind at up to 20-25 rev min^{-1}. Primary screened sewage (40L) and activated sludge (4L) were introduced to the tank (10:1 dilution). A control located in the wind path comprised a circular tank containing 15L of primary screened sewage and 1.5L of activated sludge (10:1 dilution again) in which aeration was derived only by surface agitation. In both cases vapour loss by evaporation was replaced with distilled water.

2.3.5 Compartmentalisation Half-metre tests..

In these tests the original tank was re-configured to assist the nitrifiers in competition for the depleted dissolved oxygen. The inner tank (diameter 330mm, depth 220mm) is hydraulically connected to the outer annulus by 12 equispaced 5mm diameter holes located 20mm below the rim. As shown in Fig 4b the inlet and outlet pipes were as described previously, however, in the new configuration three 12mm diameter pipes were also present for sludge return to the outer annulus. This would prevent a build-up of heterotrophic flocs in the inner annulus thus slowing nitrification.

As for the single stage tank this unit was provided with 40L of primary screened sewage introduced initially to the outer annulus and aerated by two "snorkel" tube aerators before transfer to the inner tank, where it was also aerated by two "snorkel" aerators (Fig 4b). The aerators were powered initially by a fan driven VAWT, but this was later replaced by a motor for ease of operation. This system was subjected to four aeration periods, which varied in length from 9 to 10h, except the initial stabilisation period which was 12h, non-aerated periods were 12h. Both batch and semi-continuous operation were adopted in this study and the later is reported in detail later.

2.3.6 Analysis.

In all of the tests samples of 25ml, in the case of the shake-flasks, and 100ml for the half metre unit were taken at 3h intervals during the aeration period at the start and finish of the oxygen starvation period and tested for BOD, ammonia, suspended solids (SS),

nitrite and nitrate. BOD was estimated by the Winkler method (Winkler, 1982), where pre-aerated, and buffered water (247ml) has a sample (3ml) from the test placed in duplicate BOD bottles (250ml). One of these is titrated immediately to give the initial dissolved oxygen concentration, the other is incubated (at 25°C) in the dark for five days, before it is again titrated for dissolved oxygen. The difference between these titrations is equivalent to the samples' BOD concentration. Ammonia, nitrate and nitrite were tested using Quantofix Test Strips (Camlab), which are sensitive down to 1mg L^{-1} (nitrite) and 5mg L^{-1} (ammonia and nitrate). The sensitivity on the ammonia test strips was increased by the use of standard ammonia solutions of 2.5mg L^{-1}, 7.5mg L^{-1}, 15mg L^{-1} and 30mg L^{-1}, into which strips from each new batch were dipped to provided a new colour scale. The same protocol was used for the nitrate/nitrite strips to increase their sensitivity.

2.4. RESULTS FROM SINGLE TANK TESTS.

2.4.1 Shake-Flask Tests.

Fig 2.5 shows the time course of ammonia and BOD concentrations for intermittent aeration periods of 1h compared with continuous aeration. With intermittent aeration the nitrification rate is reduced from around 8mg ammonia L^{-1} h^{-1} oxidised for continuous aeration to about 5mg L^{-1} h^{-1} (from the gradient of the line). Below an ammonium concentration of about 10mg L^{-1} the removal rate dropped appreciably. This is consistent with the expectation that ammonia is becoming the limiting factor instead

of oxygen. BOD concentration also fell steadily at 15mg L^{-1} h^{-1}, reflecting only a slight reduction (from 17mg L^{-1} h^{-1}) from that obtained without Dissolved oxygen limitation. Allythiourea (a nitrifying bacteria inhibitor) had little effect on the BOD reduction rate and we note the lowest prcticable level to which the BOD could be reduced (35mg L^{-1}) was the same with and without any allylthiourea. It is expected that the residue itself would be removed if the experiment would have run for an extended period.

Fig 2.6 shows results of a parallel test; similar to the last except the intervals of 3h on: 3h off were longer. We can now see that following 3h without shaking a full 3h of aerobicdisolved oxygen supply still failed to restore full activity. In contrast the BOD fell steadily, albeit at a reduced rate of 8mg L^{-1} h^{-1} compared to 15mg L^{-1} h^{-1} achieved with continuous aeration. Clearly the heterotrophic organisms have a higher affinity for the dissolved oxygen in line with literature expectations (Metcalf and Eddy, 1992) and with 3h cycling there is insufficient dissolved oxygen to leave any residue for the nitrifiers. In the present tests we noticed that nitrifiers activity in fact persists over the first 3 cycles (18h) before halting with residual ammonia at 35mg l^{-1}. Preliminary tests showed that nitrifier dormancy can be reversed by exposing the bacteria to aerated conditions for longer periods (4-6h), and to investigate this more fully, we undertook an extended aeration test (shown by ammonia trace b). In this the aeration was maintained after 15h, and the results clearly show that the ammonia oxidation and hence nitrifier activity was restored to 4mg L^{-1} h^{-1}, giving a final value of 10mg L^{-1}.



The crucial issue here is the maximum period of restored aeration required to restore nitrifying activity after prolonged anoxia. As can be seen in Fig 2.6 ammoniacal oxidation rates were not fully renewed within a further short interval of restored aeration although oxidation did continue at a diminished rate throughout the remainder (aerated periods) of the test. The ammonia oxidation rate decreased from 6mg L^{-1} h^{-1} to around 3mg L^{-1} h^{-1} following the first two starvation periods (60% of that obtained in 1h cycles). However, we noticed that during this period (greater than 15h from start) the BOD levels were also reduced and a further issue is the extent to which competition hindered this recovery we return to this point later.

Fig 2.7 shows time traces of ammonia and BOD during intermittent aeration imposed over a 5h (rather than 3h) periods. We can see for both ammonia and BOD, removal occurred essentially only during aeration. When the air was off, ammonia levels did not drop at all, and BOD actually increased. On resumption of aeration, oxidation of BOD and of ammonia immediately restarted. This behavior was similar to that in the study with 3h alternating periods, when BOD levels also consistently dropped during aeration, in cotrast to the slight increases found during the unaerated periods, and it is suggested that this drop is perhaps attributed to aerobic bacterial death and decomposition with consequent cell lysis. The very slight increase in BOD degradation rate when allylthiourea is present inhibiting the nitrifiers is therefore readily explained on the basis that all of the available dissolved oxygen under these conditions can now be utilised for BOD breakdown.

Fig 2.8 continues the survey with intermittency intervals extended to 12h. Such a period more realistically approaches a downtime for a slack condition over the summer months. The actual intermittent periods were 12h off followed by 7h of aeration (sufficient for ammonia oxidation), a further 12h starvation followed by 9h aeration, to allow complete treatment of effluent. The trend is broadly as expected from the preceding accounts and we can see clearlyt a 4h or so delay interval prior to resumption of nitrifier activity following the restored aeration. The reader will remember that one of the main objectives was to maintain nitrification even under limited dissolved oxygen conditions. The evidence of nitrification following anoxic periods is therefore most encouraging, as it provides evidence that the nitrifiers remain viable throughout the test (with ammonia reduction rates of around 4mg L^{-1} h^{-1}). This results in ammonia reduction to 5mg L^{-1} by 40h, after which lack of ammonia became the limiting factor.

The records for BOD degradation (Fig 2.8) also show steady and recoverable activity during periods of aeration, and again a slight increase in the absence of aeration. Similar results with and without allylthiourea also again indicates that the nitrifiers cannot compete for intial dissolved oxygen, due to their lower affinity for oxygen compared to the heterotrophs. BOD reduction rates here (of around 15-17mg L^{-1} h^{-1}) compare favourably with the expectations of published literature (Bailey and Ollis, 1988; Metcalf and Eddy, 1992) indicating that the supply of oxygen from the aerators amply meets the biological demand.

2.4.2 Half-metre tank tests: one compartment.

Having established a broadly understandable picture (the shake-flasks tests) we now consider how well our findings relate to a relevant small-scale configuration. It should be noted that in previous work (Fowler, 1994) the aerators were shown to deliver consistently 1Kg O_2 KWh^{-1}, which is comparable with the 0.75Kg O_2 required to oxidise 1Kg BOD (Bailey and Ollis, 1988). Fig 2.9 shows BOD and ammonia concentrations against time for the half-metre pilot plant model and the unagitated control (to check for surface aeration effects) under constant aeration conditions (four snorkel aerators rotated at 70-85rpm). Clearly, such constant aeration conditions in the tank suffice to reduce BOD and ammonia to well under the discharge consent levels. In particular the BOD level was reduced from 275 to 20mg L^{-1}, (92%), whereas with the control it was only reduced to 150mg L^{-1} (36%). We note also that the majority of the oxidation occurred in the first 8h.

Ammonia oxidation (Fig 2.9) took the concentration down from 70mg L^{-1} to 10mg L^{-1} in the first 20h, (a reduction of 85%) and a further reduction, effectively to zero in the next 25h. On the other hand the control (a tank containing 10:1 dilution of sewage and activated sludge, placed in the wind path) showed a reduction from 70mg L^{-1} to approximately 55mg L^{-1} in 45h; only a 20% decrease, and this occurred mainly in the first 6-7h. In general terms, the events in our tank are broadly consistent with the previous shake-flask work.

Fig 2.10 shows the ammonia and BOD oxidation time profiles with intermittent aeration; this test involved alternating 6h periods on and off. Both time profiles indicate that significant oxidation occurred only during the aerated periods. During each aeration period, nitrification occurred at a rate of 2mg L^{-1} h^{-1}, until towards the end of the test when it slowed, perhaps due to ammonia limitation. We note that the rate of ammonia oxidation here is 50% lower than that obtained previously with continuous aeration. Over the entire period of the test; that is 60h, BOD was reduced from 410 to 50mg L^{-1}, an overall reduction rate of 12mg L^{-1} h^{-1}, which we note is considerably higher than the 7mg L^{-1} h^{-1} delivered by the continuous aeration system. The reason for the increase here is unclear, but is perhaps due to the action of other bacteria.

2.5 RESULTS: TWO TANK TESTS.

2.5.1 Shake flask trials.

According to literature data (Downing, 1964; Laanbroek, 1994) and as indicated by our own earlier work it would seem that nitrification is inhibited by high carbon loadings, in other words nitrifiers cannot compete successfully against metabolising heterotrophic organisms for dissolved oxygen. Various tests outlined below were directed to quantify and characterise this behaviour, also to establish recovery times for nitrifying organisms following various periods of oxygen starvation. They were conducted as four separate trials using four different initial BOD loadings (280mg L^{-1}, 140mg L^{-1}, 70mg L^{-1} and 0mg L^{-1}). Three different population sources activated sludge, ammonia stressed

activated sludge, and a pure nitrifier population (Biolyte NC2000) were used, but with the same intermittency periods (12h off 5h on). The stressed activated sludge population was produced by placing municipal activated sludge in an aerated tank for several weeks, during which time the background ammonia level was progressively increased.

(A) Activated sludge

Fig 2.11 shows four time profiles corresponding to initial BOD strengths of 280mg L^{-1}, 140mg L^{-1}, 70mg L^{-1} and 0mg L^{-1}, for an aeration regime of 5h on and 12h off, since previous results have suggested 5h on as sufficient for recovery. It is clear from the graph that there was no ammonia oxidation during the starvation periods. Moreover an examination at the end of these starvation periods starting at 17h and 35h clearly indicates that higher BOD levels inhibit resumption of nitrification. At the end of the 40h test, however, the differences in the amounts of ammonia remaining were small ranging from 10 to 15mg L^{-1}, albeit with the highest relating to the highest initial BOD loading. Effective mean values for nitrification ranged from 3.4 to 4.4mg L^{-1} h^{-1}, a 20% reduction at the higher initial loading. It is worth noting that in the absence of BOD, nitrification appeared to lag slightly behind the profile corresponding to an initial BOD of 70mg L^{-1}. This is as might be expected due to the carbon dioxide requirement of the nitrifiers (DiPinto, 1990).

Time profiles for nitrite and nitrate concentrations showed that nitrate production clearly lagged behind nitrite production, again in line with expectations (Metcalf and Eddy,

1992). Over the interval 17-22h from start-up, there is an indication that nitrite concentrations are higher at the lower BOD values, and that finding here is consistent with the reductions in ammonia (Fig 2.11) mentioned earlier. Nitrifiers need carbon dioxide as carbon-source (DiPinto, 1990) and whilst BOD is inhibitory for the reasons already mentioned, complete absence will provide no Carbon Dioxide. These results therefore also confirm that there must be an optimum BOD level above zero for effective nitrification. One conclusion can immediately be drawn from these findings regarding the inhibiting effects of BOD when nitrifying activity is resumed, following starvation. At a BOD concentration of 70mg L^{-1} it is clear that nitrification was resumed immediately aeration re-commenced following each of the starvation periods. However, at the higher BOD values, namely 140-280mg L^{-1} it is equally clear that nitrification was time-delayed by perhaps 1-2h in both cases, which is consistent with published literature (Laanbroek, 1994).

Ammonia stressed activated sludge:

Fig 2.12 shows the ammonia concentration time profiles for the "pre-stressed" population. The initial nitrification rates for the first 5h interval increased from 4 to 5mg L^{-1} h^{-1} for a BOD loading of 140mg L^{-1} and from 6 to 7mg L^{-1} h^{-1} for the lower BOD loading of 70mg L^{-1}. In contrast, the nitrification rate during aeratio;, that is the period 17-22h; was approximately 5mg L^{-1} h^{-1} in all four cases, though we note with zero BOD there appears to be a short lag of 1h or so before ammonia oxidation resumes. Futher evidence for such a lag arises from observations of the final period of resumed aeration starting at 34h. Here, lag periods of some 2h are evident in each case, presumably as the BOD, and thus CO_2 levels drop below the necessary minimum. Overall, then, we have a

broadly semi-quantitative picture emerging which shows that nitrifiers cannot compete successfully for oxygen with heterotrophic organisms at higher BOD levels, but equally that the nitrifiers require a minimum level of carbon dioxide substrate. These conclusions relate specifically to the resumption of aeration following periods of anoxia.

Pure culture:

In view of the improved performance with conditioned culture, it is of interest to ask what performance is achieved with pure culture and Fig 2.13 shows the results. A regime comparable with the previous population tests was employed with starting ammonia 75mg L^{-1}, with initial periods of aeration 4h followed by a 12h starvation period and then 5h aeration, 12h starvation and aeration until the end, again for the four BOD values 280, 140, 70 and 0mg L^{-1}. We see that ammonia oxidation ceases during the periods of anoxia and this is consistent with our previous findings. However in the case of the pure culture there were significant differences in repect of the lags noted previously. Thus, in the present series of tests, lags were noted only with the BOD of 280mg L^{-1}, the highest value tested, following resumption of aeration at 16h and 34h. Final concentrations of ammonia showed no significant differences for the three lowest BOD loadings but the residual level at the highest BOD loading was a little higher at 20mg L^{-1}.

Overall nitrification rate, that is nitrification averaged over the aeration periods ranged from 4.6 to 5.8mg L^{-1} h^{-1}, the higher values relating to the lower initial BOD loadings. We note that this range of values is about 20% higher than that obtained from the

stressed population. Although far from clear cut there are again indications that the absence of BOD does appear to give rise to slightly reduced nitrification (7.5mg L^{-1} h^{-1}) about 25% less than the 10mg L^{-1} h^{-1} obtained at higher BOD loads. This is again consistant with the nitrifiers requirement for carbon dioxide as their carbon substrate. However, the subsequent periods of aeration showed no significant variations although slope estimates indicate nitrification rates ranging from 5mg L^{-1} h^{-1} at 280mg L^{-1} initial BOD to 8.3mg L^{-1} h^{-1} for initial BOD 70mg L^{-1}.

Recovery lag following anoxia:

In view of the important implications for wind powered aeration plant, our central practical concern here, we pursued the issue of recovery periods in slightly more detail as follows. Fig 2.14 shows the recovery time versus initial BOD loading recovery time here defined as the time for noticeable ammonia oxidation from the system following starvation. The graph also shows the recovery time for ammonia stressed AS and pure culture along with normal municipal activated sludge, after one and two starvation periods. There is an indication from the graph that modest BOD (70mg L^{-1}) is associated with shorter recovery times than either zero BOD or 140mg L^{-1} initial BOD although perhaps the most that can be said with confidence is that BOD values in excess of 140mg L^{-1} cause increased lag times of around 1h for BOD's up to 280mg L^{-1}.

The results for pure culture are again consistent with the concept that competition for dissolved oxygen from heterotrophic organisms can extend the lag time for nitrifier recovery. Thus, with pure culture the lag times are reduced; indeed they are only

significant for the highest BOD loading (280mg L^{-1}). Our findings here thus lead us to an overall conclusion that is broadly consistent with established practice for water treatment plant; namely that two-stage oxidation is desirable. The first stage should reduce BOD levels but not eliminate BOD altogether (down to approximately 100mg L^{-1}), While the second stage is for nitrification (Metcalf and Eddy, 1992; DiPinto, 1994). With this in mind we now proceed to describe the results from the half metre pilot plant work.

2.5.2. Half-metre pilot-plant tests: two compartments

Following the shake-flask results indicating an optimal BOD range for the resumption of nitrification after anoxia (wind intermittency), the configuration of the half-metre pilot plant unit was altered (Fig 2.4b) to incorporate two zones; one for BOD removal (outer annulus) the other for nitrification (inner annulus). pilot-plant performance was then tested using the protocol described in the materials and methods, with both constant and intermittent aeration.

2.5.2.1. Constant aeration.

Fig 2.15 shows the time profile of ammonia oxidation, starting value 120mg L^{-1} over a 75h period involving batch transfer from the outer annulus to the inner zone after 26h and 52h, with constant aeration. Fig 2.15 shows that very limited ammonia oxidation occurs in the outer tank over the first period of 26h. However, after transfer to the inner annulus the nitrification rate increases considerably and significantly outperforms that in the outer annulus. Quantifying this, in the first period in the outer annulus the

nitrification rate was 0.5mg L^{-1} h^{-1} as compared with the 0.25mg L^{-1} h^{-1} from the control.

Referring to Fig 2.16 we see that over this period the BOD had been reduced to around 75mg L^{-1} which is therefore the starting BOD for the second period oxidation in the inner annulus following transfer. As demonstrated in Fig 2.15 there is a striking increase in the nitrification rate jumping to 3.6mg L^{-1} h^{-1} in the second interval (a decrease of 95mg L^{-1}) as compared with 0.5mg L^{-1} h^{-1} in the first interval in the outer tank. During the period 26h to 52h in the outer annulus the nitrification rate was again severely inhibited (rate 0.5mg L^{-1} h^{-1}) consistent with the first 26h. Ammonia oxidation during the second interval from 52h mirrored this behaviour. Overall the nitrification rates as a ratio between the inner and outer tanks was approximately 8:1 in both tests.

Fig 2.16 also shows the BOD time profiles for the two stage tank with sample transfers at 26 and 52h as in the earlier example. As can be seen from the graph BOD decreased from the initial 280mg L^{-1}, to 96mg L^{-1} between zero and 26h representing an oxidation rate of 7.3mg L^{-1} h^{-1}. Following transfer to the inner tank the oxidation rate decreased to 2.7mg L^{-1} h^{-1} with BOD going from 90 to approximately 30mg L^{-1} during that 26h period. During this second interval the oxidation rate in the outer tank was comparable with the 7.3mg L^{-1} h^{-1} from the first; similarly following transfer at 52h the oxidation rate in the inner tank was comparable with the first at approximately 2.7mg L^{-1} h^{-1}. In contrast to these values, the oxidation rate in the control in which dissolved oxygen was only replenished from the surface, since there was no aeration, was much smaller. Here, the ammonia concentration declined from 280mg L^{-1} down to 180mg L^{-1} over the entire

test period of 75h. Comparing Fig 2.16 with 2.17 we see clearly now that the two zone tank demonstrates quite strikingly the advantages of two stage oxidation. In other words, in the first tank initial BOD is reduced to less than 100mg L^{-1} while in the second stage ammonia is oxidised, going from 110 to approximately 10mg L^{-1} while BOD is further reduced from 100 down to around 30mg L^{-1}.

2.5.2.2. Two zone tank with intermittent aeration.

Fig 2.17 shows the time profiles for ammonia oxidation in the two zone tank for intermittent aeration at 12h intervals. Samples were transfered from outer to inner anulus at 29h. As expected from the earlier work in shake-flasks and half metre unit, ammonia oxidation occurred more rapidly in the inner tank where the BOD load was low following starvation periods. The lag period before resumption of nitrification in the inner annulus was 2h following the first anoxic period and was instantaneous following the second. Similarly, the nitrification rate for the inner tank, at 5.5mg L^{-1} h^{-1}, was considerably greater than the 0.5mg L^{-1} h^{-1} achieved in the outer annulus.

Fig 2.18 shows companion time profiles for BOD reduction. As expected from earlier work, the BOD in the inner tank was always lower than the outer tank, typically 100mg L^{-1} or so. Comparison with Fig 2.17 suggests that, within the inner tank, heterotrophic organisms were not present in sufficient numbers effectively to continue BOD removal. In the outer tank however, the BOD removal is substantial. The average BOD reduction rates for both tanks can also be calculated from Fig 2.8, at 10.5mg L^{-1} h^{-1} outer and 2.6mg L^{-1} h^{-1} inner. In all cases during the non-aerated periods the BOD

concentration showed a slight increase in line with our earlier findings and which we attributed to aerobic bacterial death and lysis.

The above results clearly show that BOD reduction and ammonia oxidation are strongly linked during periods following prolonged non-aeration (anoxia), explicable on the basis that nitrifiers cannot compete with heterotrophic organisms for the first available oxygen. This was overcome by splitting the treatment tank into two distinct zones, one for each species, and controlling the concentration of BOD that transferred between them. However, the system itself was run etentially batchwise and ideally the system should be run as a more semi-continuous operation. This was, indeed, attempted, but in this case, there was so much carryover of BOD into the inner zone due to the amount of agitation produced by the snorkel aerators that ammonia oxidation in the inner tank was reduced by 50%, and this problem was therefore deferred, being considered in more detail in the next chapter.

2.6. DISCUSSION.

Clearly there are two factors potentially limiting performance of such treatment plants as we are considering here. First, there is the question of anoxia associated with intermittent aeration. Secondly there is the question of competition for dissolved oxygen from heterotrophic organisms. A major difficulty is that these two problems are linked. Thus, in the presence of competition for dissolved oxygen, the nitrifiers are

effectively inactivated not only in the periods of anoxia but also subsequently following resumed aeration.

The most obvious solution to this problem is to incorporate a sloid matrix. This would then retain the heterotrophs as fixed film, and this, in turn, would ensure that the concentration of dissolved BOD is sufficiently reduced before transfer. Development of this concept is dealt with in more detail in the next chapter. On the other hand providing there is no significant competition for oxygen from the heterotrophs and that they have evloved sufficient carbon dioxide substrate for biomass continuity the nitrifiers seem capable of resuming their activity almost immediately following resumed aeration. The findings which support our broad conclusions here have been shown to be consistent between shake-flasks conducted under controlled conditions and pilot scale half metre tank tests in which the aeration was supplied by mechanical agitator aerators powered by either a motor or fan driven wind turbine.

Representative time scales for these effects would be 2h or so without aeration prior to collapse of activity of the nitrifiers and a comparable time scale for their recovery from dormancy following the resumption of aeration. There appears to be little sensitivity to the duration of anoxic periods up to about 36-48h after which significant cell death occurs. It is, however, important to compare the percentage of ammonia oxidised in the WATER system (85-85% in the compartmentalised batch system) with that in other waste water treatment systems, such as the activated sludge, 85-92% (Nakajima, 1989), and the "Carrousel" process 85-90% (Rachwal *et al.*, 1983). This shows that their

percentage oxidation is slightly higher than the figures we obtained in the WATER system, a reflection, perhaps, of the more intensive electric-powered aeration system.

Whilst sequencing batch operations do have their protagonists, it is never the less true to say that most practical treatment works operate continuously, or at least for semi-continuous cycles. In this connection, one obstacle to the process configuration as evaluated here is clearly sludge carryover between the BOD and ammonia oxidation zones. Of course, continuous inflow would in turn, imply increased loading of the first stage. The residence time would therefore have to be increased to accommodate this extra duty. Our test reported here has therefore not provided a fair example of the plants potential in that respect.

On the other hand, were the plant continuous, the continuous supply of liquor would carry to the inner tank considerable amounts of dissolved BOD and suspended solids (probably as sludge flocs). These are clearly both going to be deletarious to the nitrification which is intended to take place in the inner zone.

2.7 CONCLUSIONS.

It is clear from the initial shake-flask work that reduction of the BOD concentration to approximately 100mg L^{-1} or less is needed for our process is to be used under intermittent aeration conditions, such as those provided by wind power. Both at shake-

flask and at half-metre scale, it has been concluded that nitrifiers are unable to compete with heterotrophs for oxygen. The solution was to re-configur the system as outlined in 2.5.2 to incorporate two distinct zones, one for BOD reduction, the other for ammonia oxidation. Moreover, if the plant is to be continuous, it is necessary to have some method of limiting BOD to not greater than 100mg L^{-1}, and of retaining biomass in the outer annulus, preventing floc-transfer. The simplest solution seems to be to incorporate a solid matrix. So long as the BOD concentration entering the inner ammonia oxidation zone is not above 100mg L^{-1}, it is concluded that this system provides a favourable environment for nitrifier activity and should ensure that overall plant performance is not limited by lack of nitrification.

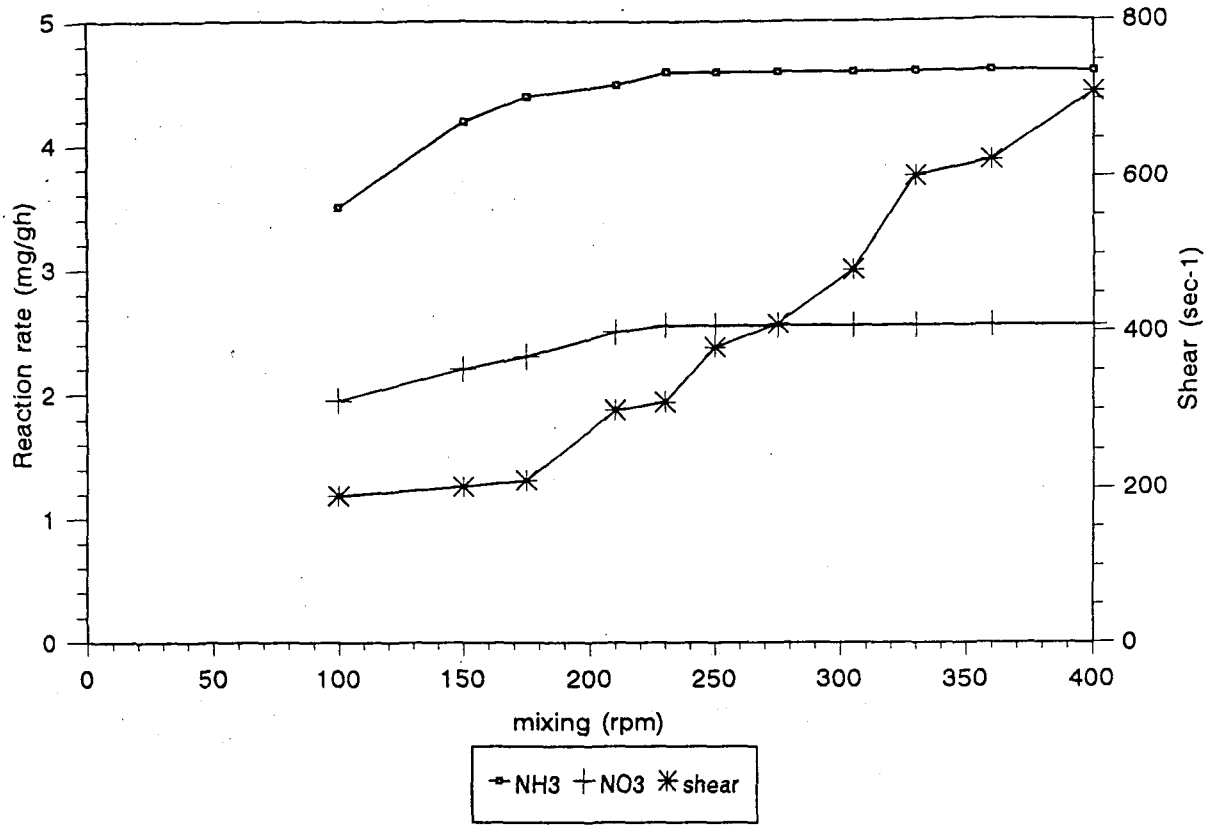


Fig 2.1. Specific ammonia oxidation and nitrate production rate as a function of mixing intensity (Stenstrom and Song, 1991)

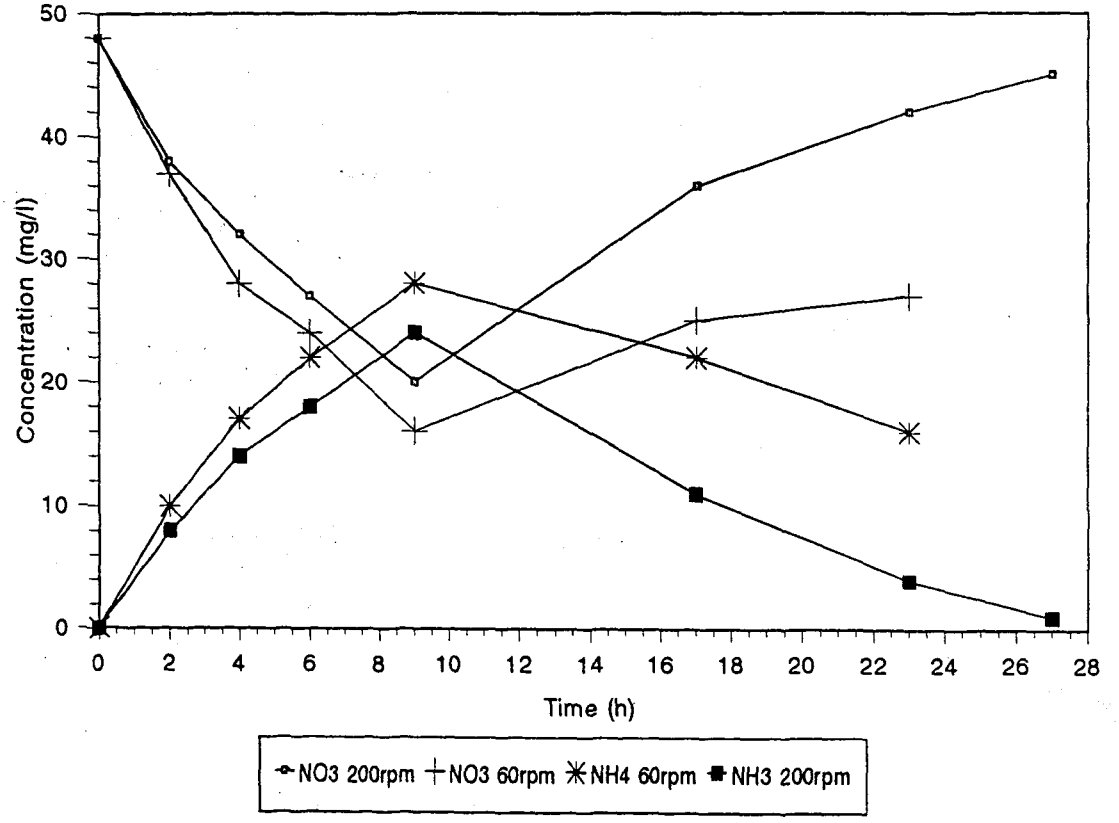


Fig 2.2. Ammonia and nitrate concentrations against time at different mixing speeds. From Stenstrom and Song, 1991.

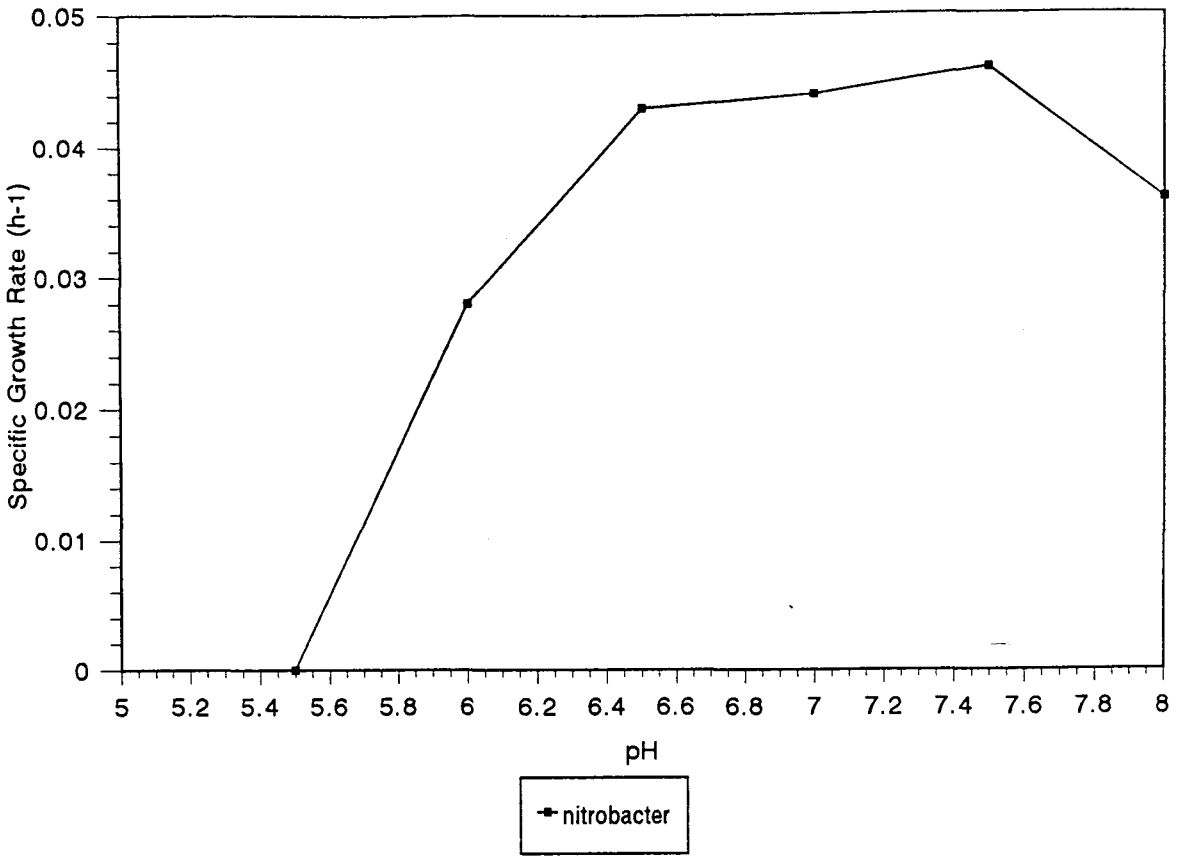


Fig 2.3. Effects of pH on the growth rate of nitrobacter. From Prosser, 1989.

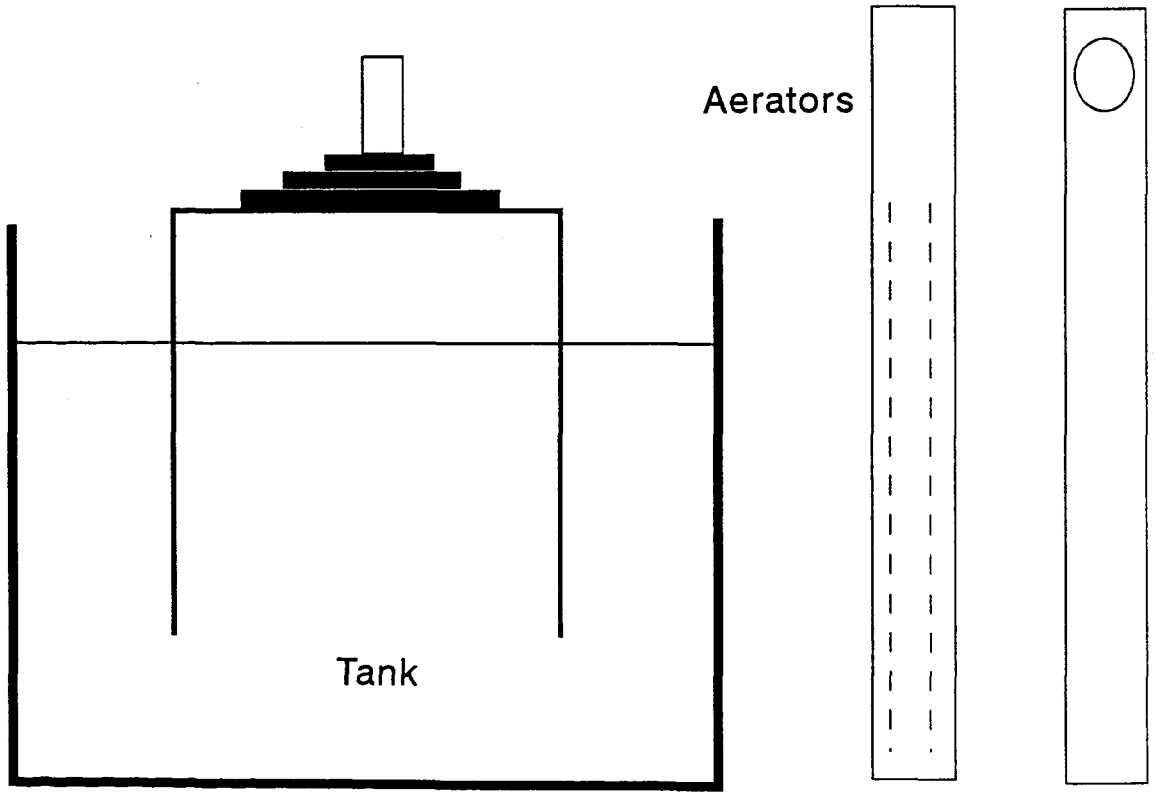


Fig 2.4a. a schematic of the original configuration of the half metre pilot plant unit. Tank size 500mm by 330 mm, aerators 200mm.

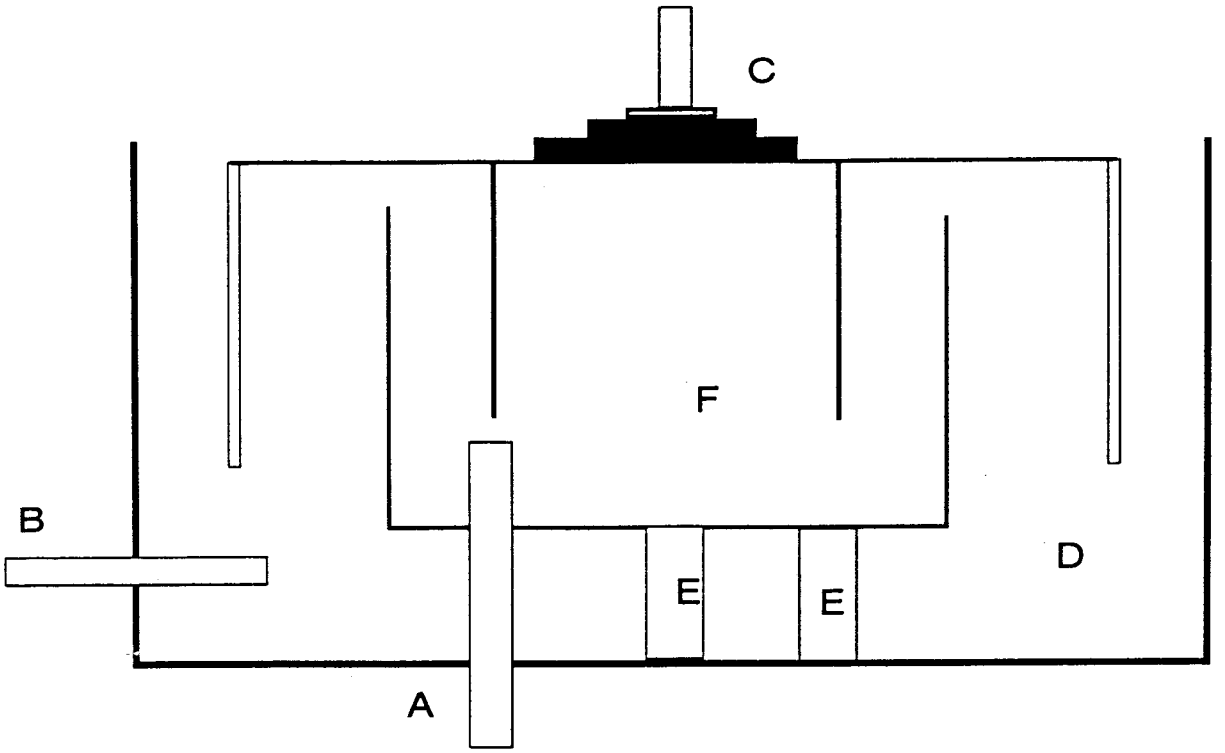


Fig 2.4b. Half metre pilot plant unit, new configuration. A= treated effluent outlet pipe; B= effluent inlet; C=Bearing; D=outer tank; E=Support legs and sludge return point; F=inner tank.

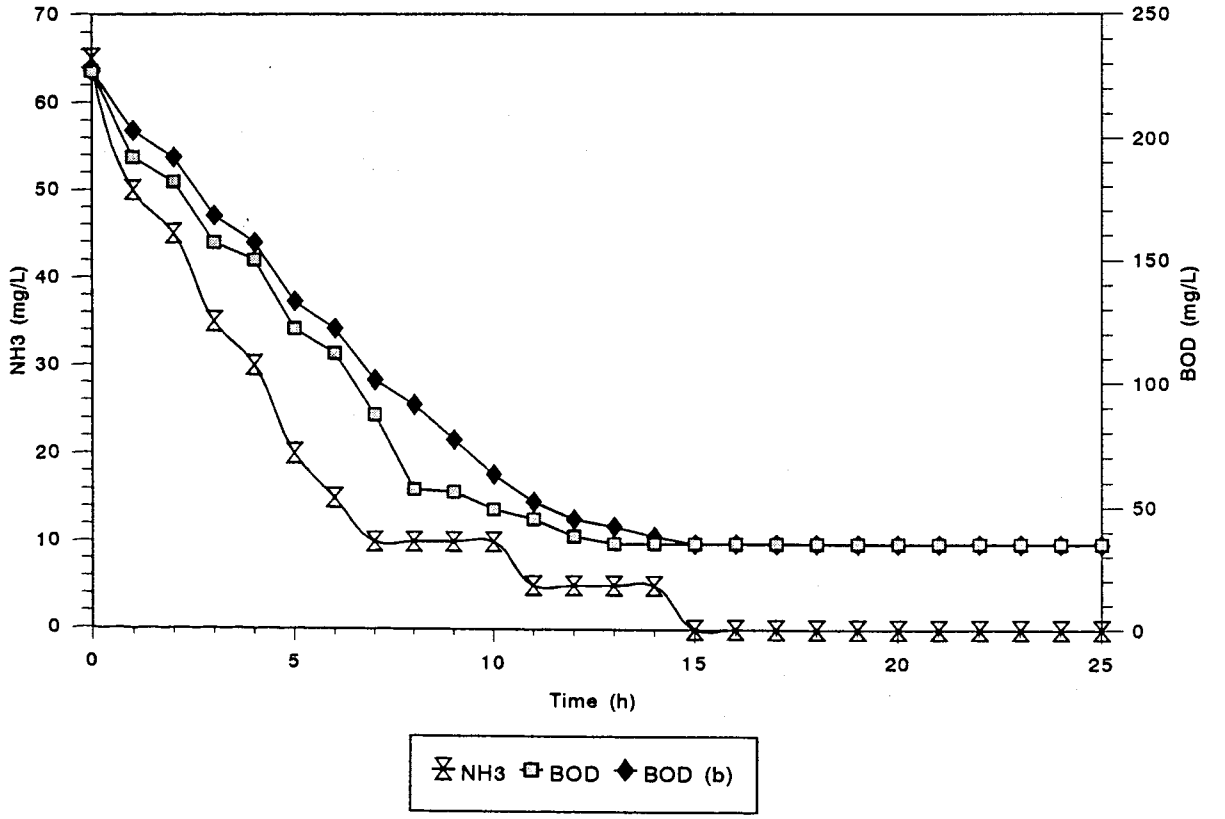


Figure 2.5. 1h intermittent aeration periods of 1h on: 1h off
Oxygen starvation began after 1h, BOD (b) is with allylthiourea

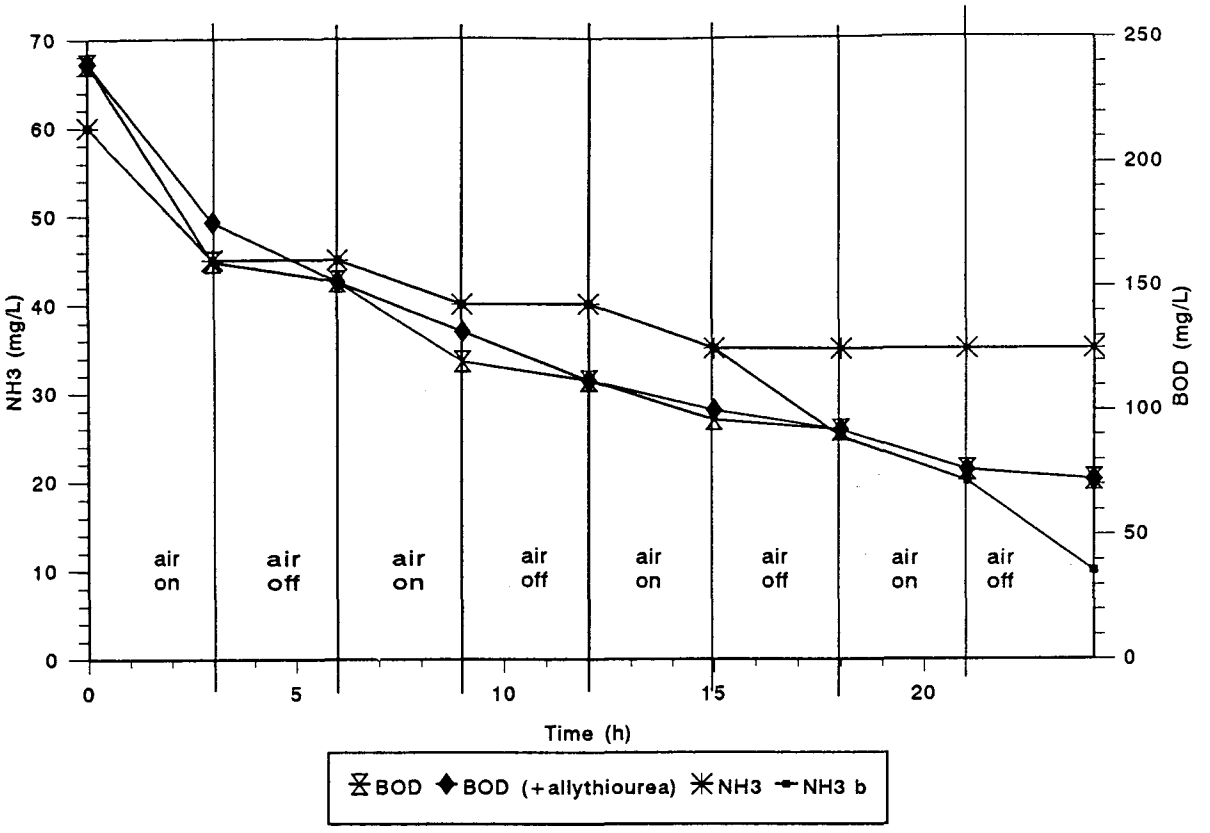


Fig 2.6. 3h intermittent aeration periods of 3h on: 3h off
Oxygen starvation began after 3h, BOD (b) is with allylthiourea
Ammoniacal nitrogen (b) test had constant aeration from 15h

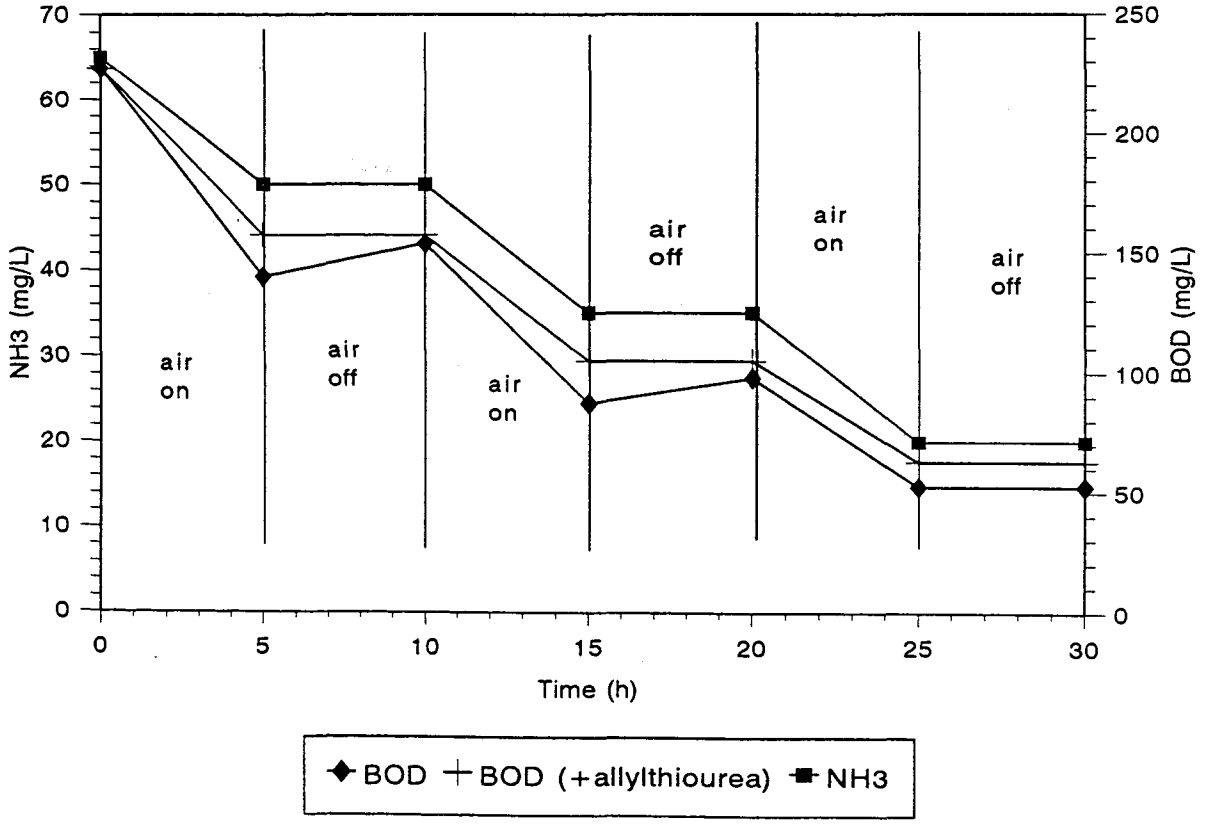


Fig 2.7. 5h intermittent aeration periods 5h on 5h off.
Intermittency began after 5h

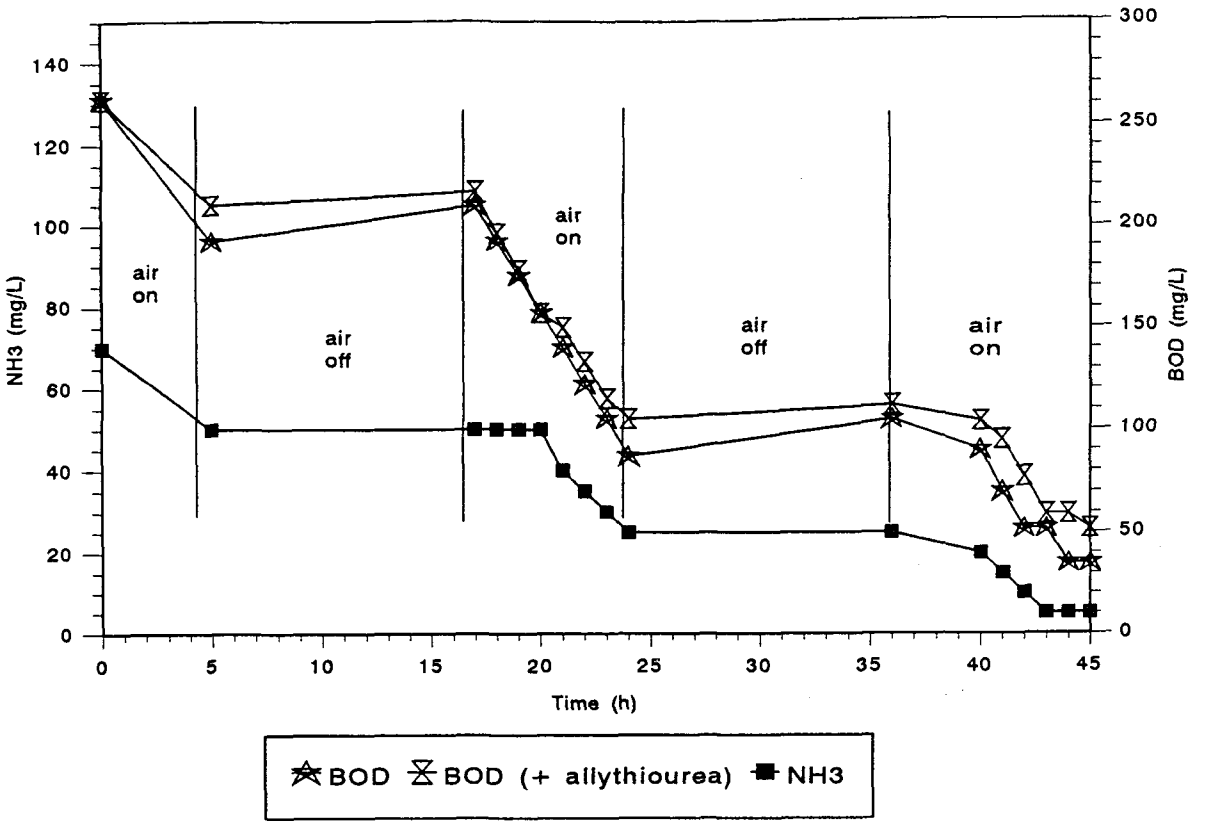
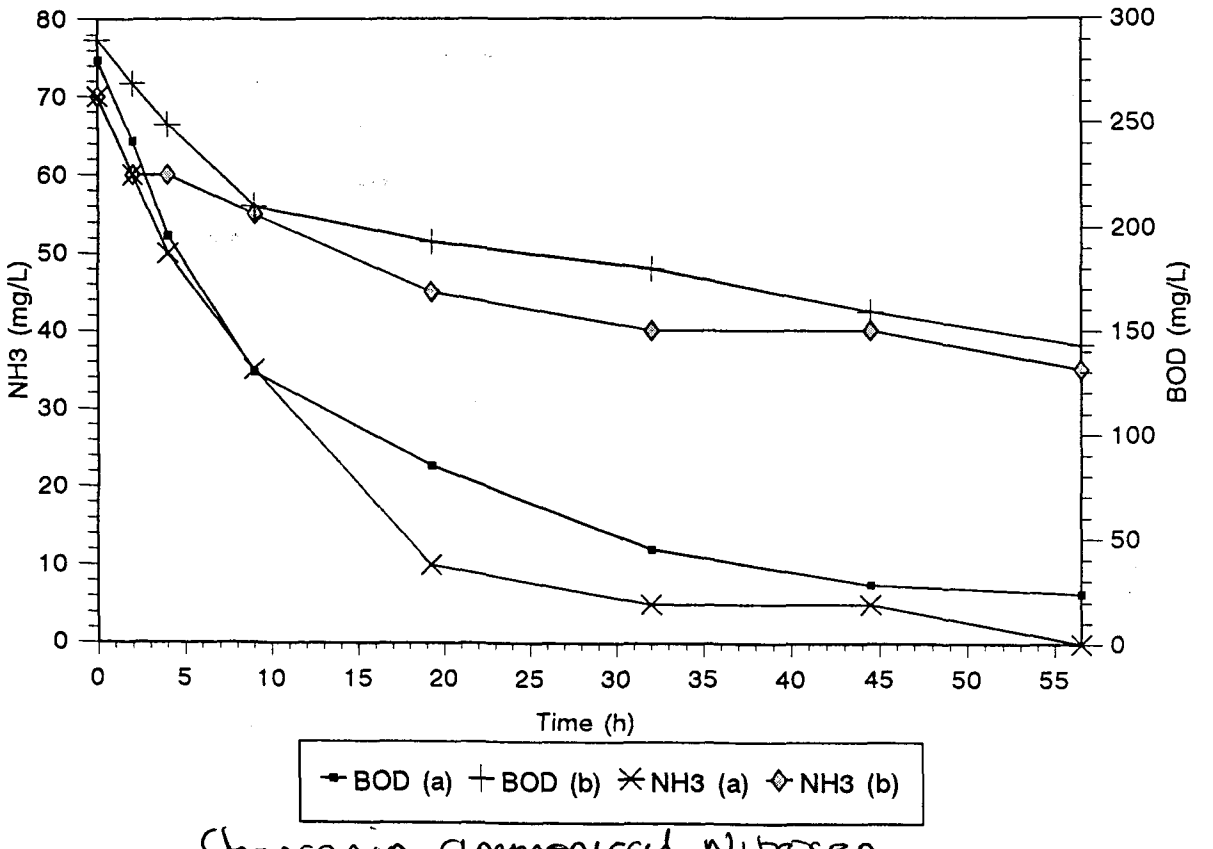


Fig 2.8. recovery time for nitrifiers with 12h starvation periods (5-17), (24-36) starting aeration period was for 5h



Changes in ammoniacal Nitrogen

Fig 2.9. Nitrification rate and BOD reduction rate in the half metre pilot plant unit using constant aeration. (a) - wind powered, (b) = Control

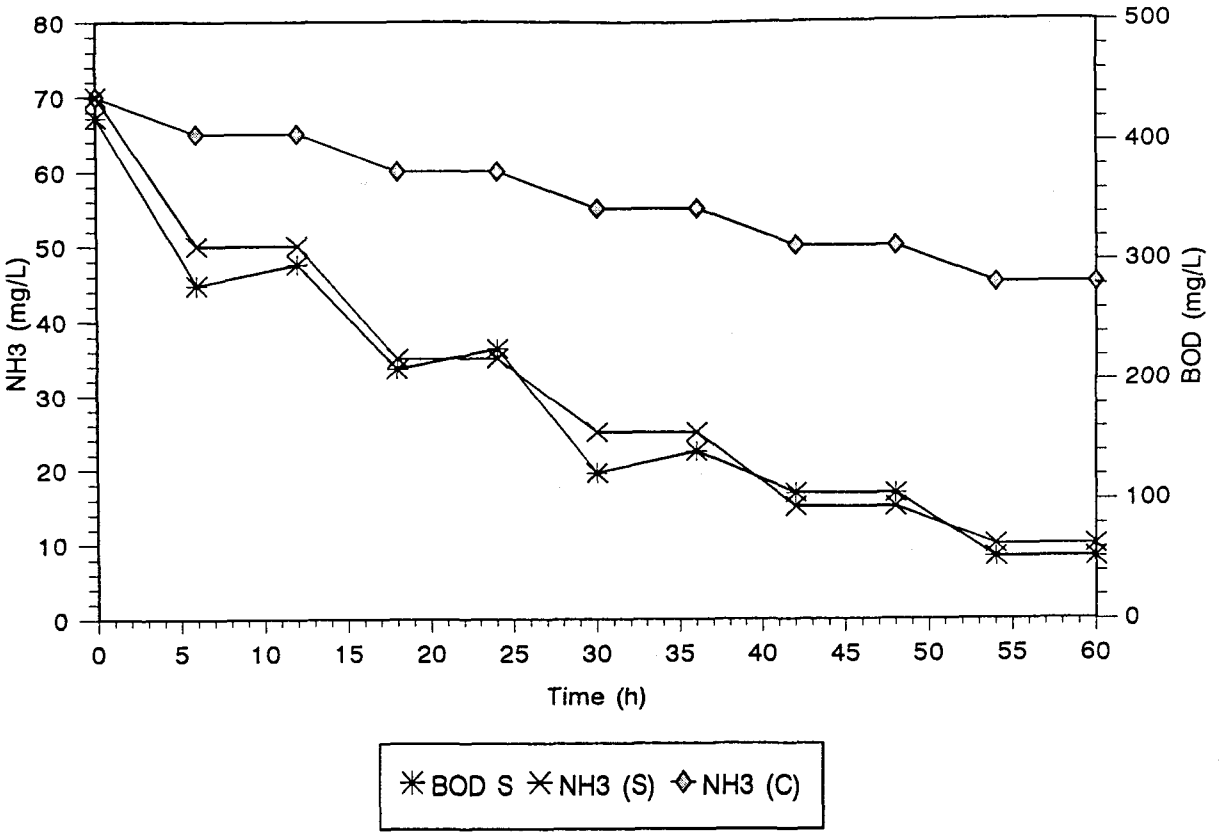


Fig 2.10. recovery time of nitrifiers in the half metre pilot plant model, starting aeration was for 6h, intermittent periods were 6h on: 6h off. S = Sample, C = Control

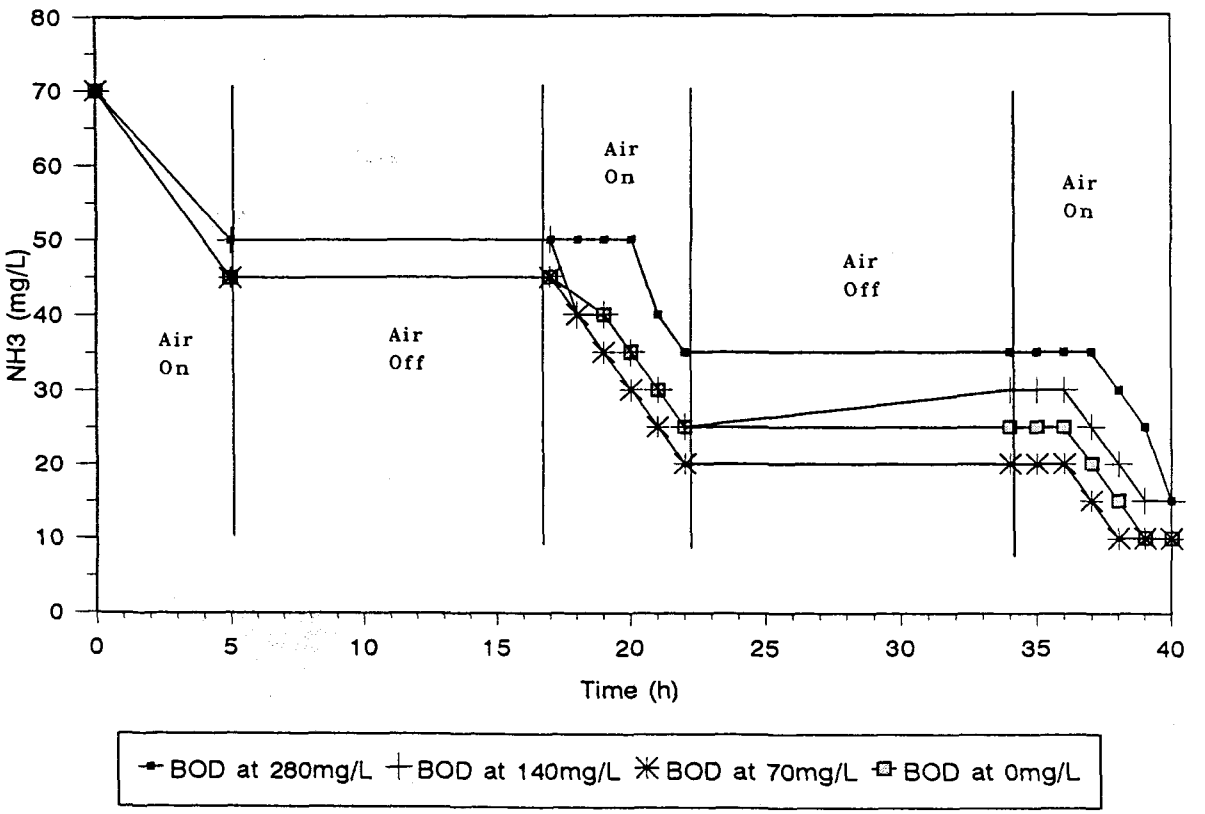


Fig 2.11. Effect of BOD loading on the nitrification rate under intermittent aeration. Intermittency began after 5h, while starvation periods were 12h

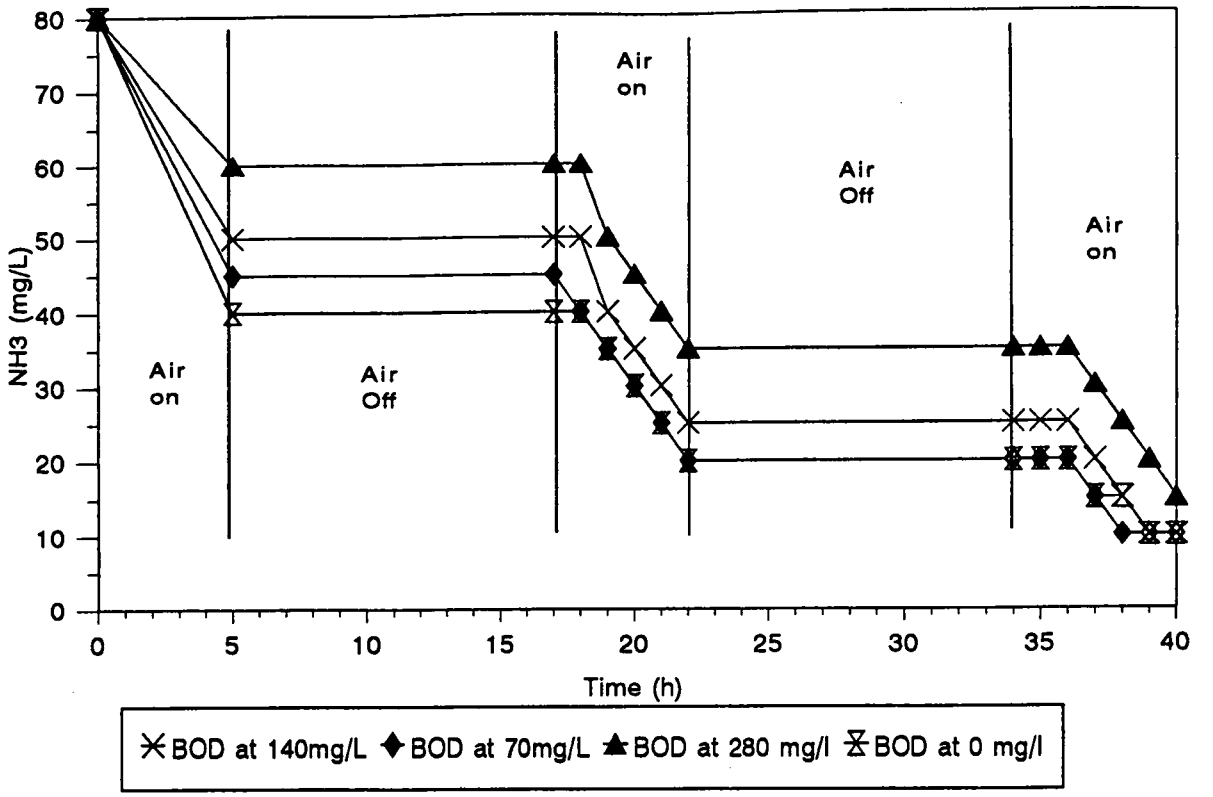


Fig 2.12. Effect of BOD loading on the nitrification rate of stressed AS under intermittent aeration. Intermittency began after 5h, starvation periods were 12h. Stressing as described in materials and methods.

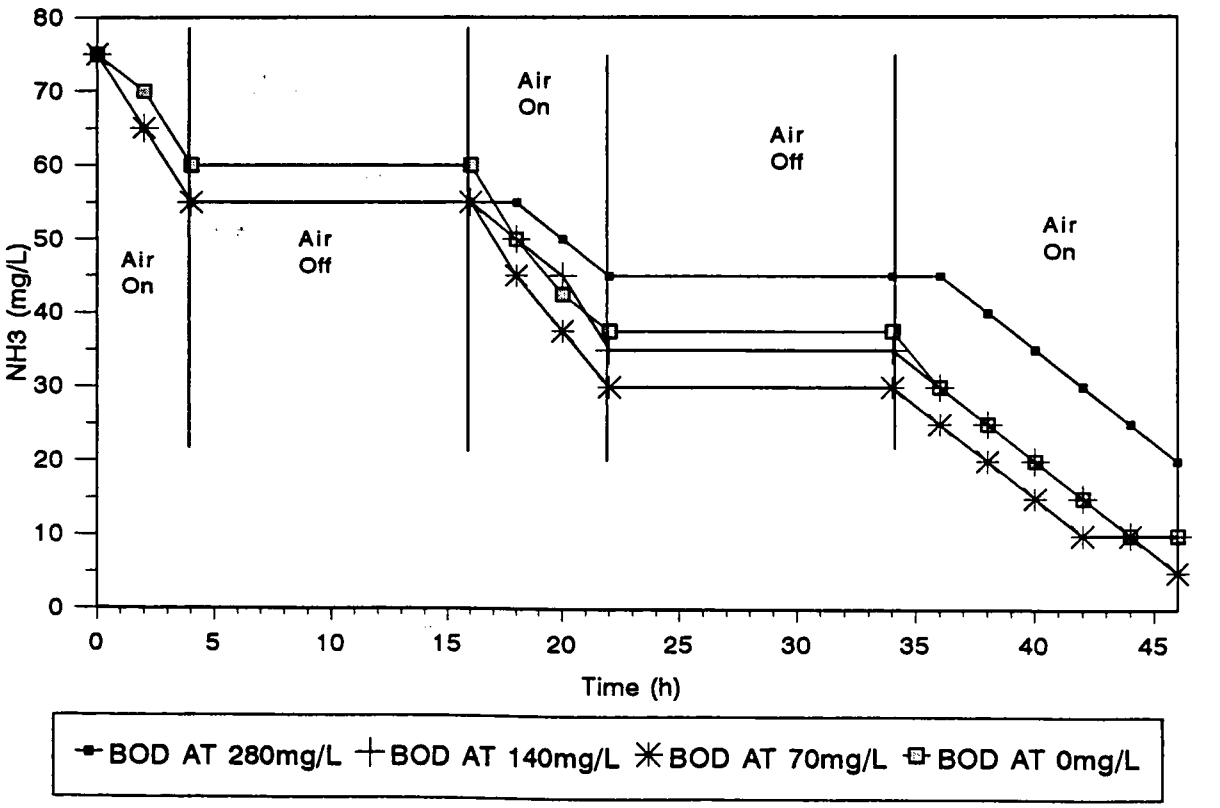


Fig 2.13. Effect of BOD loading on the nitrification rate using pure culture with intermittent aeration. Intermittency began after 4h, starvation periods were 12h. Bacterial concentrations were similar.

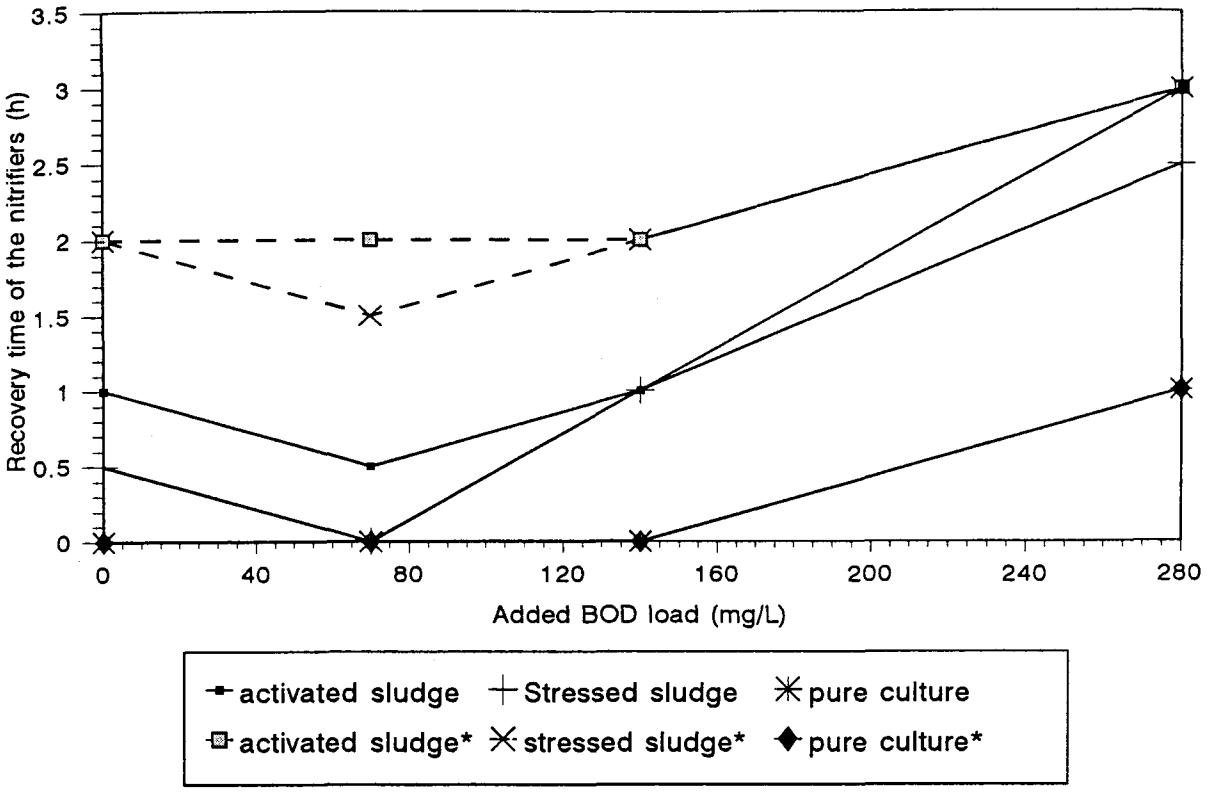


Fig 2.14. Change in recovery time against BOD load, for various populations. Recovery followed a 12h starvation period. Stressing was for 7 days. *=denotes following second starvation period.

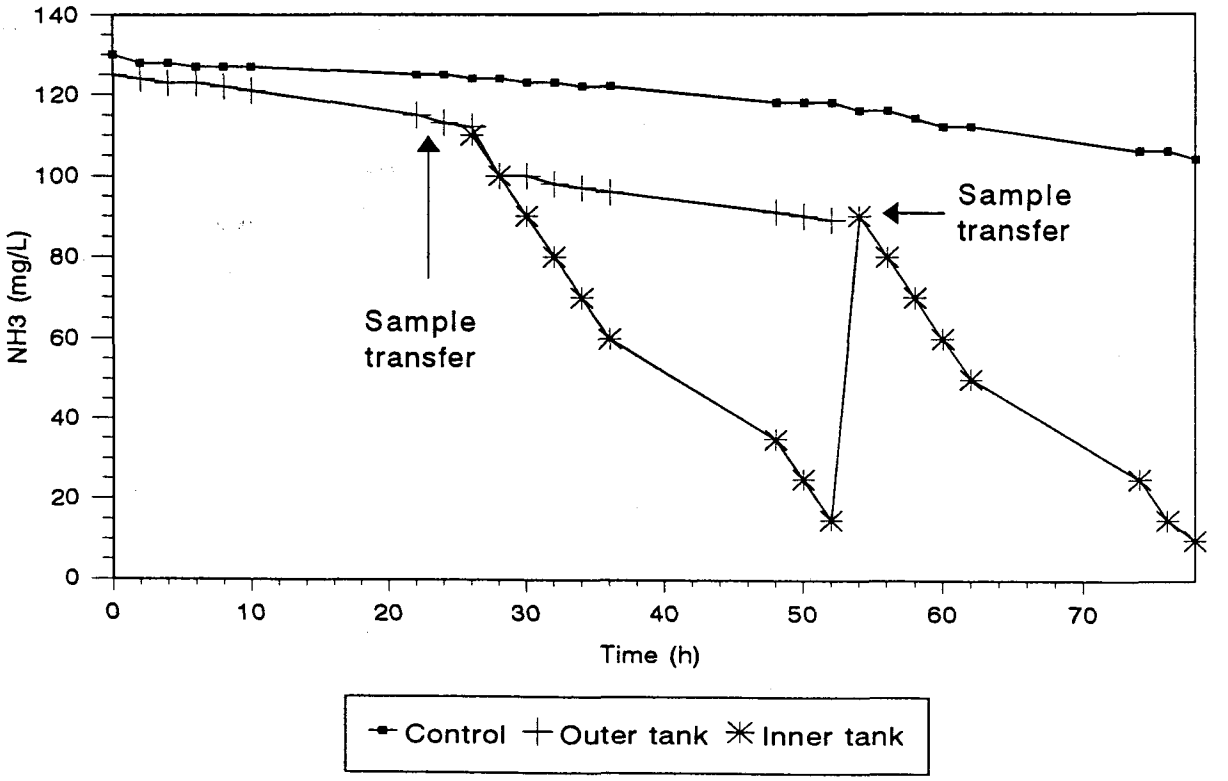


Fig 2.15. Ammonia results for stirred tank tests annular configuration outer annulus BOD removal, inner annulus ammonia oxidation with batch transfer outer to inner as indicated.

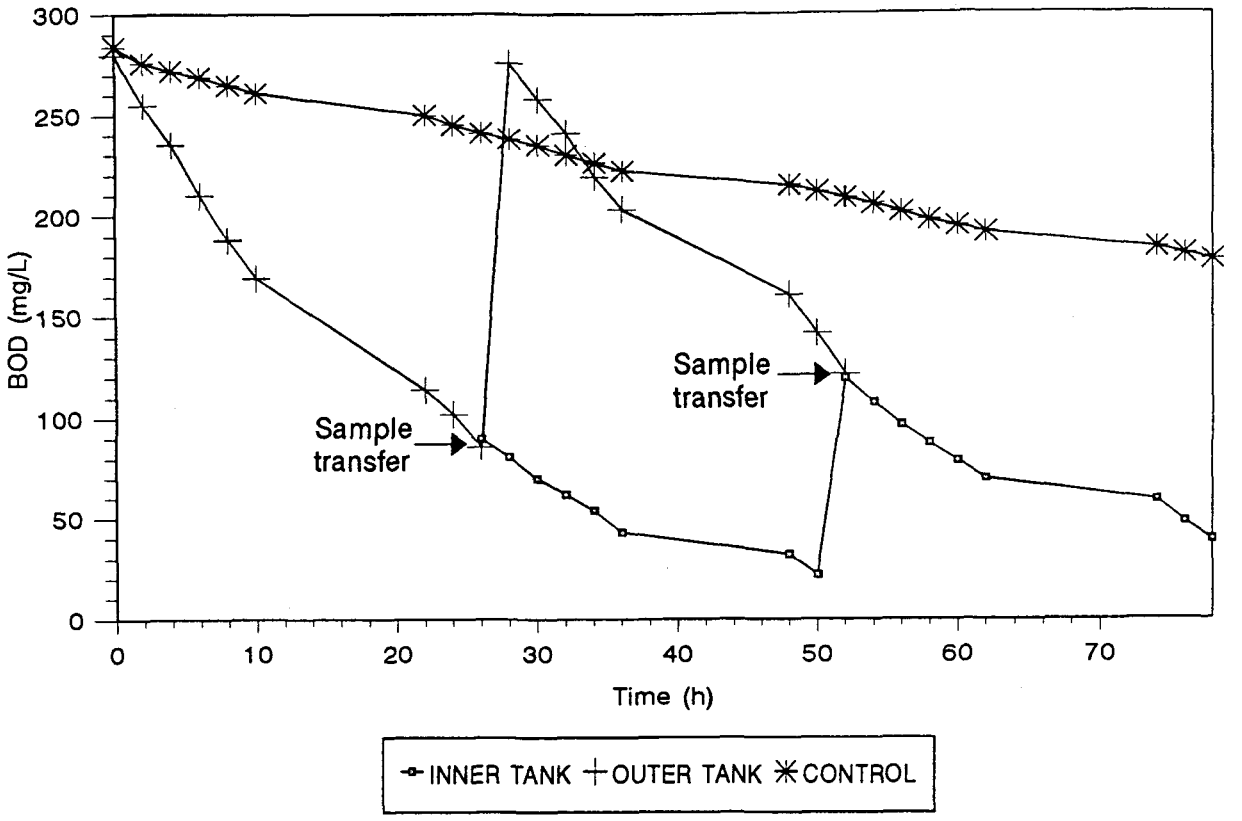


Fig 2.16. BOD time profile for stirred tank annular configuration Conditions as for Fig 2.15

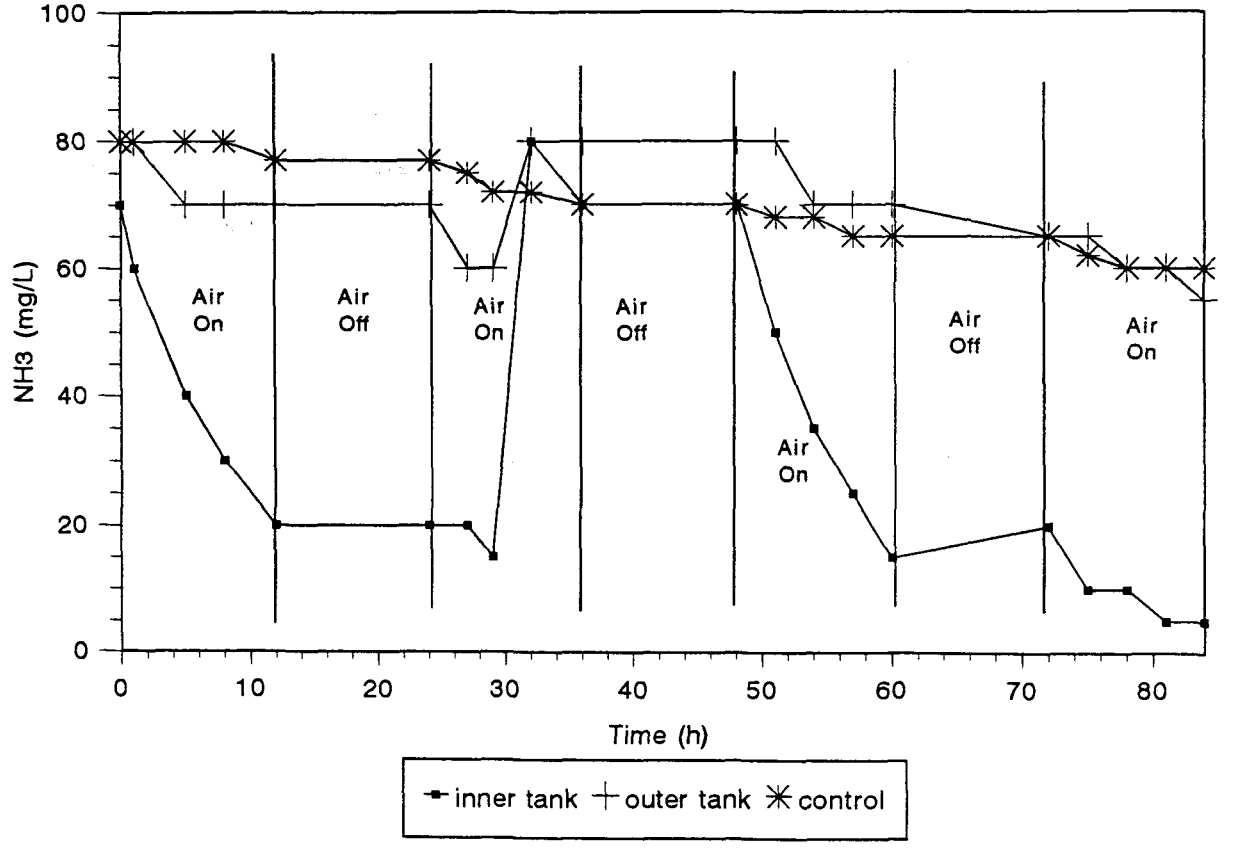


Fig 2.17. Ammonia time profile for two zone tank with intermittent aeration as indicated.

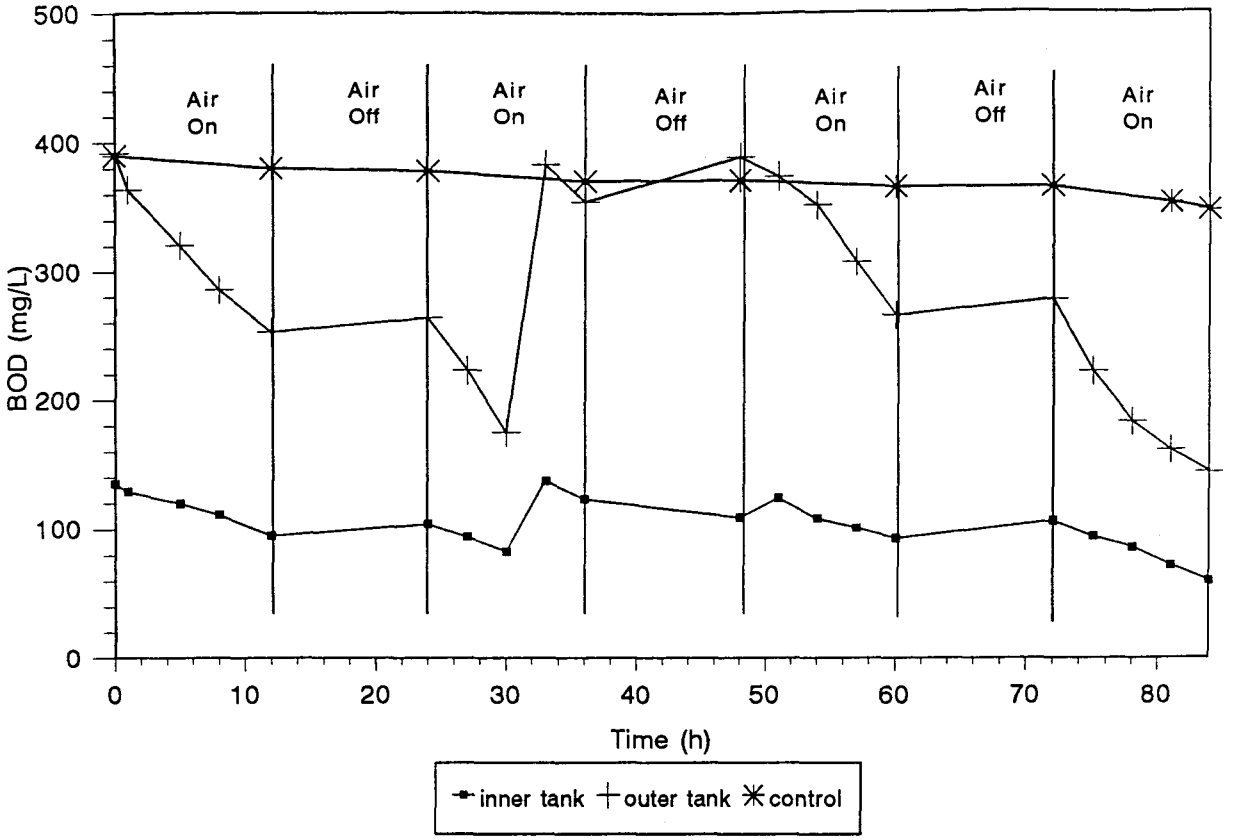


Fig 2.18. BOD time profile for two zone tank with intermittent aeration. Starvation periods were 12h as indicated

CHAPTER 3. 3. BIOFILM MATRICES FOR SLUDGE CONTAINMENT

SUMMARY

The Confinement of biomass within the bioreactor has long been a challenge. This type of retained process has been used for more than a century, in such systems as the trickling filter and more recently in RBC's and BAF's. The use of a substrate in our process was required to retain heterotrophic flocs in the outer zone of a compartmentalised system, to prevent competition with nitrifying bacteria for dissolved oxygen. This competition having been identified as the reason for the poor nitrification performance of our system under intermittent aeration conditions. Therefore, a layer of a low cost mineral substrate was placed in the outer annulus of the system as described in chapter 2. This was then tested under a variety of BOD and suspended solids loadings to determine the nitrification performance. The results of tests using this fixed film substrate showed that substantial performance increases were obtained by retaining the sludge in the outer annulus, specifically with regards to nitrification following anoxic conditions. Concerns were however raised as to the effect of high solids on the fixed bed and whether cleaning would be required, hence the desire for a mobile substrate.

3.1. INTRODUCTION.

The concept of support media for the growth and stabilisation of microbial populations in the wastewater treatment industry has long been appreciated. The main application of this principle has been in the use of fixed films, containing a mixed population of microorganisms in such systems as trickle filters (Christensen, 1989), rotating biological

contactors (Ware, 1990) and biological aerated filters. The trickling filter is by far the most widely used microbial fixed-film device in the waste-water treatment industry, and has been in use since the turn of the century (McGowen *et al.*, 1910; Stanbridge, 1977). In this system a bed of support material, such as gravel or plastic is used, to which a microbial film has attached over a period of time. It is this film of very varied microbial composition that degrades the waste present in solution as it percolates down through the bed, over the film. Aeration for the process is provided via a counter-current air stream, which moves up through the bed. The dissolved oxygen requirements of microbial types in part responsible for their vertical stratification with the aerobic ones near the bottom. This type of process does however have limitations due to the mass transfer limitations of the bacterial film and its ability to cope with shock loads (Stover *et al.*, 1985), and it was these limitations that were the driving force for the development of new processes such as biological aerated filters. Trickling filters come in several types (as discussed in chapter 1); low, intermediate and high rate, with the only real difference being in bed depth (high rate deeper).

The new processes mentioned have developed along two distinct pathways, one in which the films mobility was provided as a solid body translation/rotation supporting the biofilm, such as a rotating biological contactor. The other direction has been to augment the mass transfer coefficient on a fixed film or packed bed reactor such as the biological aerated filter. There are of course several varieties of fixed film reactor (aerated submerged fixed film, up-flow and down-flow filter etc) in which the mass transfer is augmented by motion (forced flow or bubble flow), but all use a very similar process.

As discussed in chapter 1, rotating biological contactors (RBC) consist of several discs (often plastic) to which a bacterial film is attached. These are rotated through the effluent then the air, thus providing the oxygen for the waste products' degradation (Ware, 1990). The typical performance of these reactors is affected by several factors including recirculation rate, submergence, rotational speed etc, but under conditions where there is an average organic loading of 0.005-0.0075 Kg m^{-2} d^{-1} and a hydraulic retention time of 1-2h (Metcalf and Eddy, 1992), BOD is reduced to around 15-30mg L^{-1}. However these systems do have several problems. One such problem is fouling of the discs thus reducing performance, and causing shaft failure (Ross *et al.*, 1994). Another problem arises from inadequacies in design specifically with reference to temperature. The reactors themselves often become too hot during the summer (killing the bacteria) and too cold during the winter (increasing throughput time).

Another important example of aerobic support matrices is the Captor process (Cooper *et al.*, 1985), which uses reticulated polyurethane foam pads as its matrix. In this process the micro-organisms colonise intercellular spaces of the pads, increasing the sludge age and therefore decreasing the throughput time, typically to 1.5-2h (Richards *et al.*, 1985) at a mean BOD load of 1.2-2.0 Kg m^{-3} d^{-1}. Again there are several problems associated with this process, including the removal of biomass from the pads which involves pressing them between rollers, forcing the biomass out. A further problem involved retaining the pads within the reactor as both wind and flow caused them to escape.

All of the support matrices mentioned operate in a relatively similar way; they capitalise on bacterial predisposition to colonise surfaces. This is due to the microbes' abilities to

produce extracellular polymers, permitting adhesion of films containing them to a support medium. Once fixed in/onto such a medium these bio-films become very much multifunctional and highly complex. The main limiting factor for this type of biofilm is oxygen and substrate transfer across the film, with carbon loading and dissolved oxygen levels being closely associated. Indeed dissolved oxygen should normally be considered to be the limiting factor if BOD loadings increase above 400mg L^{-1} (Schroeder, 1976). Below this value the transfer of either BOD or ammonia can be limiting. This stable environment also seem to have an effect in increasing sludge age and allows an increase in the biomass loading within the system, a good example of this being the aerated submerged fixed film reactor (Al-Haddad *et al.*, 1991).

There can, however, also be drawbacks with the use of a support matrix, including a reduced reactor volume due to the matrix. This leads to larger reactors being required. For example trickling filters using mineral support matrices have in general a void space of around 40-50% meaning that reactors have to be 50-60% bigger. However, void space does increase to 90-95% (Metcalf and Eddy, 1992) with new plastic media. This is normally to some extent negated by the increase in bacterial stability and sludge age. If the bacterial film grows too thick there can also be severe problems with the diffusion of materials across and through the film, particularly with reference to nitrification (Beccari *et al.*, 1992; Sarner, 1981). For nitrification it is clear that the internal diffusion resistances for oxygen cannot be neglected in evaluating the overall kinetics, with a marked increase at thicknesses above 100μm and with dissolved oxygen values below 2mg L^{-1} (Beccari *et al.*, 1992) . A final problem centres on the solids that can accumulate

in these matrices. This build-up can mean that the matrix becomes blocked with solids from the waste, or by any excess biomass produced by the microbes. This blocking could cause the aerobic layers to become anaerobic and/or decrease the performance of the system.

There have been several different types of support matrix used in wastewater treatment including wood (Lo and Liao, 1987), plastic (Ware et al, 1990), calcium carbonate (Goronszy and Eckenfender, 1987) and gravel etc, all of which have several advantages and disadvantages, but one of the main considerations was cost.

(A) Wood; advantages include low cost and it can be utilised as a food source by the bacteria, therefore has a shorter start-up time (Lo and Liao, 1987). However disadvantages include leaking lignin (a recalcitrant product) into the water, low surface area and, potentially, relatively rapid disintegration of the support.

(B) Sintered glass has several advantages, such as good porosity and pore size distribution, very large surface area (up to 90,000 m^2 m^{-3}) and high resistance to attrition (Breitenbucher, 1990). However it can often be disintegrated by microorganisms growing in the pores, and it is very expensive and therefore uneconomic.

(C) Calcium carbonate (granular) possesses a large surface area, and is relatively cheap. However its main disadvantage is that it needs constant replacing as it can

dissolve in the wastewater. This dissolution of course provides buffering capacity when treating acidic wastes (Goronszy and Eckenfender, 1987).

(D) Packed bed substrates i.e. plastic, gravel and slag products; cheap, (plastic when mass-produced), long lasting and has large surface area (Ware et al, 1990). Main disadvantage is low void fraction (Except certain high cost plastics).

(E) Reticulated foams (polyesters and polyethers), cheap, large surface area (Webb and Black, 1992) potential problem of attrition-damage (porous silicone rubber alternative (Perry and Knights, 1994)).

As can be seen the options for suitable substrates to retain the sludge in a specific area are numerous. It is, however, vital to make the right choice, as the competition between the heterotrophic and nitrifying populations can seriously curtail plant performance, if the configuration is sub-optimal. The choice of which substrate adopted is covered later, but centred on cheapness, robustness and the requirement of a large surface area. The work is presented in four sections; the materials and methods, results from half-metre pilot plant tests, discussion and conclusions.

3.2. MATERIALS AND METHODS

3.2.1 Equipment

The test unit (half metre scale) as described by Janes and Thomas (1992) and schematised in Fig 3.1 comprises a 500mm diameter circular outer tank of depth 330mm. Supported inside this tank is an inner tank (diameter 340mm, depth 220mm). This latter tank had twelve symmetrically deployed 5mm-diameter holes around the wall, 20mm from the top. These holes were to facilitate the transfer of water from the outer to the inner tank. As indicated in figure 3.1 the inner tank was supported at a height of 100mm above the base of the outer tank and the space between these two was filled to a depth of 80mm with Lytag as a packed bed support medium (discussed later). As shown in Fig 3.1, pipe work included a supply line into the outer annulus and the outlet from the inner annulus. In addition two drainage holes were incorporated to facilitate the return of accumulated sludge to the outer annulus. As in chapter 2-power supply to the aerators was provided either by a motor (2kW) or a fan driven Vertical Axis Wind Turbine (VAWT). Although the motor size is large, only a small proportion of this power was used with the power draw typically being as little as 2-5W at 125rpm, which was the maximum used throughout this experiment (Fowler *et al.*, 1993).

3.2.2. Procedure

The tests were conducted with the tank primed with screened sewage (40L by volume) and activated sludge as inoculum (4L by volume, 10:1 dilution). They were both

obtained from Minworth Sewgae Treatment Works and poured into the outer annulus. As in chapter 2 regimes of continuous and intermittent aeration were evaluated here with zero aeration for a period of 12h followed by aeration for 6h then a further 12h-starvation period. This sequence is initiated by priming the effluent with a 12h period of aeration allowing the system to come to equilibrium. Strategy options were assessed with regard to semi-continuous operation, in all cases transferring liquid from the outer to the inner tank by gravity displacement over a 1h period through the array of transfer points while the aerators were turning.

As control we used a tank of diameter 300mm and depth 400mm, located in the path of the wind but without any submerged aeration; in other words oxygen transfer was by diffusion. The wind speed produced by the fan was approximately 4m s^{-1} and caused only slight rippling of the control's liquid surface. The control tank contained 15L of primary screened sewage and 1.5L of activated sludge (10:1 by volume as biocatalyst). Vapour losses due to evaporation for the control and for the test facility were replaced by distilled water at regular intervals of 4h. The overall amount added was 1L in the control and 2.5L for the test rig. To maintain nutrient supply, 3L of primary screened sewage were added daily. Before embarking on the evaluation a 5-week period of bacterial colonisation onto the substrate was allowed, A regime of constant aeration was maintained throughout this colonisation period. The work itself was split into three tests; (i) motor driven constant aeration, (ii) wind driven constant aeration, (iii) motor driven intermittent aeration.

The duration of the first two tests was 15 days with a representative sample of the data being discussed here (days 2, 7, 12 and 15). The third test consisted of four 40h intermittent aeration regimes, which were run consecutively. In all three cases the starting effluent conditions (for the majority made-up of primary screened sewage from Minworth Sewage Treatment Works) were the same with initial BOD approximately 400-450mg L^{-1}, suspended solids 350mg L^{-1}, ammonia 50-55mg L^{-1} and negligible nitrite and nitrate. The initial ammonia concentrations for each test were made approximately equal by the addition of an appropriate volume of 1M ammonium carbonate ($(NH_4)_2CO_3$) solution to the influent header tank. All new effluent additions were made via this header tank, under gravity through the inlet pipe as indicated in Fig 3.1, and consisted of 4L of effluent being added daily at time zero for the graphical data; the 3L for initial colonisation was insufficient for the tests. This daily addition meant that the nitrifiers were competing with the heterotrophic bacteria constantly in a high BOD environment, thus this represents a tough test for nitrification.

For the intermittent aeration experiments the following aeration protocol was used, first after effluent addition the system was aerated for 5h, then subjected to a 12h-starvation period. Following this it was re-aerated for 6h, during the final hour of which the liquor was transferred from the outer to inner annulus using gravity displacement. The system was then not aerated for a further 12h, following, which aeration was resumed for a final 5h, immediately after which new effluent was added (approximately 30L) and the regime repeated. Up to the first 22h samples were taken hourly during the aerated periods, from the outer annulus, after 23h from the inner annulus, the sample volume being some 75-

100ml. All tests on these samples were performed either immediately after sampling (ammonia, suspended solids, nitrite and nitrate) or the following day (BOD) after being kept in a fridge overnight at less than 4°C. Sampling for the first two tests was carried out at 8 hourly intervals on each of the 15 days.

All samples were tested for BOD, ammonia, suspended solids (SS), nitrite and nitrate. BOD was estimated by the Winkler method (Winkler, 1982), where pre-aerated, and buffered water (247ml) plus a sample (3ml) from the test are placed in duplicate BOD bottles (250ml). One of these is titrated immediately to give the dissolved oxygen concentration; the other is incubated at 25°C in the dark for five days, before it is again titrated for dissolved oxygen. The difference between these titrations is equivalent to the sample's BOD concentration. Ammonia, nitrate and nitrite were tested using Quantofix Test Strips (Camlab), which are sensitive down to 1mg L^{-1} (nitrite) and 5mg L^{-1} (ammonia and nitrate). The sensitivity of the ammonia test strips was increased by the use of standard ammonia solutions of 2.5mg L^{-1}, 7.5mg L^{-1}, 15mg L^{-1} and 30mg L^{-1}, into which strips from each new batch were dipped to provide a new colour scale. The same protocol was used for the nitrate/nitrite strips to increase their sensitivity (calibration values for nitrate as for ammonia, nitrite values were 0.5mg L^{-1}, 1mg L^{-1}, 2.5mg L^{-1}, 5mg L^{-1}, 7.5mg L^{-1} and 10mg L^{-1}). Suspended solids were measured using the standard filtering method (Metcalf and Eddy, 1992), which means you allow a set volume (50ml) of sample to settle out (2-3h), giving a volume of sludge. This sample was then filtered through a pre-weighed filter paper (Whatman GF/C, pore size 1.2μm) and then placed on a watch glass in an oven at 105°C, overnight to remove moisture. Finally the filter paper

was re-weighed, giving the weight of suspended solids, which could then be related to sludge volume.

The choice of support matrix proved to be simpler than envisaged due to the tight cost restraints placed upon us, which automatically discounted several of the options including sintered glass, high void fraction plastics and porous silicone rubber. This left a choice between packed bed media such as gravel, reticulated foam and calcium carbonate. Calcium carbonate was discounted because dissolution would have led to constant replacements, and this was too expensive. Reticulated foam was also discarded due to the problems encountered with its cleaning in the Captor process (Webb, 1992), and its associated problem of loss through wind capture. This left various gravel based packed bed media, with the eventual choice being Lytag, a power station slag that had been used in aquaria for several years. This was lighter than gravel but still very resistant to attrition. Also due to its high temperature treatment in the power station it was inert with no obvious toxic properties. As shown in Fig 3.2 the surface of the Lytag seemed to lend itself to microbial colonisation very well, due to its uneven nature. The only treatment required for the Lytag was sizing (to diameter 8-14mm) and washing to remove any coal dust. Fig 3.3 shows the Lytag layer in situ in the half-metre tank.

3.3. RESULTS MAINTAINED AERATION

3.3.1 Maintained aeration - Motor Driven

Fig 3.4 shows BOD concentrations versus time for 8h periods monitored on days 7, day 12 and day 15 on an extended run with maintained aeration. While broadly demonstrating repeatability there is also clear indication of an improving trend with age of the system. Typically, then, BOD is reduced from 400mg L^{-1} down to 75-100mg L^{-1} or so during each 8h period, which corresponds to oxidation rates of 35-40mg L^{-1} h^{-1} which is substantially higher than that achieved in the earlier tests without substrate. The decay in oxidation rates from a peak value of 50-55mg L^{-1} h^{-1} down to 10-20mg L^{-1} h^{-1} is consistent with substrate utilisation and in line with expectations from reported results. The overall oxidation rate is better than the 10-15mg L^{-1} h^{-1} achieved in intermediate biological filters (Christoulas *et al.*, 1990).

Fig 3.5 shows the nitrification rate represented as ammonia, nitrite and nitrate concentrations versus time for an 8h period during day 7. It was noted that the ammonia decrease is tracked by a corresponding increase in nitrite, which subsequently decays to nitrate, whose level increases during the total test period. Overall nitrification rate was approximately 4mg L^{-1} h^{-1}, which is comparable with the 4mg L^{-1} h^{-1} achieved without support matrix (chapter 2). The nitrate production rate was also unaffected by the introduction of the Lytag. The best nitrification occurred between 4-5h, consistent with a decrease in BOD (Fig 3.3) to 120-100mg l^{-1} in line with the results in chapter 2. It is

noted that there are inconsistencies in the nitrogen balance in Figs 3.5 and 3.6. This is due to errors in the test methods used to identify the ammonia, nitrite and nitrate levels.

Turning now to Figs 3.6 and 3.7 the two graphs show ammonia oxidation increased compared to Fig 3.5 with rates ranging up to 6mg L^{-1} h^{-1}. This is consistent with the notion of biological conditioning. The maximum nitrite concentrations in both cases were also increased from 5mg L^{-1} to 10mg L^{-1}, but this falls short of the ammonia level suggesting that nitrite oxidation to nitrate is as quick as that of nitrite production (Laanbroek *et al.*, 1994). Unlike Figs 3.5 and 3.6, in Fig 3.7 the nitrate profile shows a peak after 4h elapsed time followed by a monotonic drop to a low final value of 5mg L^{-1} at the end of the test. The decline in nitrate concentration is consistent with the establishment of denitrifying anoxic bacteria within the depths of the Lytag substrate. The question arises as to the concentration profiles of dissolved oxygen in the centre of the packed bed, which is the subject of other studies in progress. The reason for the lack of denitrification in the other tests is the slow growth rates of anoxic bacteria (Stryer, 1992). This means that it took time for the initial low population number to proliferate to a noticeable level.

Finally Fig 3.8 shows the time profile for the suspended solids concentrations against time for the three tests reported above. Initial concentration of suspended solids for this primary screened sewage fed to the treatment plant was approximately 350mg L^{-1} ± 30mg L^{-1} at the point of entry, below the Lytag layer. Passage through then reduced it to 120-140mg L^{-1} from elapsed time zero at the start of the test to 8h from the introduction

of the sample. From this data and the graph an immediate drop of some 200mg L^{-1} can be seen. The reason for this drop is that samples could only be taken from just above the Lytag layer. The reduction in solids is therefore occurring in large part due to precipitation in the Lytag layer, so conclusions can only be drawn as to the proportion of suspended solids that have been biologically digested directly by the upper aerobic zone. As can be seen from the graph in this aerobic layer suspended solids were reduced from 120-140mg L^{-1} to approximately 50mg L^{-1} at a rate of 10mg L^{-1} h^{-1}, which is comparable to that achieved in most trickling filters (Quasim, 1985). Thus there is an open question remaining from this data as to the degree of anoxic/anaerobic digestion that might be achieved within the depths of the Lytag layer and, in turn whether sludge removal is required and its frequency.

3.3.2 Maintained aeration - Fan driven..

Fig 3.9 shows BOD time profiles for 6h periods monitored on days 2,7,12 and 15 on an extended run with fan driven maintained aeration. As before the graph broadly demonstrates repeatability while it is also clear that there is an improving trend with the age of the system. In all cases satisfactory oxidation was achieved in a time period of 6h, following the addition of fresh effluent. Moreover with the Vertical Axis Wind Turbine powered aeration BOD removal rates of up to 60mg L^{-1} h^{-1} were delivered as against 36mg L^{-1} h^{-1} achieved with the motor driven aerator, due primarily too the increase in aerator rotation speed from 125rpm for the motor to 150rpm for this test. The actual oxidation rate of 60mg L^{-1} h^{-1} compares favourably with those achieved in more established plants such as trickling filters (10-20mg L^{-1} h^{-1}, Quasim, 1985).

Corresponding ammonia concentration profiles for this test appear in Fig 3.10, clearly demonstrating satisfactory ammonia oxidation in all cases, with rates of up to 5mg L^{-1} h^{-1}, which is comparable with the motor driven tests. The nitrite and nitrate concentrations appear in Figs 3.11 and 3.12, and it is noted that Fig 3.11 shows zero initial nitrite levels; presumably a legacy of earlier ammonia oxidation. There is, furthermore, an overall nitrite reduction with time, down to values of 75mg L^{-1} at the end of the 6h test. Moreover, the nitrate profile (Fig 3.12) for day 12 shows a peak value at 3h of 25mg L^{-1}, followed by a decrease. This again suggests that there is some denitrification occurring somewhere in the depths of the Lytag layer.

Suspended solids concentration profiles appear in Fig 3.13, and it is clear from the graph that at the end of the test the liquor transferred from the outer to inner annulus had a suspended solids concentration of around 50mg L^{-1}. Initial concentration of suspended solids for this primary screened sewage fed to the treatment plant was approximately 350mg L^{-1} ± 30mg L^{-1} at the point of entry. Following passage through the Lytag bed values of 120-140mg L^{-1} from time zero at the start of the test to 6h from the introduction of the sample were measured. From this data and the graph an immediate drop of some 200mg L^{-1} can be seen, as it was in the motor driven tests. The reason for this drop is that samples could only be taken from either just above the Lytag layer. The reduction in solids is therefore occurring in large part due to precipitation in the Lytag layer as it was for the previous test, so the only conclusions that can be drawn are for the proportion of suspended solids that have been biologically digested directly by the upper aerobic zone. As can be seen from the graph in this aerobic layer suspended solids were

reduced from 120-140mg L^{-1} to approximately 50mg L^{-1} at a rate of 10mg L^{-1} h^{-1}, or 85% which is comparable to the 90% achieved in both trickling filters (Quasim, 1985, Anon, 1988) and 95% in AS plant (Anon, 1988). Again questions remain as to the degree of anoxic digestion that might be achieved within the depths of the Lytag layer, whether sludge removal is required and what would be its frequency.

3.3.3 Motor powered intermittent aeration.

The aeration regime used for these intermittent aeration experiments was 5h on, 12h off, 6h, on 12h off, and finally 5h on. The total duration was 40h, with effluent transfer from outer to inner annulus at 22-23h (during aeration). Fig 3.14 displays BOD concentration time profiles confirming that during maintained intervals, oxidation continued at a well defined rate of up to 25mg L^{-1} h^{-1} with an overall total period average of about 10mg L^{-1} h^{-1}. Whereas during the starvation periods there was negligible BOD removal, which is consistent with the earlier findings. The overall retention time of some 40h is significantly longer than present plant such as AS. However, if the non-aeration periods are removed from the overall retention time, then a more realistic "treatment" time of 16h is found. This is the retention time that should be used, as the process under consideration is based on wind powered aeration.

Ammonia time profiles in Fig 3.15 for the four tests demonstrates satisfactory removal rates in all cases from about 50mg L^{-1}, down to less than 5mg L^{-1} in a total time period of 40h which includes two non-aerated periods of 12h each. Nitrification was broadly maintained at satisfactory rates during the 16h of aeration. These results represent

oxidation rates of about 3.75mg $L^{-1} h^{-1}$ based on the aeration periods or 1mg $L^{-1} h^{-1}$ or so over the total period and therefore are approximately 25% reduced in comparison to the maintained aeration experiments. Again, there is evidence of a lag between resumption of aeration and nitrifier activity ranging from 2-3h according to the conditions of the test. In particular the residual BOD at the time was noted. As with the earlier results it can be seen that nitrification lag is reduced to less than 1h following the second starvation period where the BOD has been reduced to 100-120mg L^{-1}. In these present studies the nitrification is predominately within the inner annulus following transfer after 22-23h by introduction of a new addition of effluent to the outer annulus.

Sample nitrogen balance time profiles (ammonia, nitrite and nitrate) for two of these tests are shown in Figs 3.16 and 3.17 indicating that good ammonia oxidation via nitrite occurs in response to aeration even when this is intermittently supplied. Interestingly there is an indication that nitrite reduction occurs to some extent during the non-aerated period (see Fig 3.16) although this is not matched consistently with the nitrate concentrations. These also decrease during the non-aerated periods; again this decline is attributed to denitrification in the absence of oxygen, with the denitrifiers being situated again in the lower reaches of the Lytag layer.

The significance of packed substrate (Lytag) is profound in so far as BOD removal rates increased from around 7mg $L^{-1}h^{-1}$ to 35mg $L^{-1}h^{-1}$ in the motor tests and up to 60mg $L^{-1} h^{-1}$ with the wind-powered aerators. This suggests that perhaps there is significant augmentation of the gas transfer due to surface agitation. It is worth while to compare

these removal rates with established figures for currently used commercial equipment, for example for trickling filters, values range from 7 to 30mg L^{-1} h^{-1} (Metcalf and Eddy, 1992). When the nitrification rates are also compared it can be seen that the rates achieved without Lytag, 4mg L^{-1} h^{-1} were only slightly increased when Lytag was added to the system (to 5mg L^{-1} h^{-1}). However competition between heterotrophic and nitrifying bacteria for the dissolved oxygen was significantly reduced as can be seen by the reduction in the lag times following anoxia, from 3h without support substrate to less than 1h with support media. The actual nitrification rate of around 5mg L^{-1} h^{-1} is comparable with industry standards as there is very little nitrification found with intermediate and high rate trickling filters (Metcalf and Eddy, 1992).

Suspended solids concentration time profiles for the conditions of the earlier tests appear in Fig 3.18, which clearly shows that the solids transferred from the outer to the inner tank were reduced very substantially in the presence of Lytag. The reduction was from 350mg L^{-1} to around 25mg L^{-1} over the entire test period, comparable with the results obtained from Vertical Axis Wind Turbine powered aeration tests. When these results are compared to those obtained from tests without Lytag it can be seen that with Lytag suspended solids carryover was reduced by 60%, from some 100mg L^{-1} down to some 40mg L^{-1} at the time of transfer. One concern from these results is to what extent the solids decline is due to precipitation into the Lytag packed bed, as this will be associated with subsequent removal and will indicate the sludge removal intervals. It is noted that some measure of solids retention actually in the Lytag should have been made to quantify these sludge removal intervals, but unfortunately they were not.

3.4. DISCUSSION

Clearly the results from these tests should be looked at with two criteria in mind. Firstly, we should ask whether the aim of the packed bed arrangement to retain the heterotrophic flocs in the outer annulus to prevent nitrifier inhibition was met. Secondly, one should also check whether there was any performance increase with regards to both BOD reduction or ammonia oxidation by the use of this packed bed arrangement. From the results of both the intermittent and constant aeration experiments it is clear that competition between the bacterial species for dissolved oxygen was removed under semi-continuous operation. This was of course only true if the previously established criteria of BOD levels in the nitrifying zone not exceeding around 100mg L^{-1}, were maintained. In fact the lag periods found in this system, under intermittent aeration and semi-continuous operation, were very similar to those obtained under batch conditions (chapter 2) at less than 1h. This therefore means that the problem of floc carryover had been significantly reduced, if not eliminated by the use of the packed bed, although the question of bio-film sloughing off the bed matrix has not been considered in detail.

When the performance of this packed bed system is compared to the one without Lytag, and more established processes such as a trickling filter that also utilise a packed bed, the results indicate that discharge limits (40:20:10) could be easily met. When these performance data are looked at more closely it is clear that the BOD removal rate of 25mg L^{-1} h^{-1} (during the aerated periods) of the packed bed WATER system is

comparable with the 10-15mg L^{-1} h^{-1} achieved in intermediate flow trickling filters (Christoulas *et al.*, 1990). It was noted however that the starting BOD concentration of around 400mg L^{-1} is approximately double the typical effluent strength that reaches treatment works, and could have a contributory effect on the increased BOD reduction rate. The higher strength was used so that the system could be tested under the severest of conditions, to give some indication of its robustness. The ammonia oxidation rates were also very favourable at up to 5mg L^{-1} h^{-1}; this compares very favourably again with trickling filters where ammonia oxidation rate of 2-3mg L^{-1} h^{-1} has been achieved at the slowest throughput rates of less than 3.5m^3 m^{-2} d^{-1} (Metcalf and Eddy, 1992).

The reduction in suspended solids was the only area where the results raised concerns over plant performance, and centred on the precipitation of solids in the matrix. When the results are looked at more closely it is clear that some 50% (100-150mg L^{-1}) of the solids were retained within the substrate. This raises questions on whether the matrix will become blocked and will require cleaning. A further concern with this apparent accumulation of solids is whether or not the bed would become anoxic/anaerobic. An error in the protocol meant that accurate studies and measurements of the solids retained in the matrix were not made, so deductions cannot be made as to what extent solids will become a problem. A positive point that was noted from this apparent anoxic/anaerobic layer was the apparent small amount of denitrification that occurred within it.

The apparent success of this packed bed system did however raise questions on whether it was acting as an aerobic support, or merely as an anaerobic/anoxic filter. The reason for this was that the slow diffusion of dissolved oxygen across the substrate would not

allow more than the top few cm of substrate to be aerobic (Bailey and Ollis, 1988; Metcalf and Eddy, 1992). The beds' oxygen diffusion limits would also not be helped by the abundance of solids (Bass *et al.*, 1976) that was indicated by our results (over 50% removed/precipitated by matrix). This theory is backed up by other quantitative work (O'Shaughnessy, 1994) on the matrix layer which showed that the dissolved oxygen did not reach beyond 2-3cm into the Lytag, thus the rest of the matrix should be considered anoxic/anaerobic.

3.5 CONCLUSIONS

This work clearly shows that once the heterotrophic flocs were retained in the outer annulus using a packed bed then there was no competition problem between the bacterial species for the dissolved oxygen, even under semi-continuous feed and intermittent aeration conditions. The use of a packed bed can therefore potentially provide a solution to solids carry-over. However, as only the top few cm of the matrix are likely to be aerobic it also begs the question of whether a suitable mobile substrate could be found which would remove this oxygen transfer limitation. If this mobile substrate were found, then what could the packed bed be used for? In the previous tests, we showed that a healthy anoxic population had colonised it, and we confirmed this by the observed reduction in nitrate concentrations. Possibly the packed bed could therefore be used as a pre-treatment, though it is not entirely clear what benefits will be conferred on the system. This is fine in principle, but still begs the question arising from potential solids

precipitation mentioned earlier. Clearly these questions will still include queries on blockage, on cleaning requirements and downtime, and all will need eventual answering.

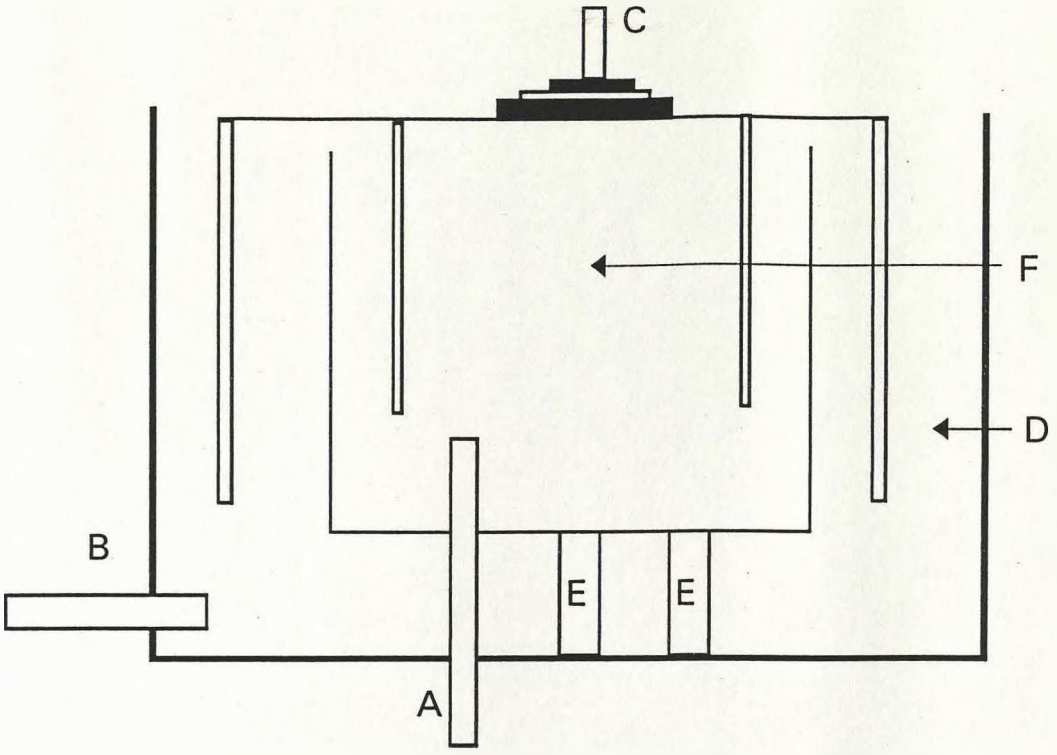


Fig 3.1 Half metre pilot plant unit. A= treated effluent outlet; B= Effluent input pipe; C= Bearing; D= Tank; E= Inner tank support and sludge return point; F= Inner tank

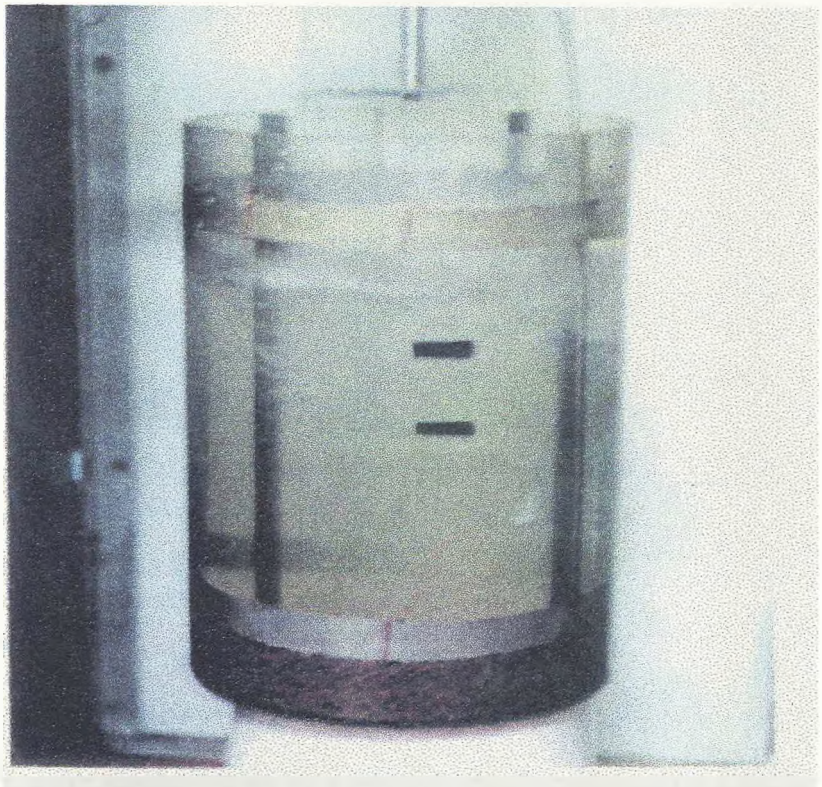


Fig 3.2 Shows a sample of the Lytag (power station slag) used in the experiment. After being sized (10mm) and washed.

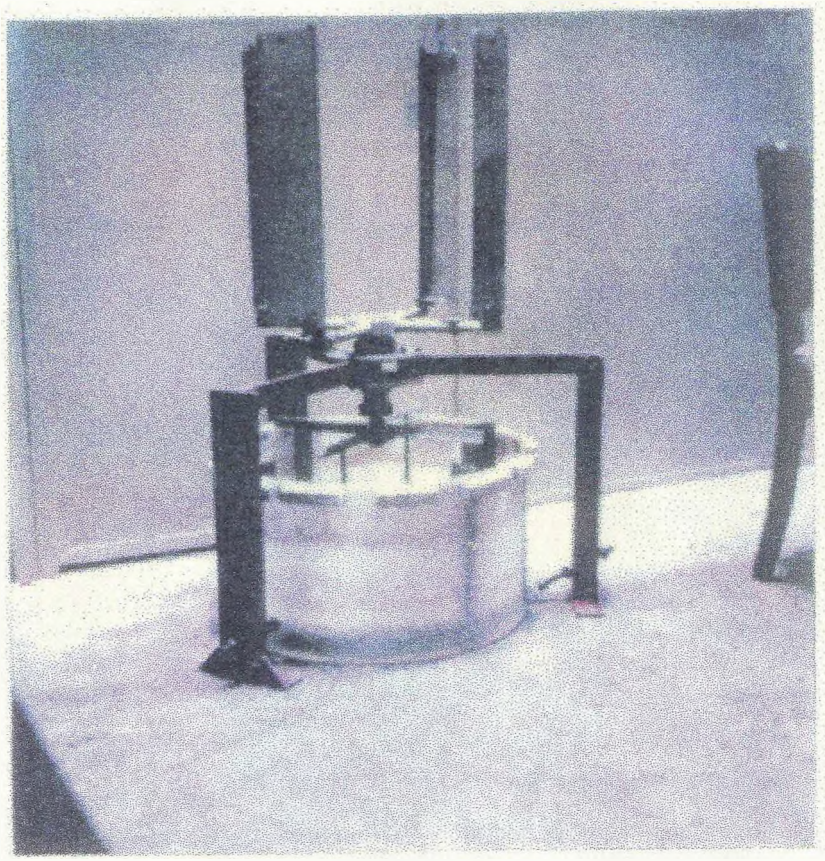


Fig 3.3 shows the Lytag packed bed in situ in the pilot unit, before it was filled.

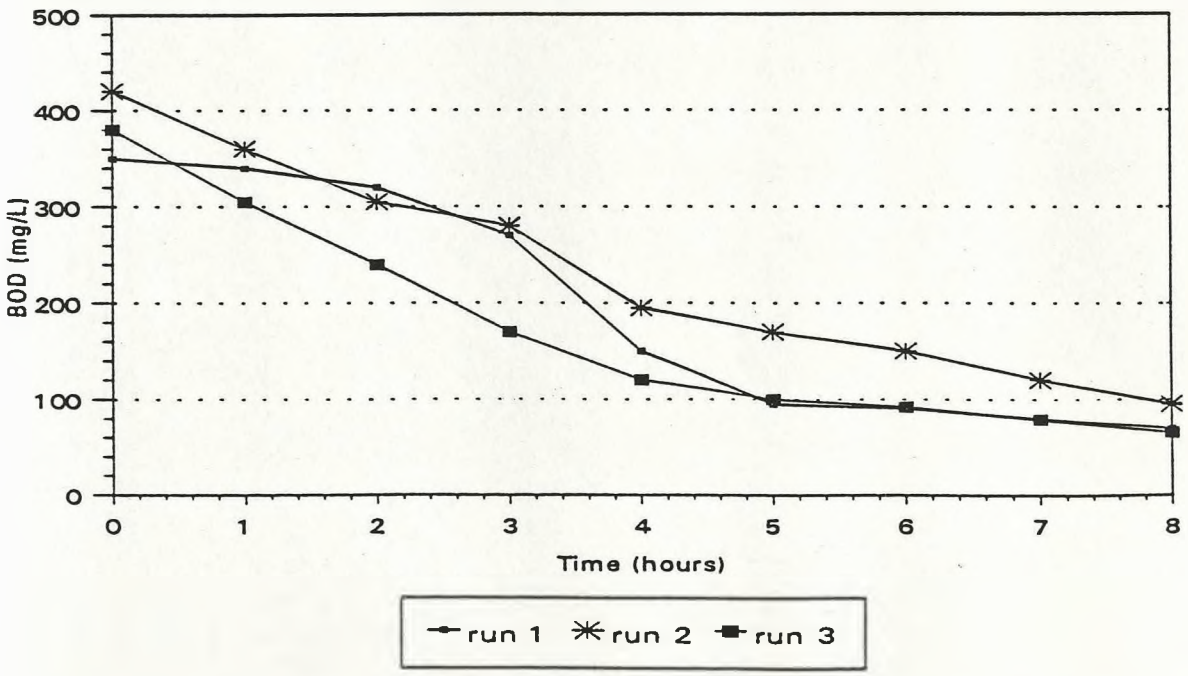


Fig 3.4. BOD removal with maintained moter aeration. System employed a packed bed support matrix (Lytag). Sample data sets after 7, 12 and 15 days

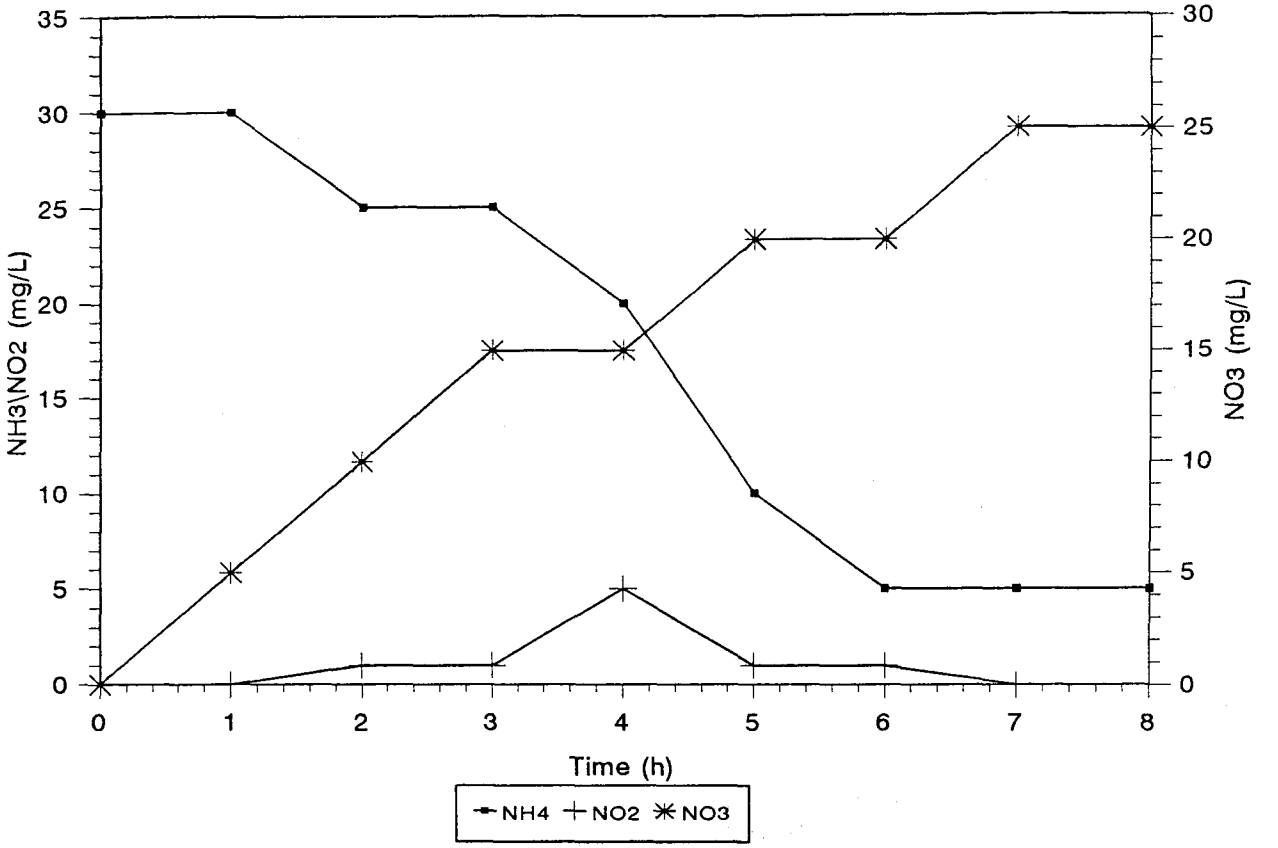


Fig 3.5. Conditions as for Fig 3.4 nitrogen balance over 8h period day 7

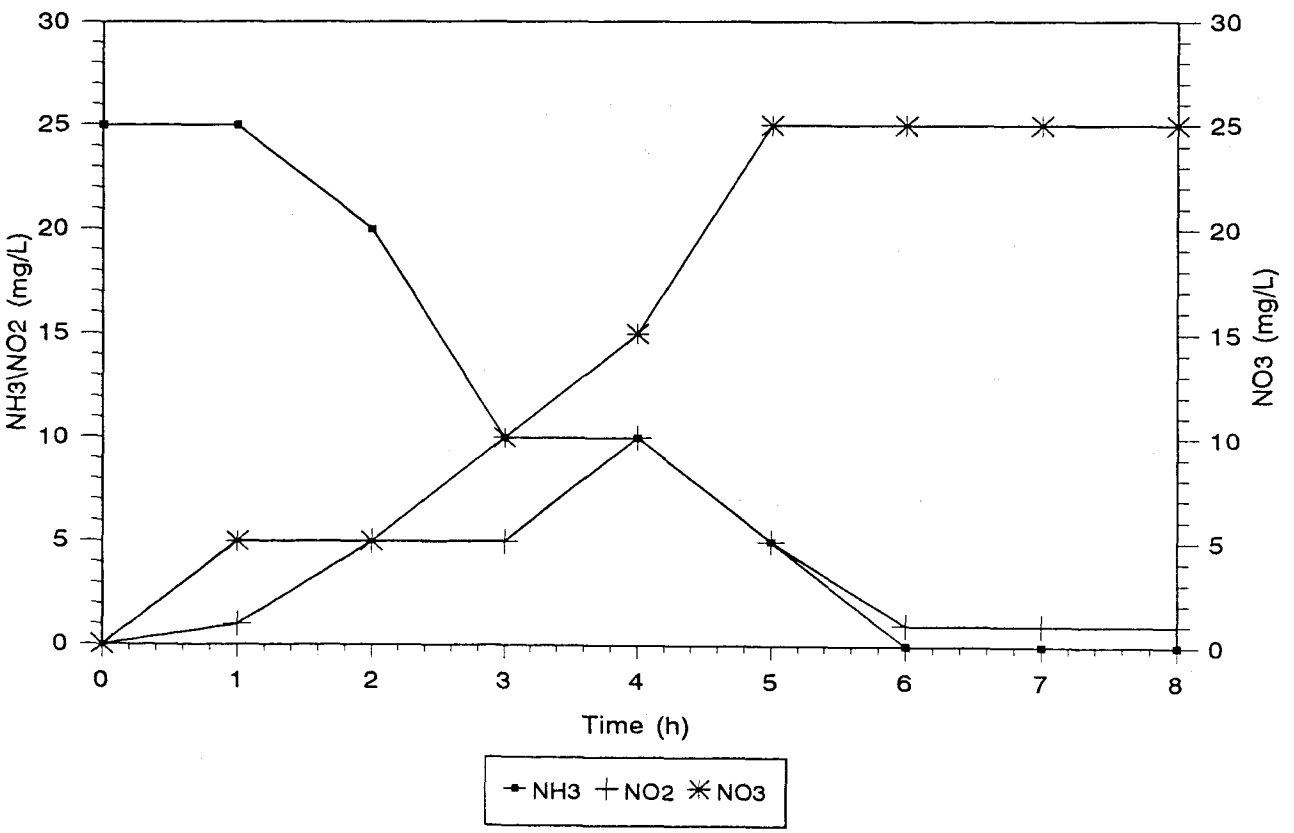


Fig 3.6. Conditions as for Fig 3.4 nitrogen balance over 8h period day 12

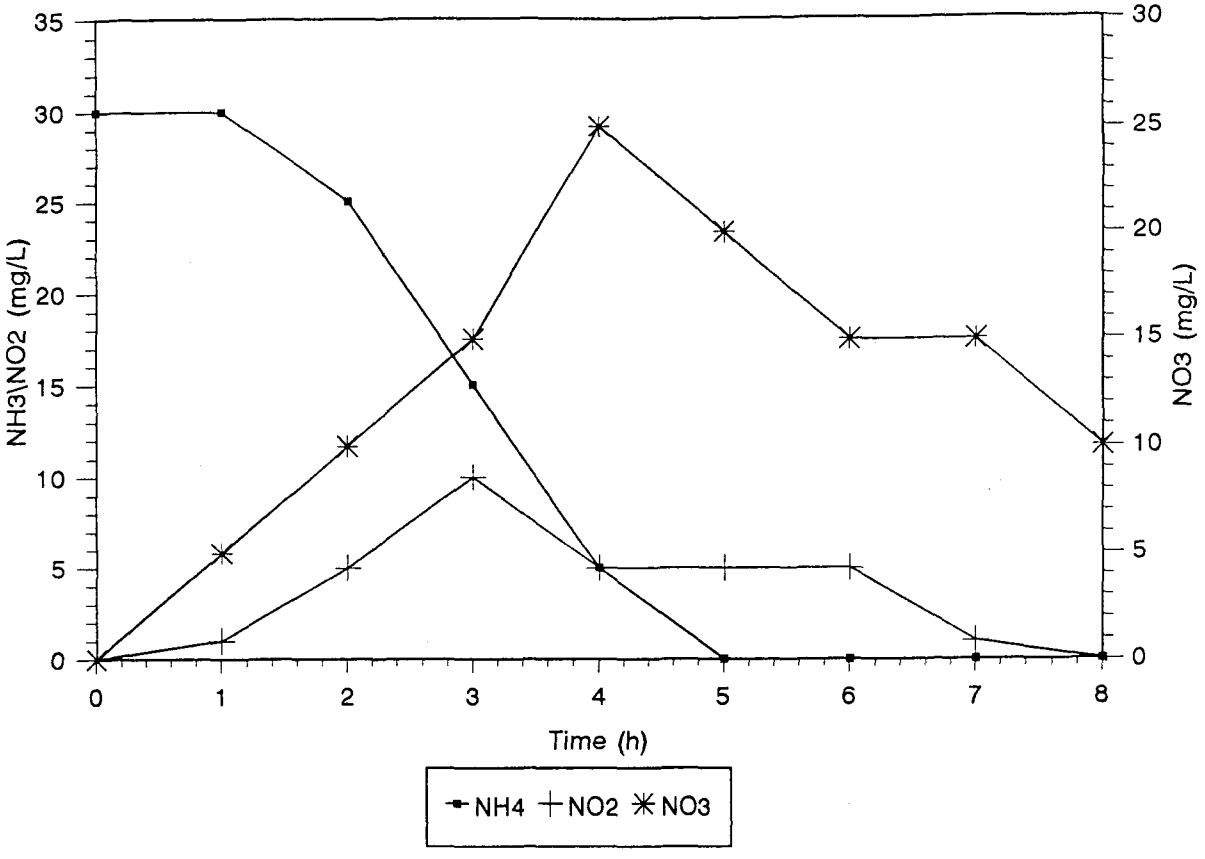


Fig 3.7. Conditions as for Fig 3.4 Nitrogen balance over an 8h period, day 15

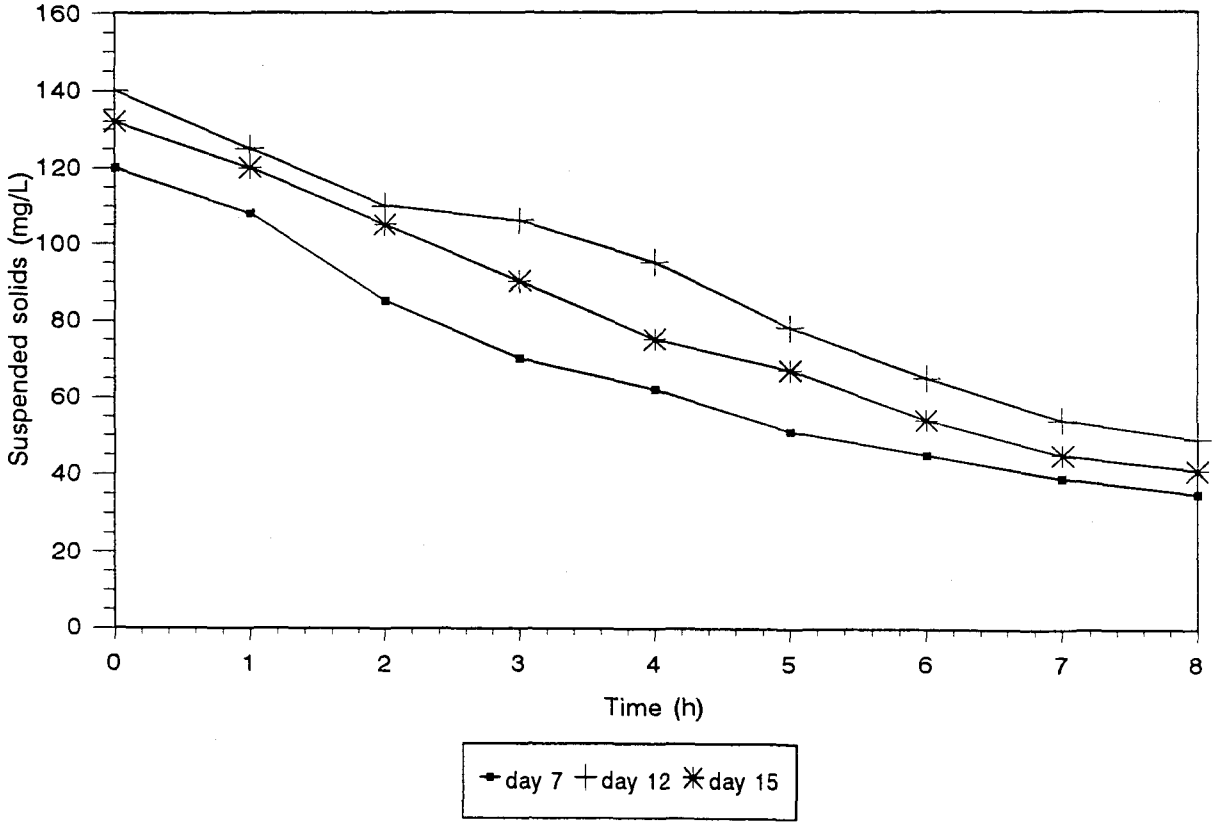


Fig 3.8. Suspended solids concentration versus time conditions as for Fig 3.4, starting concentration was approximately 350mg/L

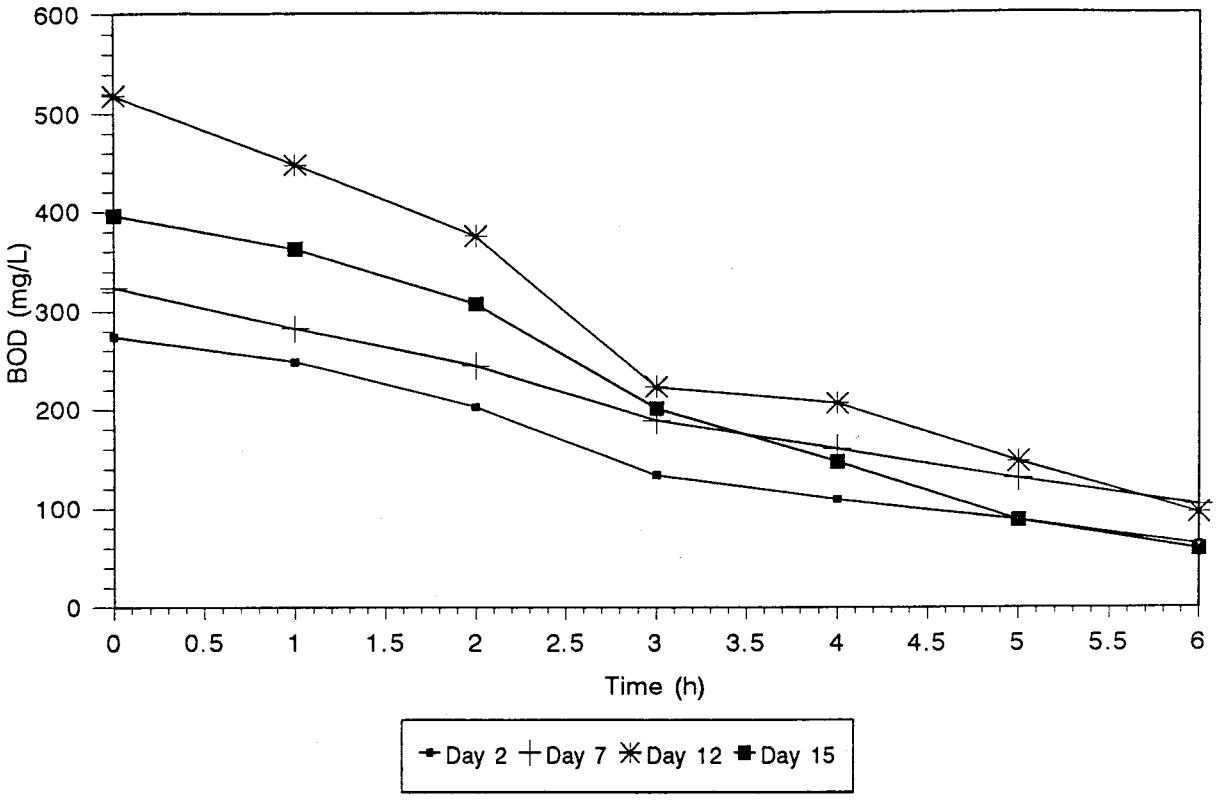


Fig 3.9. BOD removal with maintained wind-powered aeration System utilised a packed bed support matrix. Sample data sets after 2, 7, 12 and 15 days.

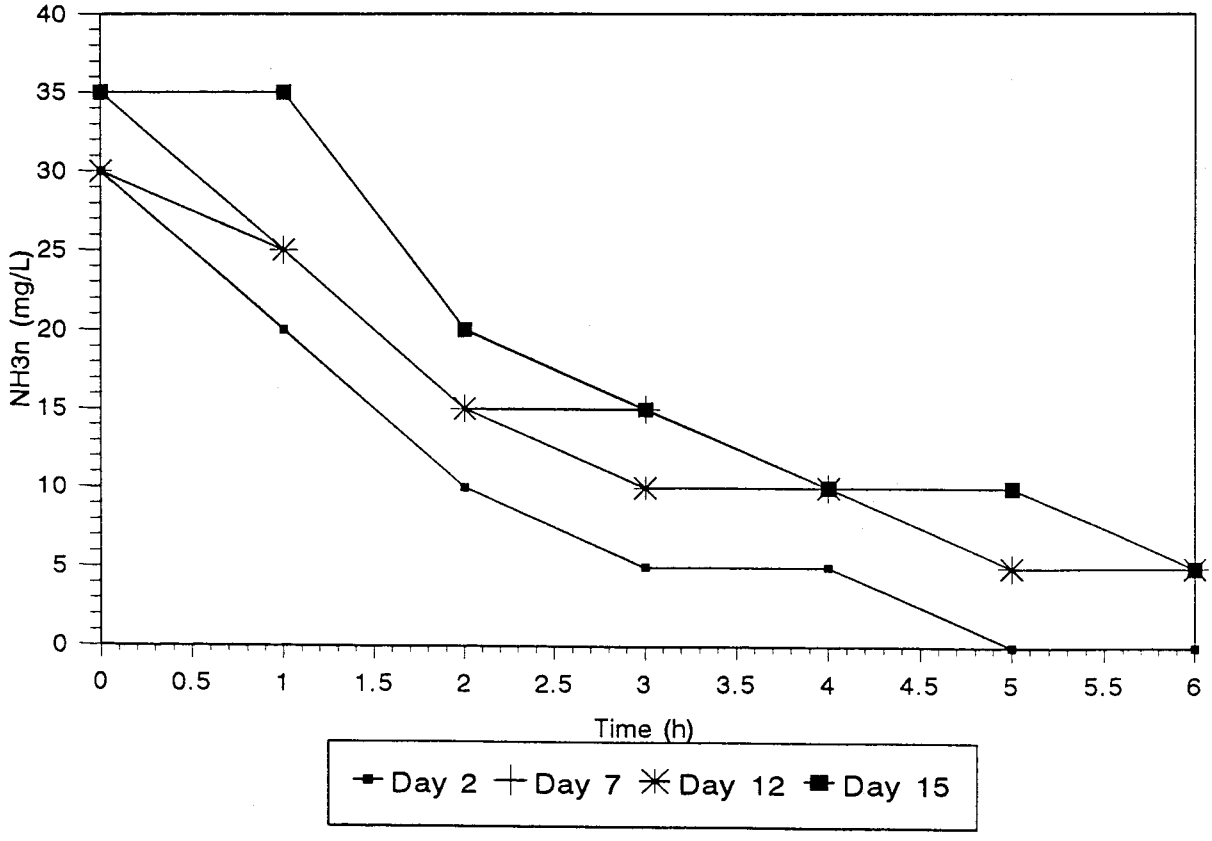


Fig 3.10. Conditions as for Fig 3.9 ammonia oxidation profile constant aeration over 6h period for days 2,7,12 and 15.

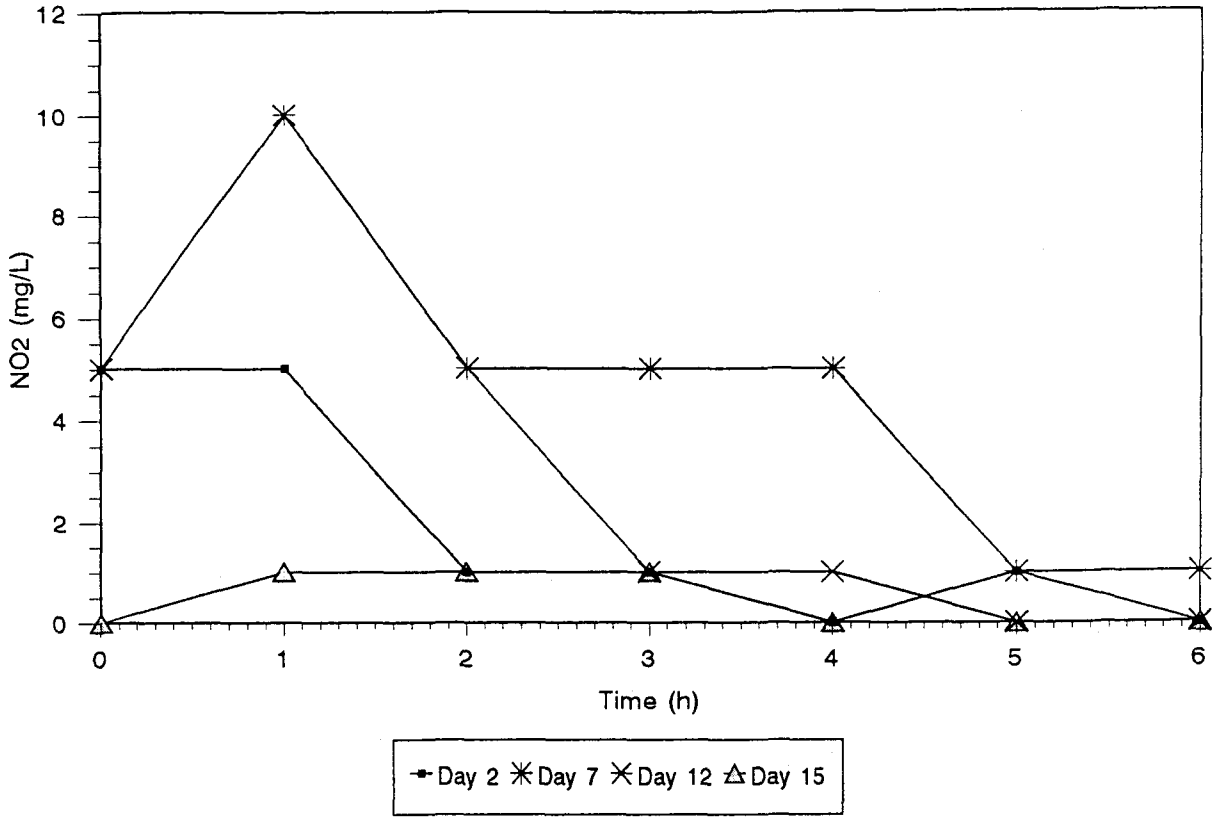


Fig 3.11. Conditions as for Fig 3.9 Nitrite time profile over a 6h period for days 2, 7, 12 and 15.

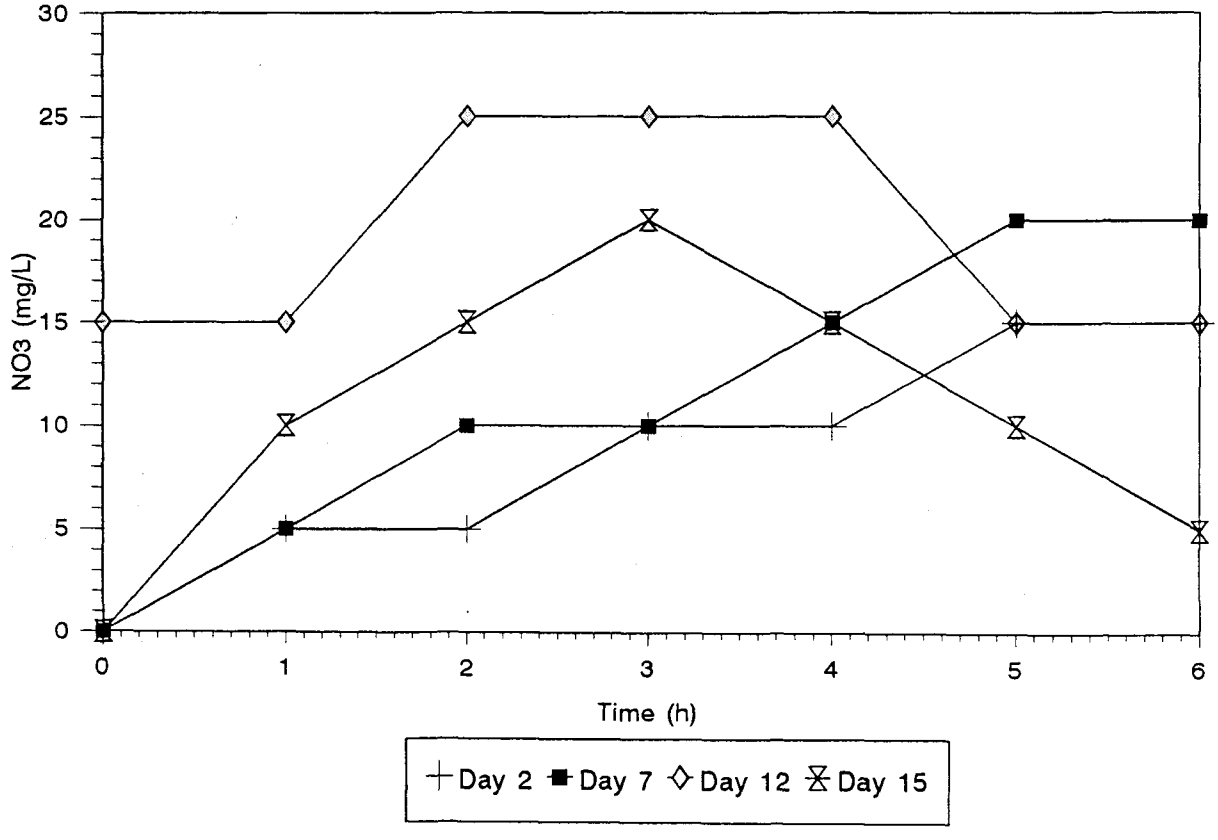


Fig 3.12. Conditions as for Fig 3.9 Nitrate time profile for 6h period after new effluent addition for days 2, 7, 12 and 15

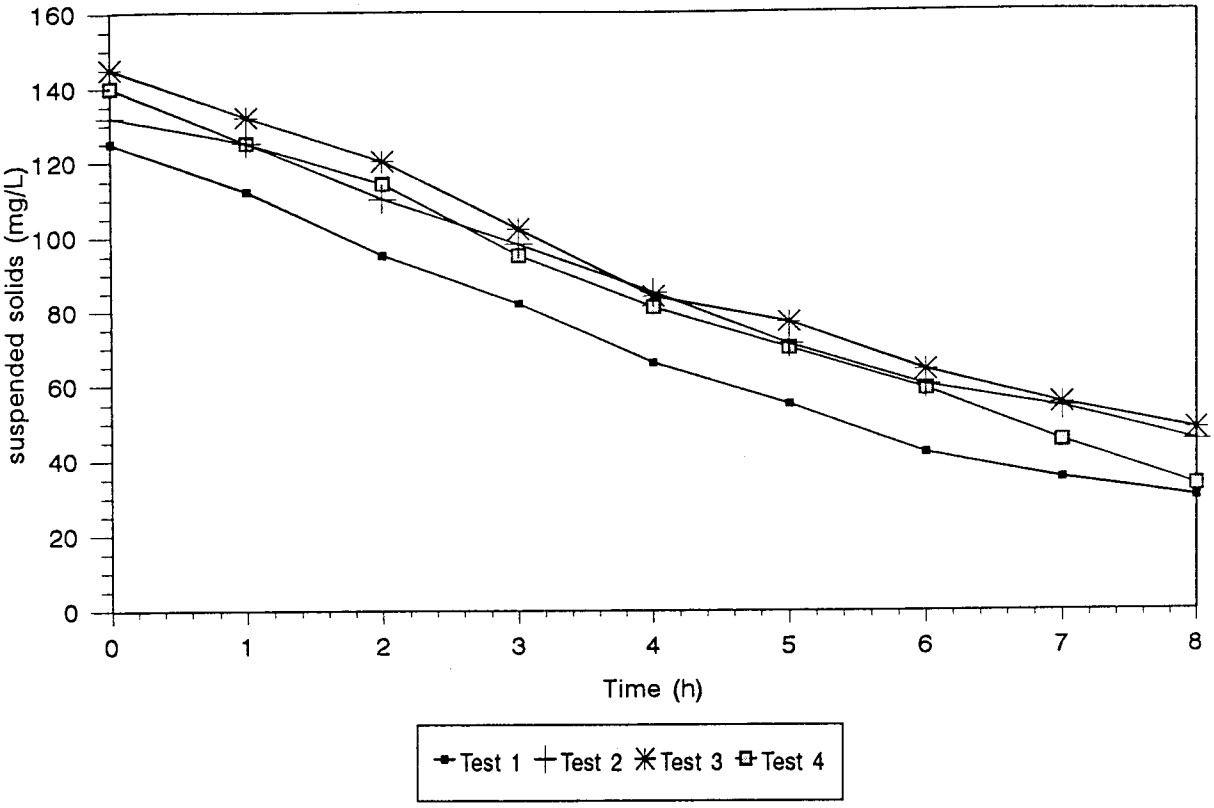


Fig 3.13. Conditions as for Fig 3.9 Suspended solids time profile for 6h. Period after effluent addition for days 2, 7, 12 and 15. starting concentration was around 350 mg/L

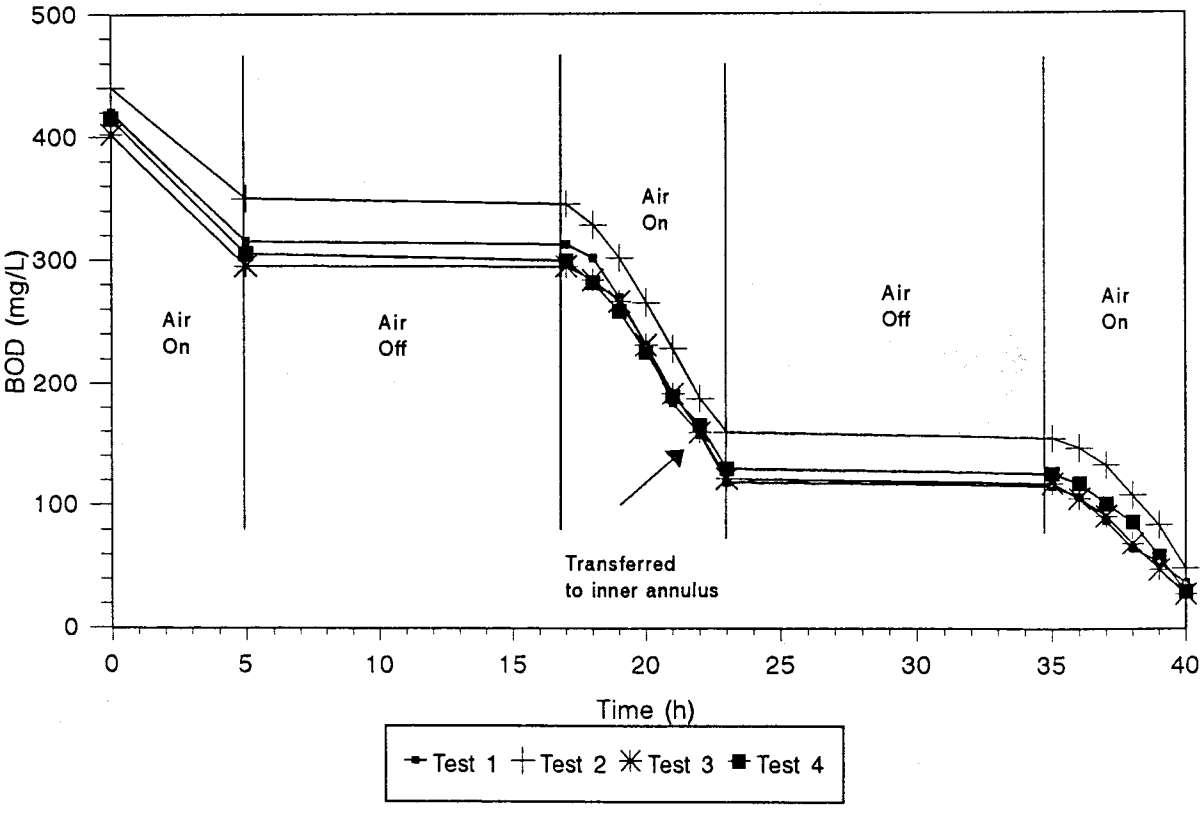


Fig 3.14. BOD concentration time profile for 4 consequtive motor powered intermittent aeration experiments. Intermittent periods were 12h as shown

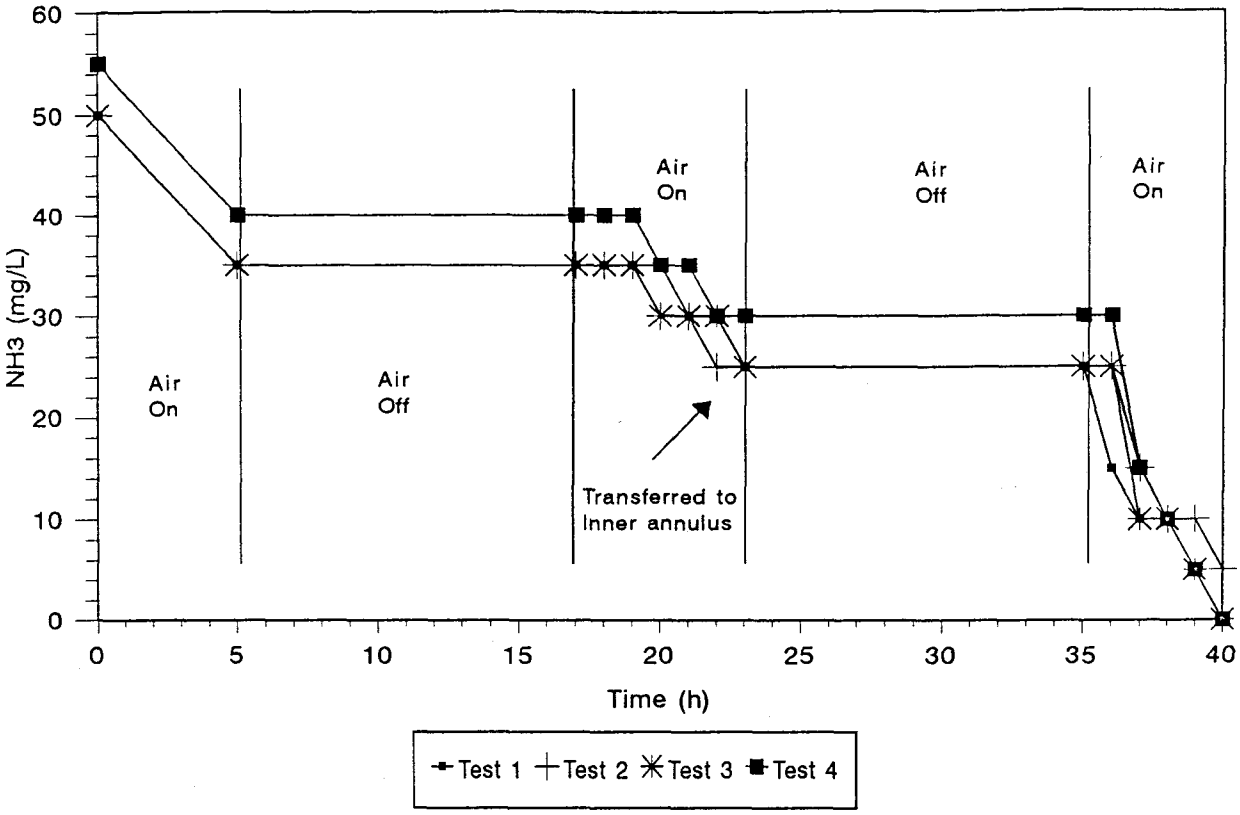


Fig 3.15. Ammonia oxidation time profiles under intermittent aeration for consequtive tests, conditions as for Fig 3.14.

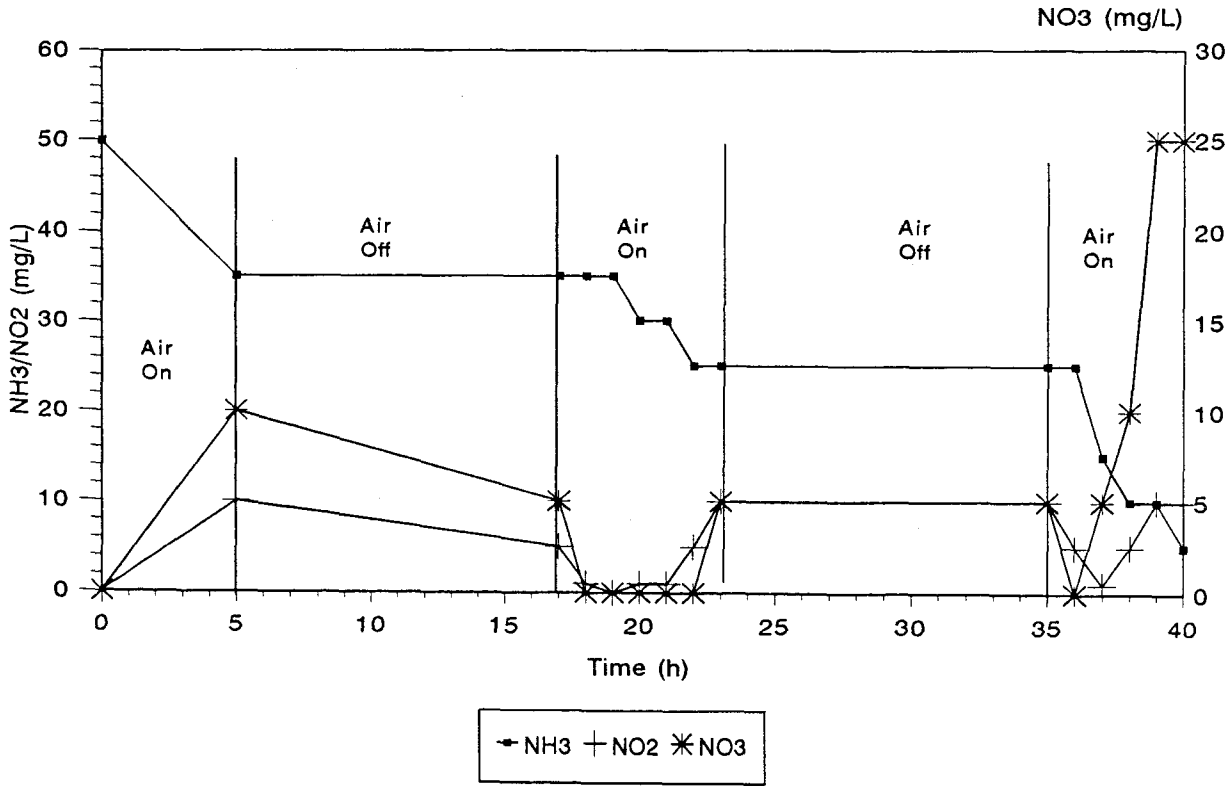


Fig 3.16. Time profile for nitrogen balance for test 2, under the same conditions as fig 3.14

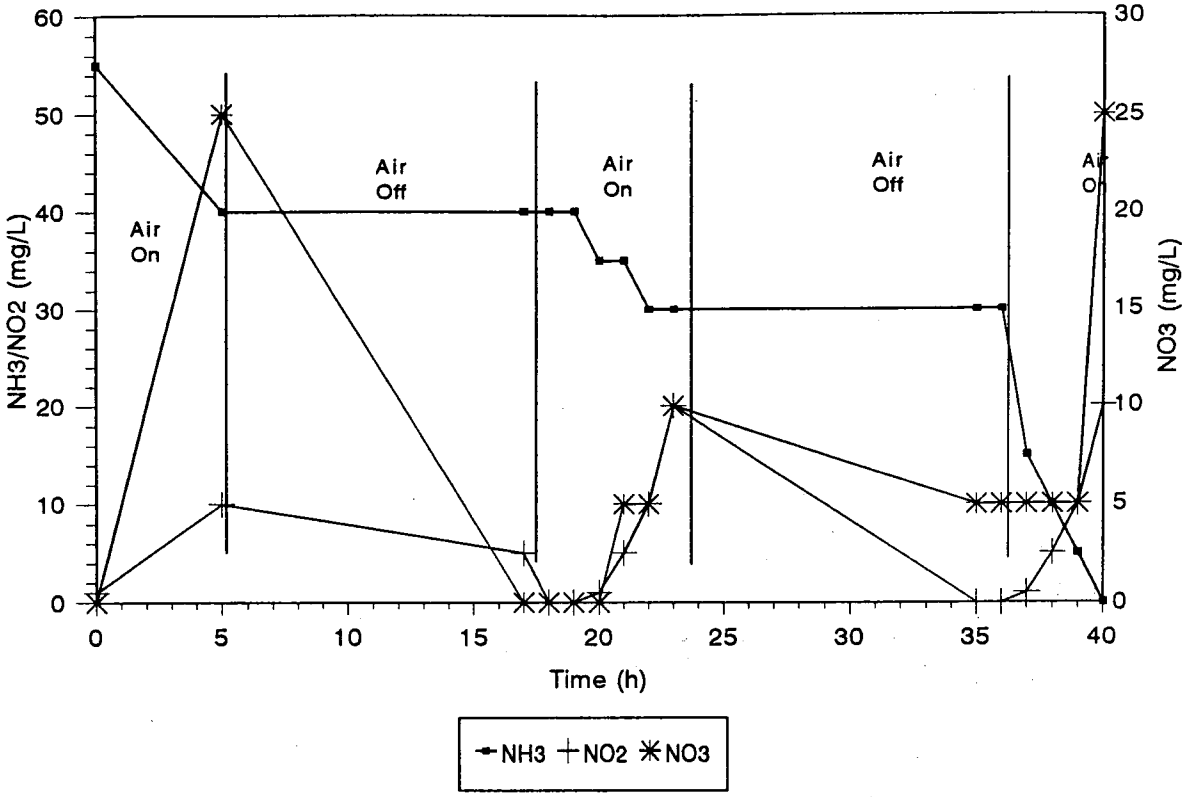


Fig 3.17. Time profile for nitrogen balance for test 4, under the same conditions as Fig 3.14

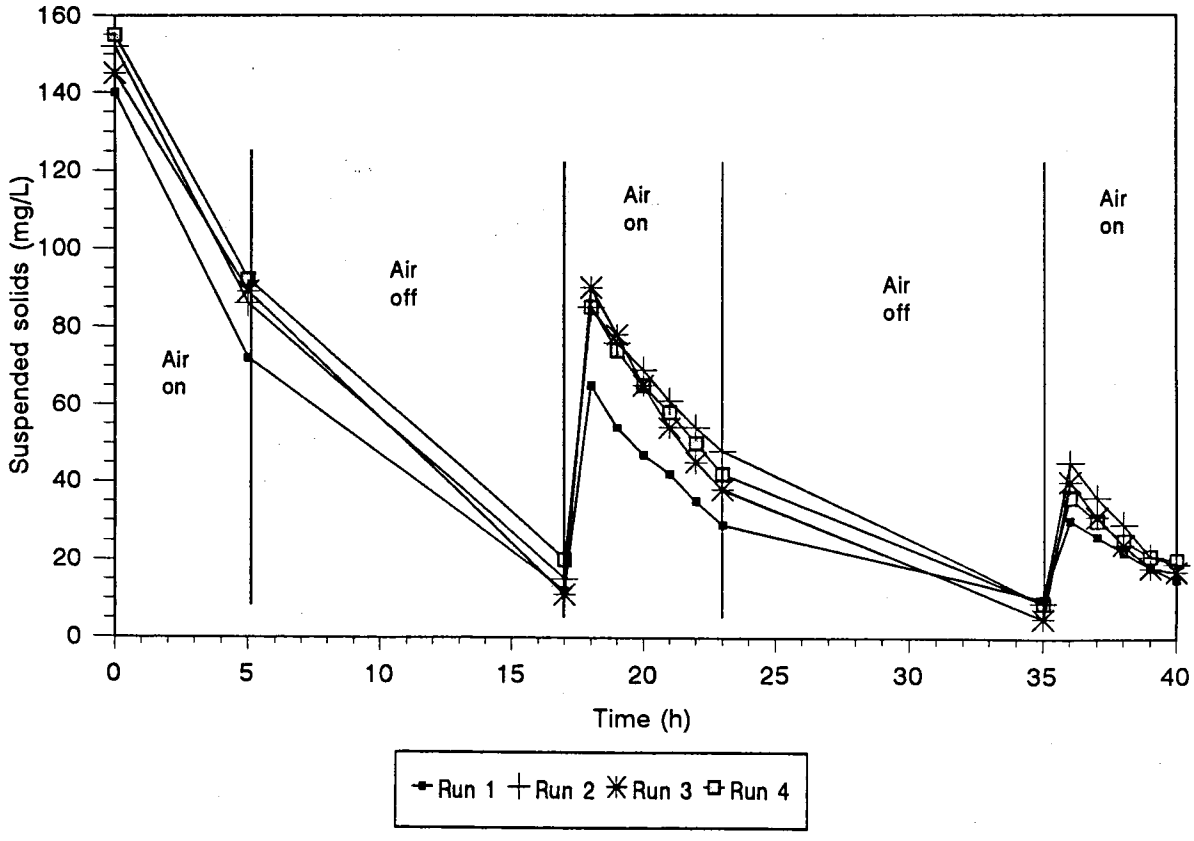


Fig 3.18. Conditions as for Fig 3.14, suspended solids time profile for 40h. Intermittent periods were for 12h as shown. Stating solids level was 350mg/L

CHAPTER. PACKED BED SUBSTRATE OPERATIONAL AND BIOLOGICAL FACTORS.

SUMMARY

Support media for confining biomass within an effluent treatment system and increasing sludge age have been used for decades, to improve plant performance. However, such systems also have disadvantages, such as reduced efficiency due to solids accumulation within the matrix, downtime for de-sludging and cleaning etc. Of particular interest here is the use of Lytag (open porous ash residue) as the support matrix, and whether it will suffer from the problem of clogging. As demonstrated previously (chapter 3) Lytag confers the advantage of biological stabilisation and segregation on the two main populations (heterotrophic and nitrifying organisms). The issue addressed in this chapter is the accumulation of solids in such matrices over a protocol representative of a five-year period of operation in a half-meter test rig. It was noticed that there was a "buffering effect" on the aerobic population by the matrix, whose activity was expected to suffer due to the high strength effluent that was being used. It is suspected that this was due to the reduction in effluent strength by the matrix, similar to that experienced in trickling filters.

4.1. INTRODUCTION

It is now universally recognised that solids support matrices confer substantial advantages in performance and operational stability on wastewater treatment plant. The presence of the support substrate provides for a considerable increase in throughput rate for oxidation and buffers the system against excursions in influent quality particularly with regard to shock loads. The adverse impact of influent excursions on trickling filters was recognised forty years ago for example by Lawton and Eggert (1957). They noted that such plants run best for maintained steady quality of effluent with excursions to either high or low strength reducing the oxidation rates. Ludzack noted similar observations and Noran (1965) on activated sludge plant and also on anaerobic digesters subjected to shock loads, with significant excursions here being in the range of two to three times the steady influent concentration. Later work by Stover *et al,*. (1985) investigated various shock loading effects on the performance of bench scale fixed-film and completely mixed suspended growth anaerobic systems. He showed that both systems were able to accommodate a 24h doubling in organic load without any significant problems. For periods greater than this, performance suffered. Also if the process was not fed, for periods up to 2 weeks only, 24h was needed to re-establish operational conditions. Process stability was however, affected by pH drops (below 6.5) and small drops in temperature (Goronszy *et al.*, 1987; Stover *et al.*, 1985).

Goronszy *et al.*, (1987) pursued these developments incorporating calcium carbonate as cheap reactive support medium accommodating high strength waste (BOD 2000-3000mg L^{-1}, suspended solids 200-500mg L^{-1}) and increased biomass density and solids retention time. Under such conditions, the biomass increased by a factor of 2-3, within a hydraulic retention time of 2 days, reducing to 1 day after 6 weeks. Solids reduction was enhanced in the presence of calcium carbonate substrates (by 50-60%). Whilst a broad perspective on the scientific understanding and practical quantitative advantages of these pre-treatment stages remains to be established in detail there is widely documented evidence of the advantages that they confer.

On the other hand the major drawback of support matrices is the risk of pore blocking by solid particle accumulation which gives rise to flow channelling. Flow channelling results in non-uniform residence times through the system with adverse effects on uniformity of treatment leading, in turn, to reduced capacity of the system (Miyahara *et al.*, 1994; Metcalf and Eddy, 1992). Moreover, in more extreme cases, solids accumulation in nominally aerobic processes can result in anoxia, thereby further reducing oxidation rates and hence also throughput rates. This is recognised to be a particular problem in high rate trickling filters (WPCF, 1988; Bruce, 1981).

The solids fraction contains the constituents with the highest oxygen demand and which therefore can best be handled by screening processes prior to aerobic treatment. For some considerable time it has been standard practice to remove suspended solids in excess of 1mm using screens or primary filters (Helfgott *et al.*, 1970) as a preliminary

treatment. Such solids typically include plastics, pebbles and other non-biodegradable material. Screens have a low energy requirement, and the recent improvements in design and performance of screening materials mean they are now capable of removing particles down to 100μm (reduce BOD levels by 25-35%).

As can be seen in the table in Fig 4.1 suspended solids may range from 100-350mg L^{-1} or so of which up to 75 and up to 275mg L^{-1} respectively comprise fixed and volatile fractions. Amounts of suspended solid are usually between 0.33-0.5 of the amounts of dissolved solid such that the total solids range from 350-1200mg L^{-1} according to descriptive range from weak to strong influent. As regulatory constraints on discharge quality limit solids content to between 20-40mg L^{-1} (Roberts, 1995), clearly a substantial portion of the work involved in total quality restoration is entailed in handling the solids. Following coarse screening to remove materials in excess of 1mm or so, a gravitational settling chamber is normally employed, typically to extract up to 25% of the remaining total solids (Metcalf and Eddy, 1992). This fraction can be boosted by the addition of flocculating agents, most commonly aluminium sulphate or manganese sulphate, to remove up to a further 15-25% of the original load, dependent on local conditions (Karlsson and Smith, 1989).

This increased removal means overall plant size can be reduced, or throughput time increased, with associated cost savings either in land required or volumes treated (Metcalf and Eddy, 1992). However, this is still negated once there has been solids accumulation in a packed bed, to a level where it has affected the plant's performance.

Once this level has been reached it obviously needs to be removed. This would normally involve either back washing or some form of manual or mechanical removal. Back-washing involves passing a liquid through the bed in the opposite direction to normal flow and at a higher flow rate (to a level where the bed becomes fluidised or at least agitated) thus causing bio-film sloughing by particle attrition (Metcalf and Eddy, 1992). If back washing is needed, the plant cannot be used during this time. Further plant is then needed to cover this downtime, thus increasing cost. The cost of the equipment required to backwash also adds to the expense, so back-washing and solids accumulation are avoided if possible.

The reduced amount of dissolved oxygen reaching the inner layers of a packed bed has long been documented (Beccari *et al.*, 1992) and the lack of aerobic bacterial activity noticed. The reason for this is the diffusional resistance's, particularly related to oxygen internal diffusional problems over a bio-film. These cannot be neglected when evaluating the overall kinetics particularly when distances over 100μm are involved (Beccari *et al.*, 1992). Varying oxygen transport effects on the mean cell retention times and autotrophic/heterotrophic competition for dissolved oxygen can combine to give apparent nitrification limiting dissolved oxygen concentrations ranging from 0.5 to 4mg L^{-1} (Stenstrom and Song, 1991) in dissolved oxygen limited situations such as packed beds. This problem is exacerbated by solids accumulation as the carbon-consuming heterotrophs scavenge the majority of the available dissolved oxygen. Further problems also occur due to reduction in oxygen solubility in wastewater due to its impurities, which decreases as solids and BOD levels increase (Bailey and Ollis, 1992).

The study conducted here concentrated on batch operation using a half metre model of the stirred tank WATER system over a period of five weeks to evaluate both oxidation and solids performance in the presence of substantially elevated influent solids concentration with a packed bed matrix of Lytag. The following sections in number order describe the materials and methods, results, discussion and conclusions.

4.2. MATERIALS AND METHODS

The half-metre pilot plant (Janes and Thomas, 1992), comprised an outer tank 500mm diameter, height 330mm, partially filled with a packed bed of Lytag 85mm deep. Into this a 490mm-diameter distribution plate (5mm thick) was added, that had 4 to 5 x 5mm holes per $5cm^2$, all as shown in Fig 4.2. The inner tank (see Fig 4.2); diameter 340mm, height 220mm supported on three legs (see Fig 4.2) was provided with 13 x 5mm diameter holes symmetrically distributed around its circumference as transfer points from the outer annulus to the inner tank. The latter was also equipped with exit pipe and sludge return ports. As reported previously aeration was delivered by two snorkel tubes (triangular cross-section) each of length 170mm (from base to air intake hole) and diameter 10mm, mounted from above on a constant speed rotor arm powered by a 2kW motor. There were two such tubes for each tank.

The inoculation procedure (again as described see chapter 3) involved priming the system with 36L of primary screened sewage introduced via the distribution plate supplemented with 4L of activated sludge in 10:1 ratio following standard industry practice (Stanibridge, 1977). All materials were obtained from Severn Trent water reclamation works, Minworth, Birmingham. A five-week period of stabilisation then followed with steady aeration and periodic addition of primary screened sewage as nutrient. BOD monitored in the outer annulus dictated the later additions; in particular 6-8L was added whenever the BOD fell below 70mg L^{-1}.

The first stage investigated solids accumulation under constant hourly addition; the second stage involved three hourly additions with maintained aeration, to see if aerobic activity was impaired under high BOD loading. Following incubation and stabilisation the first series of tests was conducted with hourly loadings of 4L of influent (BOD 300-350mg L^{-1}, ammonia 45-50mg L^{-1}) equivalent to 6h residence time (24L bulk, 30L including Lytag) but with an exceptionally high maintained influent solids content of 20,000mg L^{-1}. This is 50 times weaker dissolved pollutant 15 times stronger suspended solids than the reference concentrations adopted by Metcalf and Eddy (1992), and was maintained in suspension prior to entry using a shaft powered mixer. The liquor and solids were sourced from the sewage works primary settling tanks and primary screened sewage in proportions delivering the desired total solids content, but were un-aerated prior to being added. The inner and outer tanks were together aerated constantly using four snorkel aerators rotated at 150rpm, which delivered around 1kgO_2 KWh^{-1} (Fowler and Bradley, 1994). Again it should be noted that power draw from this motor was approximately 5-20W (increase due to higher speed), well below the motors maximum.

The total duration of each set of these extremely severe tests was 8h (five per week over five weeks) and so provided a preliminary indication of the accumulation trend within the test system. This system was run constantly over a five week period during which time some 200 effluent additions were made (8 per day five days a week), thus giving a total solids input of approximately 16kg. In this report two days results from each one of the five weeks are shown, as results were broadly consistent throughout. The only exception to this was the final week, so the complete five days results are shown and discussed. In addition to hourly monitoring of the suspended solids concentrations, BOD, ammonia, nitrite and nitrate were also recorded to gauge the overall stability profile of the plant under heavy suspended solids loading conditions, these samples (50ml) being taken from the outer annulus.

The second tests were conducted following the same procedure as the first series except that 3h intervals between were adopted. To compensate for the increase in hydraulic residence time the volume of effluent added was increased from 4L to 5L, changing the residence time from 6h to 14.5h; the SS content of the effluent remained unchanged at 20g l^{-1}. Also the duration of each days additions altered from 8h to 9h, with the overall length of the test being 5 weeks. During this time some 75 additions were made, giving a total solids loading on the system of 7.5kg. Otherwise the procedures adopted for this series were the same as the first. In this report the results of two days from each week will be presented as all results obtained over the entire five weeks were broadly

consistent. Samples (50ml) were taken of the semi-treated liquor in the outer annulus every hour, and both before and after the addition of new effluent.

It should also be noted that the three-hourly addition tests followed straight on from the hourly addition work with only some minor on-line cleaning and a further two weeks re-stabilisation period. The on-line cleaning involved the agitation of the Lytag layer with a metal rod, whilst a weak effluent solution (BOD less than 100mg L^{-1} and ammonia negligible) was continually passed through the system, which caused a considerable amount of solid to be removed from the packed bed. Following this the system had primary screened sewage passed through it (same concentrations as above) for a further two weeks to allow the system to stabilise again, after which new effluent additions (6-8L) were made as before.

BOD was estimated by the Winkler method (Winkler, 1982; see chapter 2), and ammonia, nitrate and nitrite were tested using Quantofix Test Strips (Camlab), which are sensitive down to 1mg L^{-1} (nitrite) and 5mg L^{-1} (ammonia and nitrate). The sensitivity on all three types of test strips was increased as outlined in the previous chapters. Suspended solids were measured by suspending the test sample by shaking, then taking a known volume of the test sample (25ml) and allowing it to settle out, to give the sludge volume. This sample was then filtered through a pre-weighed filter paper and dried at 105°C overnight to remove moisture. The filter paper was then re-weighed and the weight of suspended solids calculated.

4.3. RESULT

4.3.1. Stabilisation period.

During the five-week stabilisation period, ammonia and BOD levels were monitored for feeding purposes. These time profiles are shown in Figs 4.3 and 4.4. From these graphs it is clear that the rate of BOD removal and ammonia oxidation increases with time, which is expected as the system stabilises and the sludge age increase. The rate of BOD removal increases from around 5mg L^{-1} h^{-1}, at the start of the test to around 9-10mg L^{-1} h^{-1}, at the end, which is lower than previous results (chapters 2 and 3). However samples were taken only once a day so these results give only a trend. The same pattern of results can be seen for the ammonia concentrations (Fig 4.4) which show an increase in oxidation rate from around 1mg L^{-1} h^{-1}, to 2mg L^{-1} h^{-1} over the five-week period. This again is lower than the rates found previously (chapters 2 and 3) and again can only be taken as a trend due to the infrequency of sampling. It should be noted that new effluent additions were increased from every three days to every two due to the increase in both ammonia oxidation and BOD removal rates after 20 days.

4.3.2. Hourly additions

Figs 4.5 to 4.15 show respectively show suspended solids, ammonia, nitrite, nitrate and BOD over an 8h period of hourly (4L) effluent additions Figs 4.5 to 4.7 show that most

of the data were at 70mg L^{-1} or less, compared with the 20,000mg L^{-1} entry concentration corresponding to greater than 99% reduction. Throughout the test period there is a slow increase in solids concentration in the samples taken; however towards the end of the test (Fig 4.7) the level of SS increases dramatically (days 31-34), with values ranging from 80-225mg L^{-1}. This is consistent with the packed bed becoming clogged to the point where solids begin to break through; however it should be noted that by the time this breakthrough is becoming noticeable (day 30 of test), over 12kg of solids have been added to the system. If the solids removal rate is calculated for an equivalent five year period, the 27mg L^{-1} h^{-1} obtained compares favourably with published work on aerated submerged fixed film reactors and trickle filters, where removal percentages of up to 90% are common (Netzband *et al.*, 1992; Harada *et al.*, 1994). This high removal percentage in the system can be explained to some degree by the extremely harsh initial conditions used, and it should be noted that in reality some of the solids accumulated could be removed by bacterial digestion over the five years.

Figs 4.8 to 4.10 record the ammonia concentrations against time for the same test days as the suspended solids graphs. These indicate that some oxidation is maintained over the first 22 days of the test, but then from day 24 onwards and particularly over the last five days there is no significant ammonia oxidation. Indeed, by the end of the test, there is an increase in ammonia concentration to around 60mg L^{-1}, which is some 10mg L^{-1} higher than the initial ammonia loading concentration of 45 - 50mg L^{-1}. The reasons for this apparent increase are not entirely clear. When the ammonia concentration over the first 20 odd days (Figs 4.8 and 4.9) is scrutinised closely, it can be seen that there is noticeable

ammonia oxidation occurring, with levels dropping from 45mg L^{-1} to around 25mg L^{-1}. This rate of approximately 2-3mg L^{-1} h^{-1} is slightly lower than the 4mg L^{-1} h^{-1} achieved in earlier work (chapters 2 and 3). This decrease can, however, be attributed to bacterial competition for the available oxygen. The reduction in ammonia oxidation from around day 20 and its eventual cessation are consistent with the breakthrough of solids (see Fig 4.7) and can therefore be attributed to overloading the system. One possibility is that when the support matrix is blocked, there is a greater possibility of surface flow and of evaporative concentration. However, one would also expect more loss of ammonia under such conditions within the matrix leading to bacterial film breakdown, with degradation of the bacteria themselves and consequent lysis and ammonia production.

All this, however, is speculation and further experimental work will be needed before the reasons for these interesting observations can be finally established. Up to the point where solids breakthrough took place, the ammonia reduction rates compared favourably with established treatment methods such as trickling filters and RBC's which reduce 80mg L^{-1} to 2mg L^{-1} at throughput rates similar to ours (Netzband *et al.*, 1992), but much lower solids loading (around 500mg L^{-1}). Clearly until solids breakthrough, ammonia is being oxidised successfully, in our equipment, in the aerobic layer.

Figs 4.11 and 4.12 show tables of nitrite and nitrate concentrations respectively over the five-week test period. Both tables show that over the first 20 days there is nitrification occurring although at a reduced rate compared to previous results (chapters 2 and 3). It

should also be noted that there was some denitrification also detected particularly over the 16h in between daily additions. This can be seen by noting the concentration at time 8 on preceding day and compare this with time 1 on the following day (i.e. day 9 time 8 and day 10 time 0). Here the increase in residence time (to 16h), suggests that denitrification requires a longer contact time, which is consistent with the activities of anoxic bacteria. However again there was a noticeable breakdown in nitrifying bacterial activity over the last 7 or so days once suspended solids levels in the outer annulus started to increase, consistent with both heterotrophic bacterial competition and the initiation of overloading problems.

Figs 4.13 to 4.15 show the BOD time profiles for the selected days under conditions of hourly effluent additions and maintained aeration. As can be seen from these graphs the results are consistent with the previous ammonia (Figs 4.8-4.10) and SS (Figs 4.5-4.7) graphs, in that over the initial 20 or so days there was significant BOD removal with values decreasing from the initial 300-350mg L^{-1} to between 145-166mg L^{-1} (days 1-12) and around 200mg L^{-1} (days 15-24). This equates to a reduction rate of between 15-20mg L^{-1} h^{-1}, which is somewhat lower than the 35mg L^{-1} h^{-1} achieved in the previous half-metre tests (Chapter 3). However under these extreme conditions such a reduction rate is still very encouraging. Over the last five days (Fig 4.15) there is again a rapid deterioration in the rate of BOD removal from 5mg L^{-1} h^{-1} in day 30 down to virtually nothing in day 34. This is again consistent with the breakthrough of suspended solids, and the subsequent inability of the biology to deal with this increased load. All of these results (Figs 4.5-4.15) indicate that up until day 24 there were no significant problems

with system operation, but after this the accumulation and subsequent breakthrough of solids and its associated higher strength effluent placed an undue load on the process biology.

4.3.3 Three hourly effluent additions

The results are in the same order as for the 1h tests and, as may be expected, broadly show increased tolerance to this loading regime; that is 33% of the total suspended solids loading rate. In particular, referring to Figs 4.16 and 4.17, we see that over three successive cycles of this regime, there is a broad consistency demonstrated here that the test tank can accommodate this loading pattern. Thus, suspended solids show not only a continuing decline in each of the 3h interval of digestion but a trend to declining concentrations with evolving time throughout the test. The final concentration at each of the new effluent additions (3, 6 and 9h) is between 20-45mg L^{-1} which would need further reduction if it was to meet the NRA discharge limit of less than 20mg L^{-1} (Roberts, 1995). When these results are compared to the hourly addition ones it is clear that the solids limit of the Lytag was not reached during this test, and that there was no breakthrough of solids. However, this is not surprising as the total solid added in this test was around 7.5kg, some 5kg less than the 12kg (30 days into test) limit that caused problems in the previous test. To reduce the solids concentration to discharge limits a further 1-2h of treatment would be required. These results do, however, compare well to those achieved in chapter 3. Indeed the 30-40mg L^{-1} final SS level achieved in those tests was over an 8h-test period rather than the 3h in the present tests. It is clear that with increased plant capacity/throughput time, NRA discharge limits could be achieved.

Figs 4.18 and 4.19 show the corresponding ammonia time lines indicating sustained oxidation from influent levels of around 40mg L^{-1} down to less than 10mg L^{-1} during each 3h period equivalent to 6mg L^{-1} h^{-1} oxidation rate, which is comparable with the rates achieved earlier (chapters 2 and 3). It should be noted that there was no bacterial competition during this test as aeration was maintained throughout. Figs 4.20 and 4.21 show the corresponding tabulated nitrite and nitrate concentrations. These clearly show good nitrification as there was no significant build-up of the intermediary nitrite, but nitrate levels did increase in line with ammonia oxidation results (Figs 4.18 and 4.19). The nitrate concentrations warrant further attention as there was no noticeable decrease in their value throughout the test, as there was with the hourly additions.

Figs 4.22 and 4.23 show the corresponding BOD time profiles for the 3 hourly additions. These results clearly show a considerable reduction in the BOD concentration from an initial value of around 275-300mg L^{-1} to between 100-150mg L^{-1} at the end of each 3h period. This gives a corresponding reduction rate of approximately 50mg L^{-1} h^{-1}, which is comparable to the rate achieved in the previous wind-powered tests (chapter 3). Throughout this test there was no noticeable breakthrough of the high strength influent, as there was for the hourly additions, and the system gave comparable performance results to the previous work (chapters 2 and 3). At the end of the tests the system was decommissioned. While being cleaned, it was noticed that there was significant solids accumulation within the bed matrix, but unfortunately this was not measured and quantified.

4.4. DISCUSSION.

When the overall performance of the packed bed is looked at, not only for SS but also ammonia and BOD reduction it can be seen that there were no adverse effects of this exceptionally high SS loading. In fact, the packed bed as expected, conferred an increase in operational stability under these conditions. It was only when it became clogged with solids that the plant suffered problems, with solids removal increasing over previous work (chapter 3). The solids accumulation concerns do however need to be quantified in a little more detail. There is particular concern that solids clog the system should be known more accurately.

It is clear from the hourly additions that there should not be any significant clogging of the Lytag support medium until around 2-3 years of operation. This was calculated from the amount of solids added to the system after 30 days (12.8kg) which was the point where solids breakthrough became significant (Fig 4.7). If this SS concentration is divided by an equivalent weak effluent solids loading (Fig 4.1) of 350mg L^{-1}, it gives a treated volume before solids accumulation becomes a problem for this system of around 17,000 L. This can then be converted to a plant operation time by dividing by the system volume (24L) and throughput time. These SS results also indicated that after this operational period, the build-up of solids in the bed would become a problem, leading to the system requiring some type of cleaning or back washing. This figure is heavily dependent on throughput times (for the hourly additions this was around 6h), but in

commercial treatment this figure is likely to be more in the range 2-4h, which would obviously reduce the operation time before solids problems by a factor of 2. As solids accumulation is an issue, then the plant will require cleaning. This could be done by back washing, which, due to the weight and size of the Lytag, will not be particularly difficult, as it could be agitated and suspended without too much difficulty.

These figures are, of course, only a rough guideline and more extensive studies would be needed to give more accurate time scales, as the system will obviously be sensitive to many variables such as differing effluent strengths (SS, BOD and ammonia) and hydraulic loadings. Also in these studies, due to the lack of any measurement of the final Lytag solids content, the amount of solid removed by bacterial action cannot be assessed, however, it is expected that there would have been some. This lack of final solids data also hinders interpreting the results for the solids reduction in the aerobic zone, although from the three hourly addition work it was clear that the solids in the liquor did reduce significantly.

The other area of interest is the bulk liquor dissolved oxygen levels, together with analogous figures for the packed bed. Again, unfortunately there were no on line dissolved oxygen measurements taken, in part due to the high solids loading, which affect readings when on-line equipment was used. However, nitrification is a suitable measure of dissolved oxygen level due to its sensitivity to low dissolved oxygen concentrations (Beccari *et al.*, 1992). Using this, it is clear that from both the 3h and hourly addition work there was no oxygen limitation in the bulk liquid until the suspended solids

breakthrough, as expected. Again unfortunately due to the lack of data obtained from the bed itself there can be no conclusions drawn to its dissolved oxygen level but it is expected from the literature to be very low (Stenstrom and Song, 1991; Metcalf and Eddy, 1992; WPCF, 1988). This is also supported by the presence of denitrification, which is noticeable in the 3h addition work in particular (Fig 4.21), which is extensively an anoxic process (Goronszy, 1987).

Finally it is clear that the Lytag did have a beneficial effect on the performance of the plant, although it did not meet both of its objectives. It did retain solids, preventing bacterial competition, but the vast majority of its surface was not available to support the aerobic population. To counter this it has become obvious that some mobile substrate will be required (similar to the Captor process), and along the same lines as used in some processes industries. A suitable substrate, meeting all of the previous criteria, such as cost and durability etc, will be a focus of later work.

4.5 CONCLUSIONS

The packed bed support matrix did accumulate solids under this extremely harsh test, but not until a considerable amount (over 12kg) had been passed through, and if the process is to be used cleaning/ desludging will be required on a 2-3 year time scale. This does mean that some type of sludge removal from the system will be required but this would

ideally be conceived and introduced in a way consistent with the simplicity and symmetries now existing. With recent trends towards the use of mobile substrates (or suspended substrates) it seems clear that this should be a next step in the evaluation of the current water treatment plant. The overall performance increases for both BOD reduction, ammonia oxidation and suspended solids removal provided by the Lytag packed bed substrate have proved its suitability for incorporation for this treatment process.

	Minimum	Maximum	Average
Suspended Solids ($mg\ l^{-1}$)	35	130	80
Ammonia ($mg\ l^{-1}$)	6	25	13
BOD ($mg\ l^{-1}$)	50	170	80

Fig 4.1 Typical effluent concentrations for settled sewage (Robinson *et al.*, 1994)

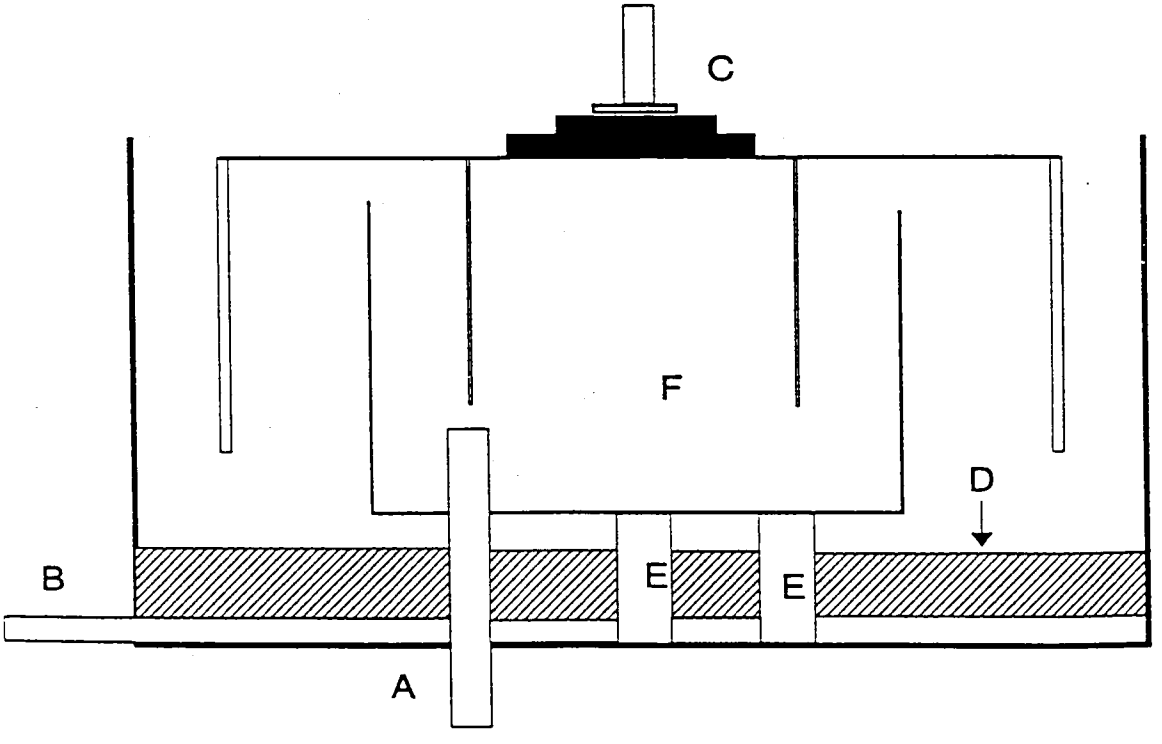


Fig 4.2. Half metre pilot plant unit, new configuration. A= treated effluent outlet pipe; B= effluent inlet and distribution plate; C=Bearing; D=lytag layer; E=Support legs and sludge return point; F=inner tank.

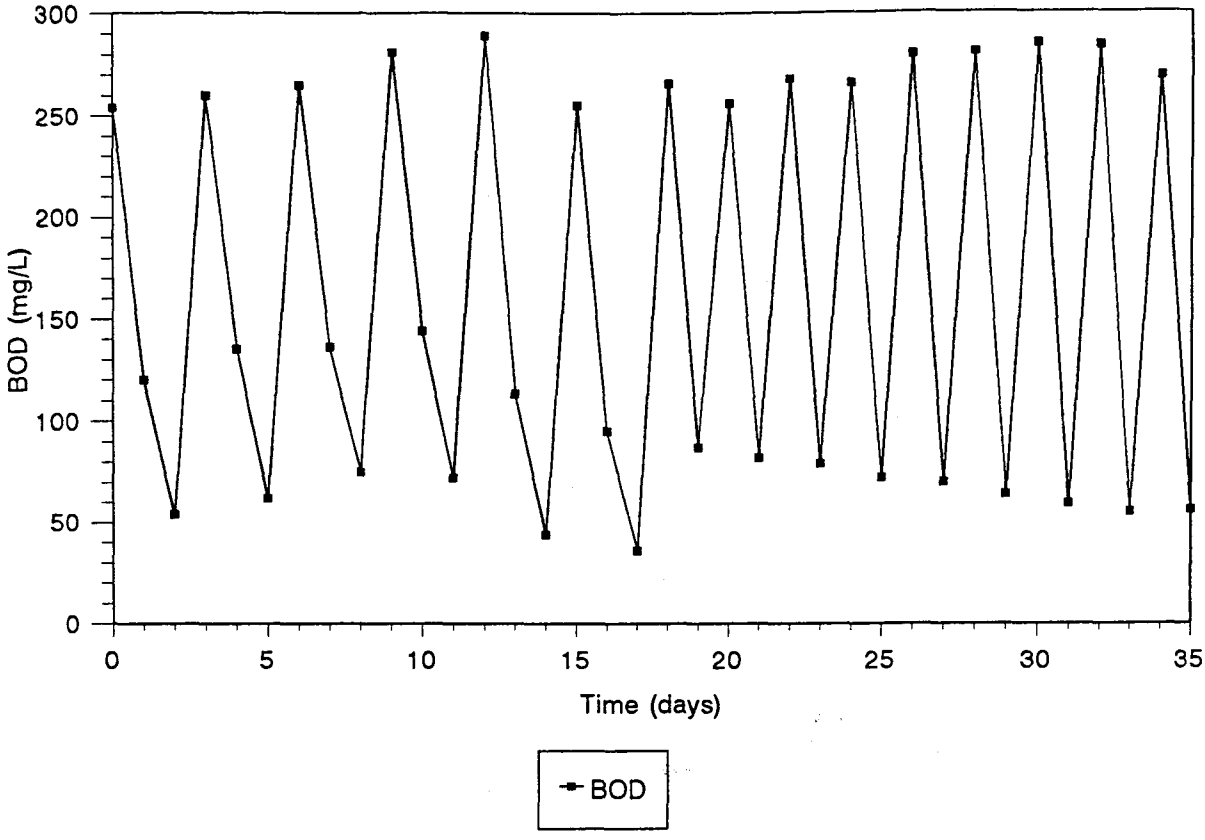


Fig 4.3 Stabilisation period for half-metre pilot plant unit, under maintained aeration. New effluent additions were every 2 to 3 days. BOD time profile.

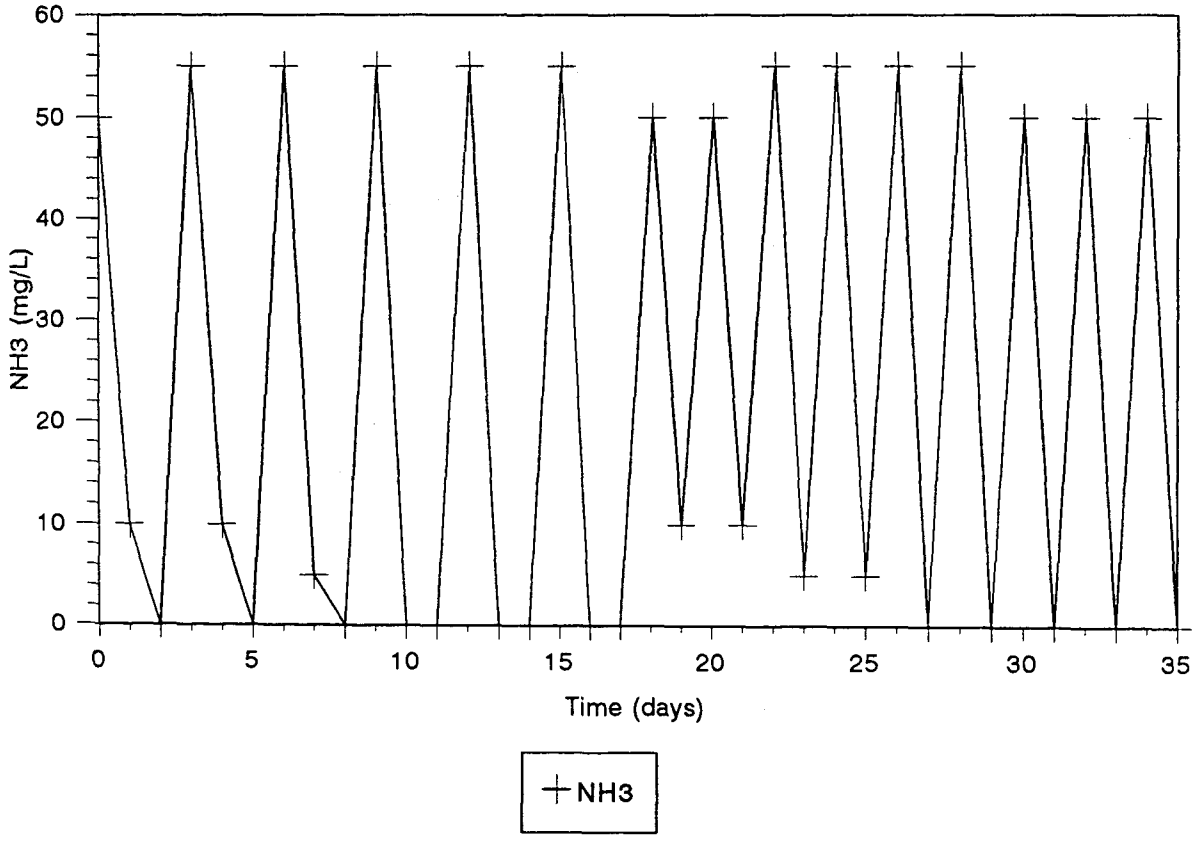


Fig 4.4 Stabilisation period for half-metre pilot plant unit, under maintained aeration. New effluent additions were every 2 to 3 days. ammonia time profile

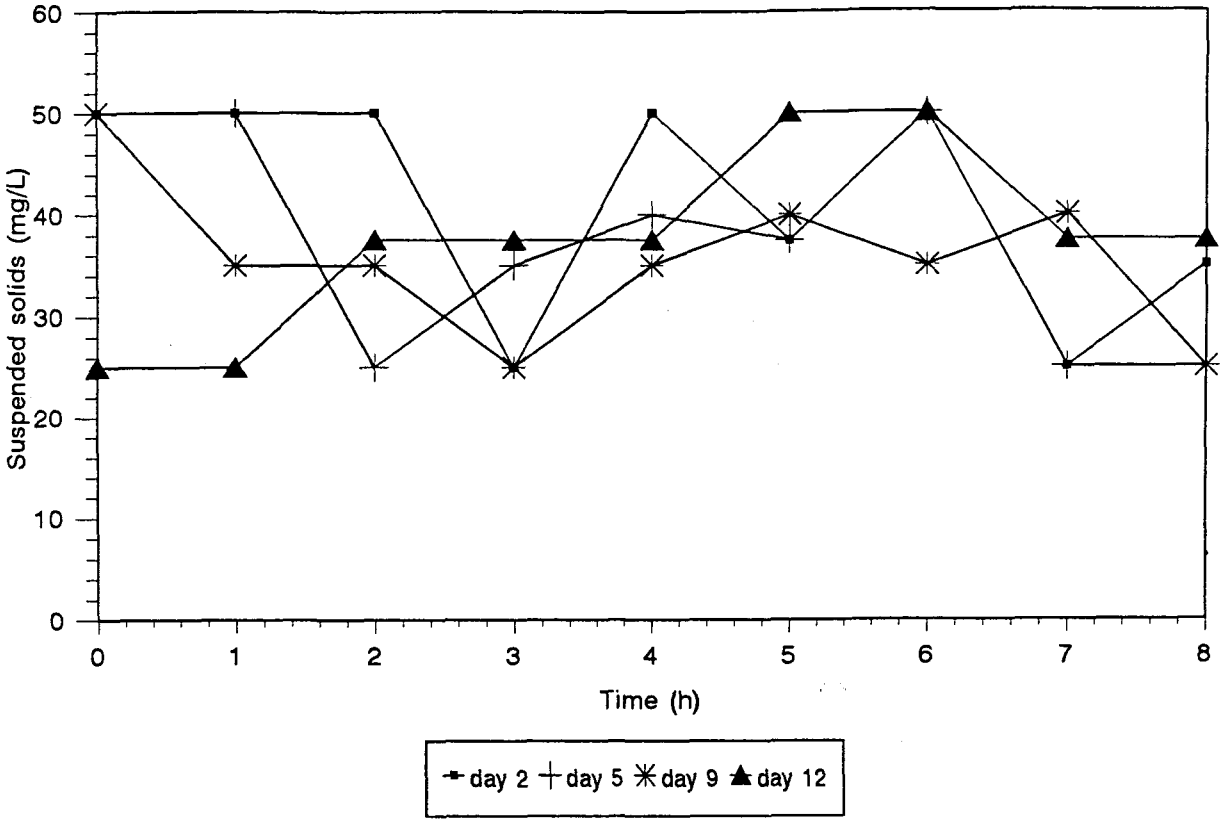


Figure 4.5. Suspended solids time profiles for days 2, 5, 9 and 12 (weeks 1 and 2) New effluent was added hourly (20,000mg/L). Aeration was maintained throughout

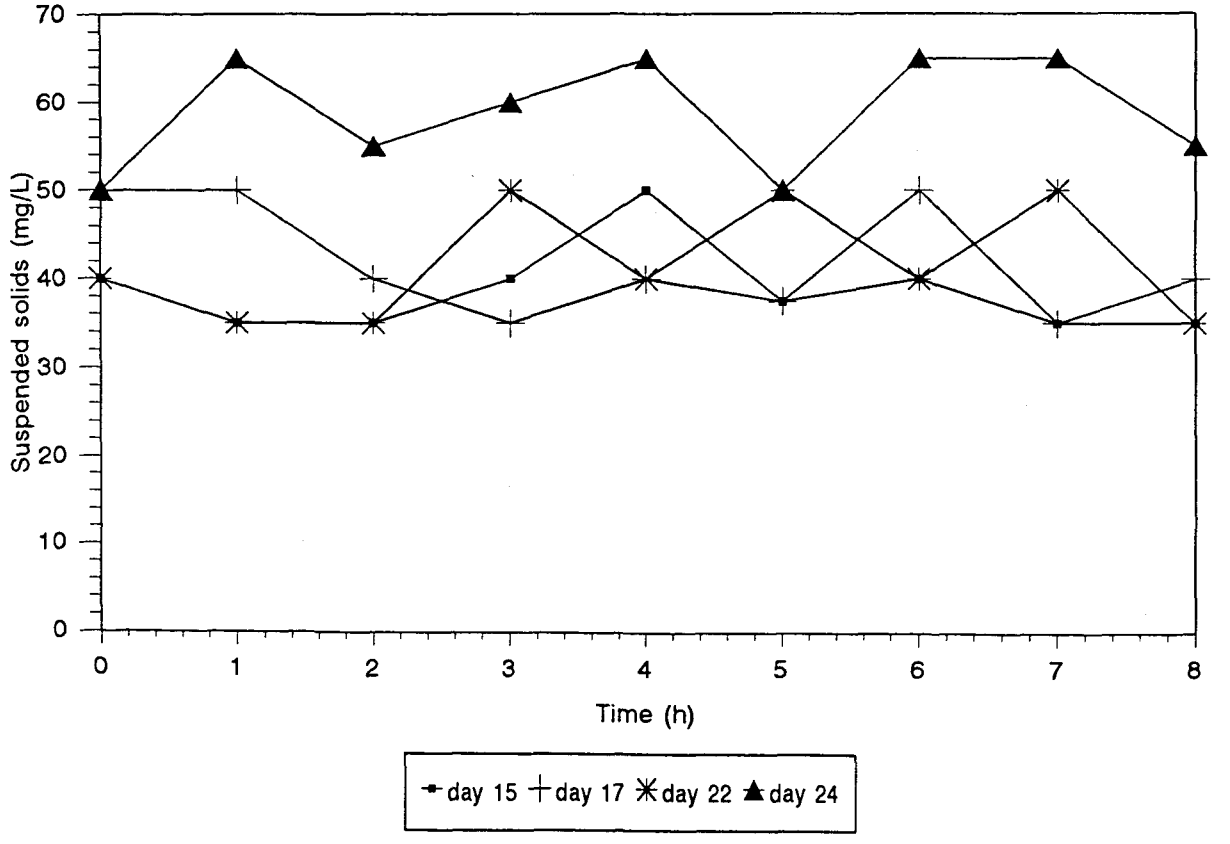


Figure 4.6. Suspended solids time profiles for days 15, 17, 22 and 24 (weeks 1 and 2) New effluent was added hourly (20,000mg/L). Aeration was maintained throughout

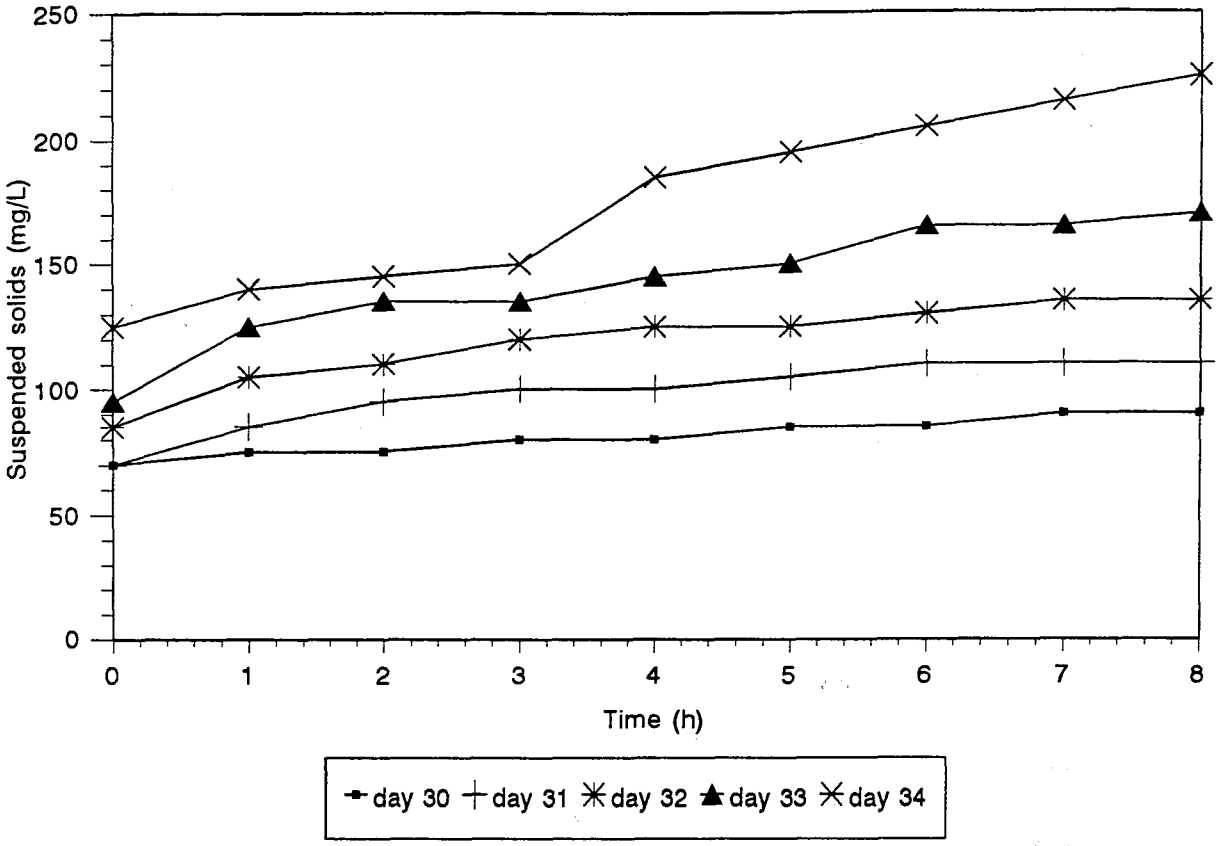


Figure 4.7. Suspended solids time profiles for the final week days 30, 31, 32, 33 and 34. Conditions as for Fig 4.5

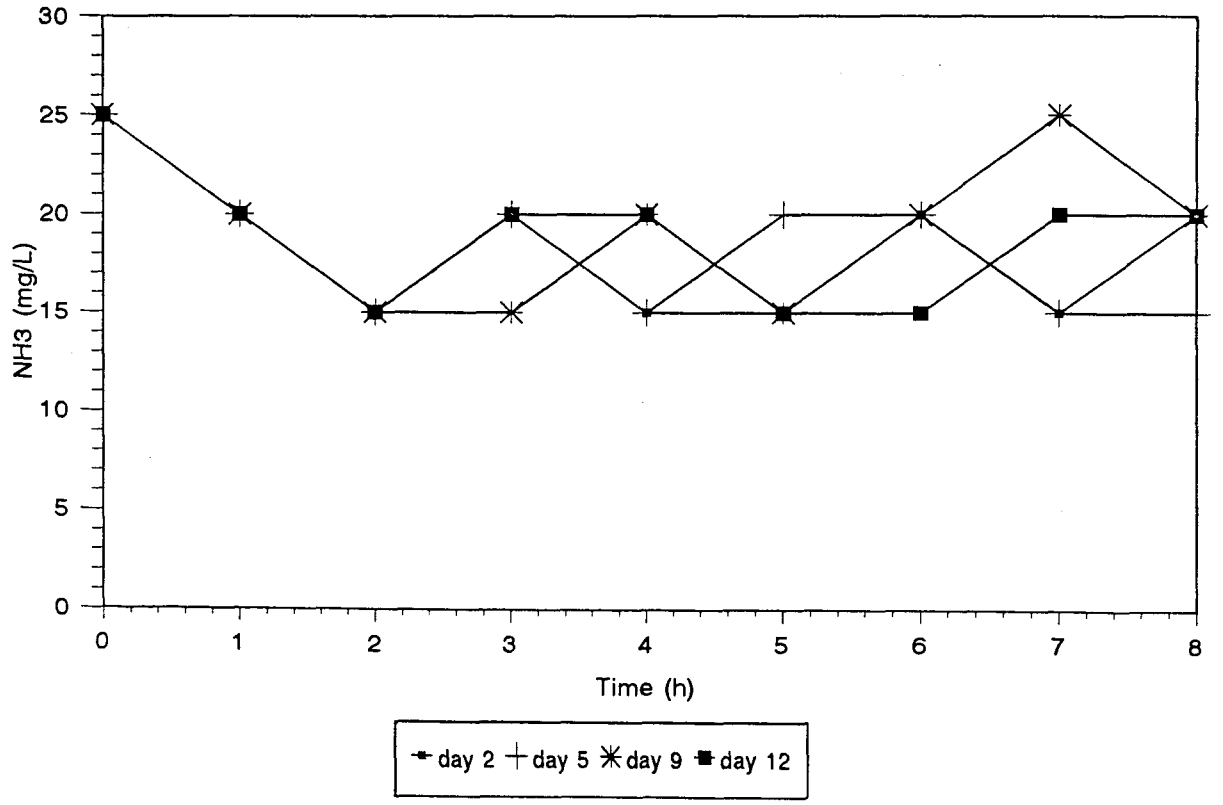


Fig 4.8. ammonia time profiles for solids loading experiment. conditions as for Fig 4.5 Hourly effluent additions.

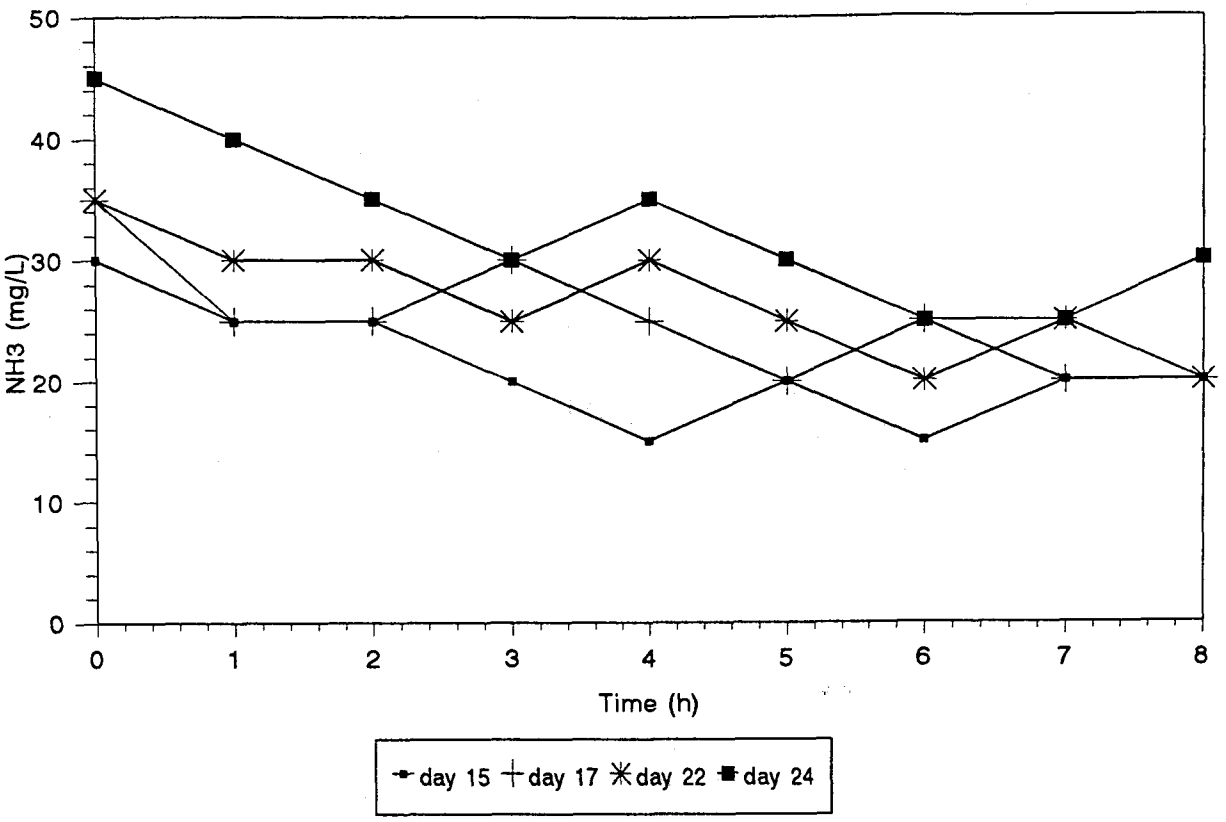


Fig 4.9. ammonia time profiles for solids loading experiment. conditions as for Fig 4.5 Hourly effluent additions.

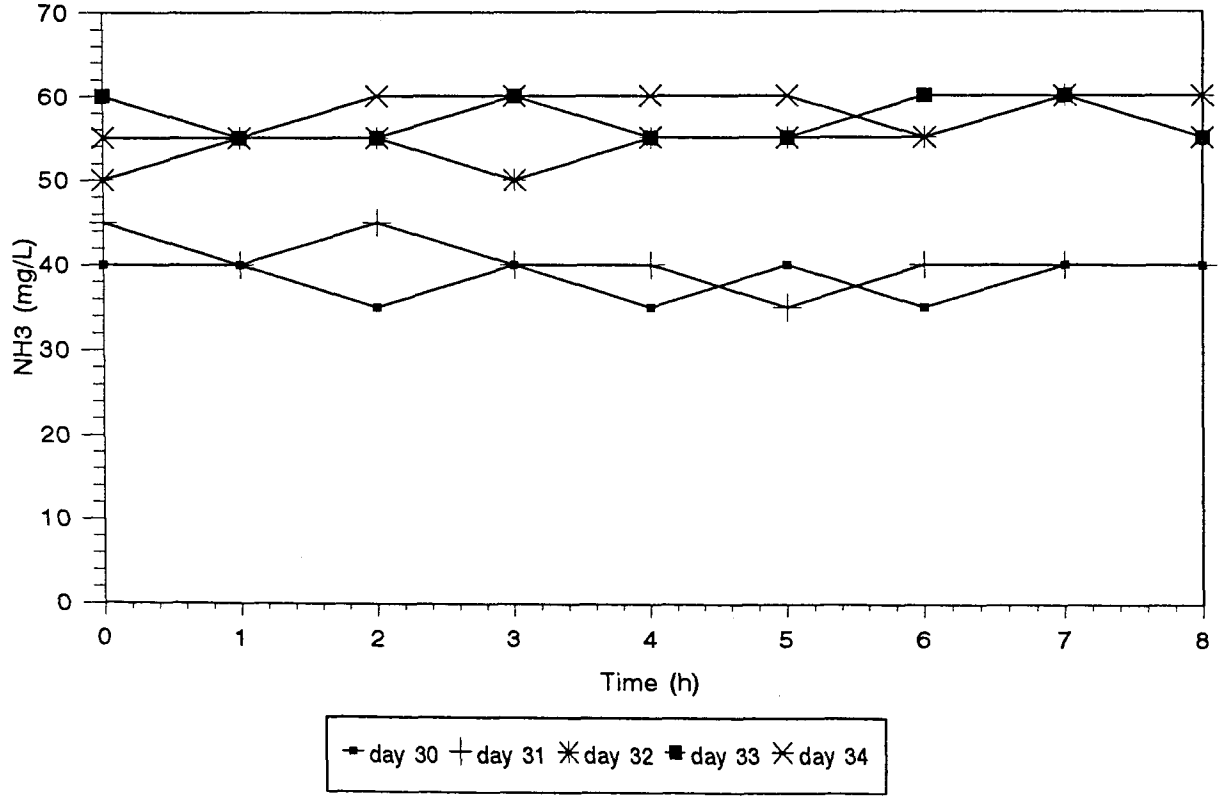


Fig 4.10. ammonia time profiles for solids loading experiment. conditions as for Fig 4.5 Hourly effluent additions.

days time	2	3	7	8	12	13	19	20	29	30	32	33	34
1	0	0	0	1	5	1	0	0	0	0	0	0	0
2	3	5	1	5	3	3	0	1	0	0	0	1	0
3	5	1	3	5	5	5	5	0	1	0	1	0	0
4	5	3	1	1	1	5	0	3	0	1	0	0	0
5	5	3	1	5	3	3	1	0	3	0	1	0	0
6	5	3	0	3	1	1	0	5	0	1	0	0	0
7	5	3	1	1	5	5	5	0	1	0	0	0	0
8	3	3	1	0	1	1	1	1	0	1	0	0	0

Fig 4.11 Nitrite concentrations following hourly effluent additions, conditions were the same as those in Fig 4.5. Aeration was maintained throughout.

Days time	2	3	7	8	12	13	19	20	29	30	32	33	34
1	0	0	5	10	10	5	15	20	20	20	25	25	25
2	10	10	10	15	10	15	20	15	25	25	25	25	25
3	5	15	20	25	25	20	25	25	25	25	25	25	25
4	10	20	25	25	25	15	25	20	25	25	25	25	25
5	10	20	20	25	20	25	20	25	25	25	25	25	25
6	15	15	25	20	25	25	25	25	25	25	25	25	25
7	20	20	25	25	25	20	20	25	25	25	25	20	25
8	20	20	25	20	10	25	20	25	25	25	25	25	25

Fig 4.12 Nitrate concentrations following hourly effluent additions, conditions were the same as those in Fig 4.5. Aeration was maintained throughout.

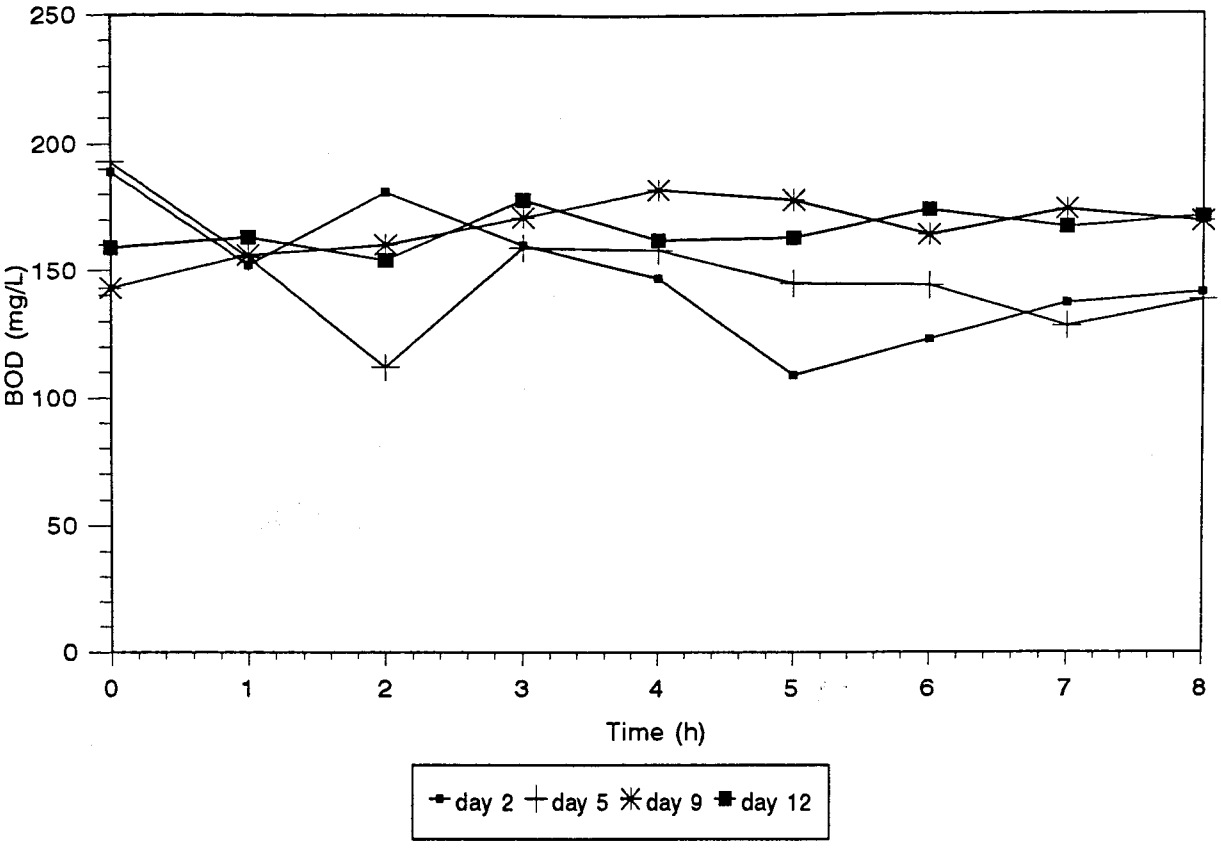


Figure 4.13 BOD time profiles for hourly additions. conditions as for Fig 4.5

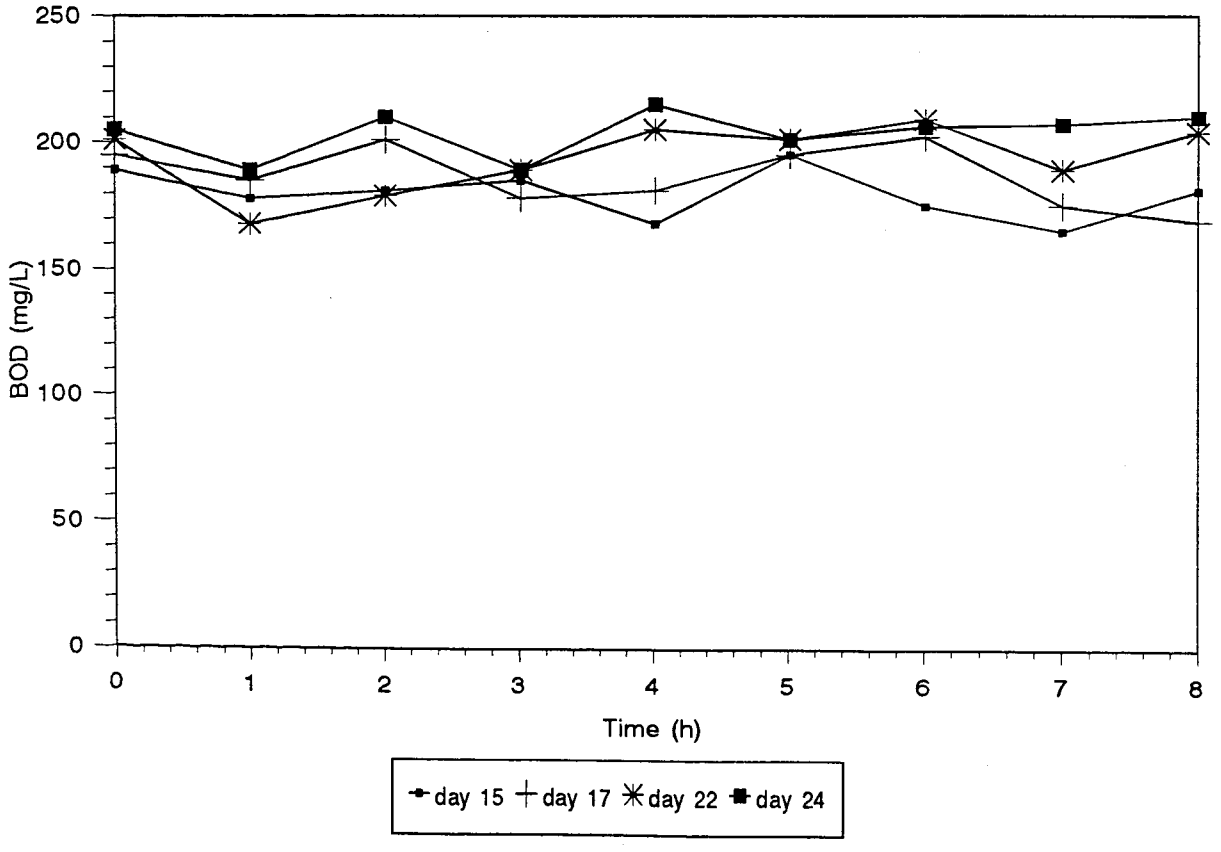


Figure 4.14 BOD time profiles for hourly additions. conditions as for Fig 4.5

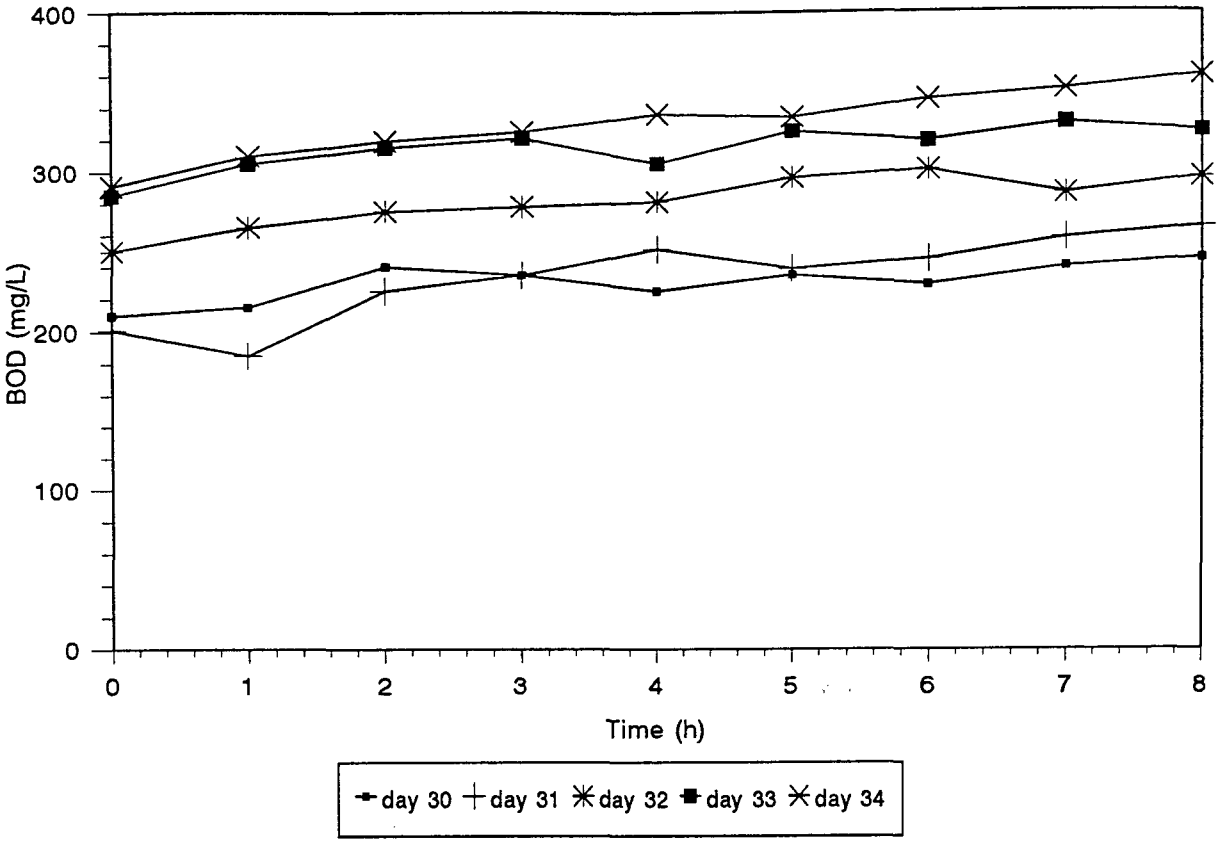


Figure 4.15 BOD time profiles for hourly additions. conditions as for Fig 4.5

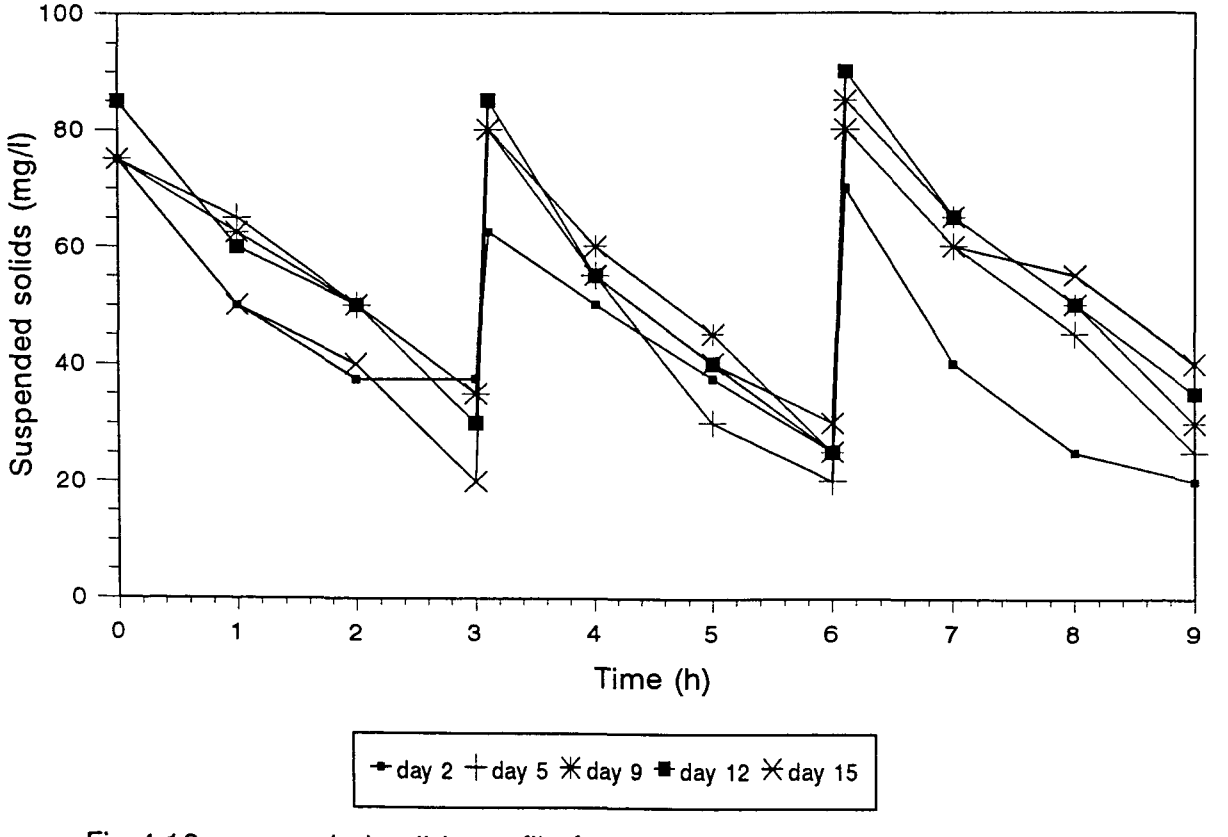


Fig 4.16. suspended solids profile for 3 hourly additions with maintatined aeration. 5L of effluent added at 0, 3 and 6 hours, concebtration 20,000mg/L

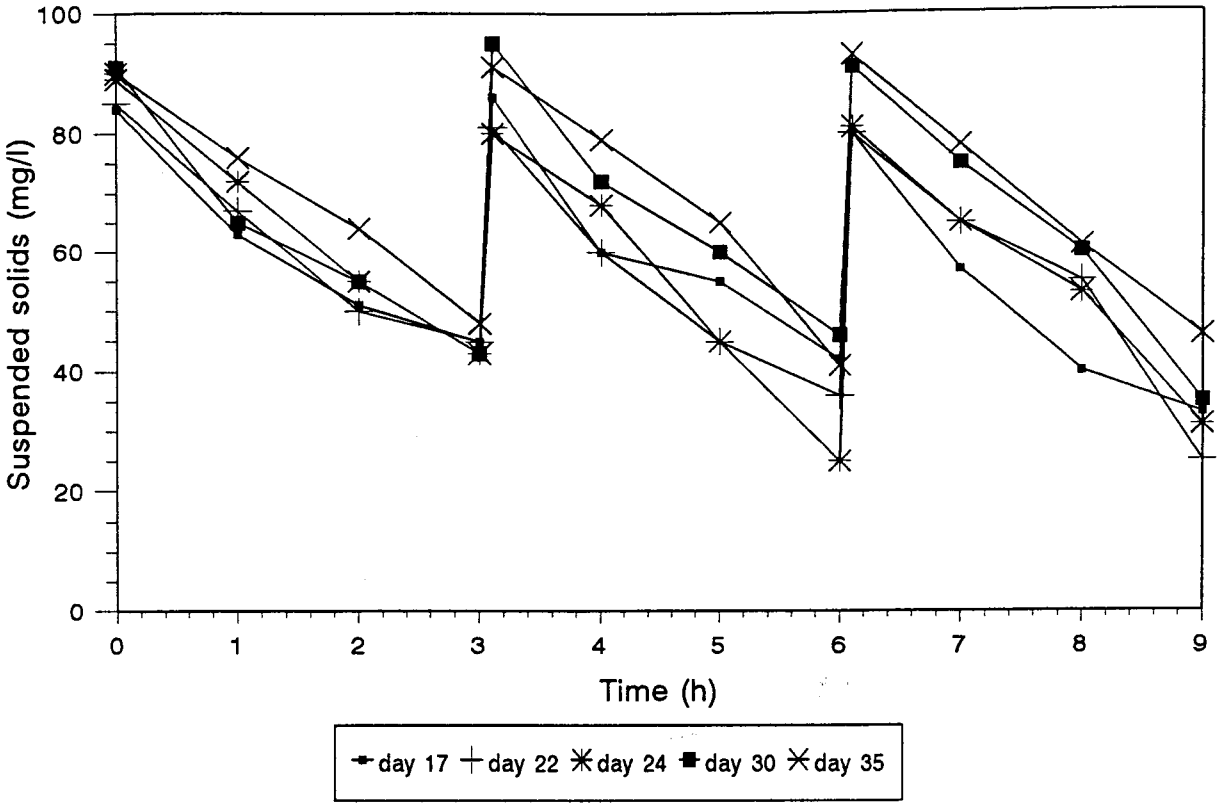


Fig 4.17. suspended solids profile for 3 hourly additions with maintatined aeration. Conditions as for Fig 4.16

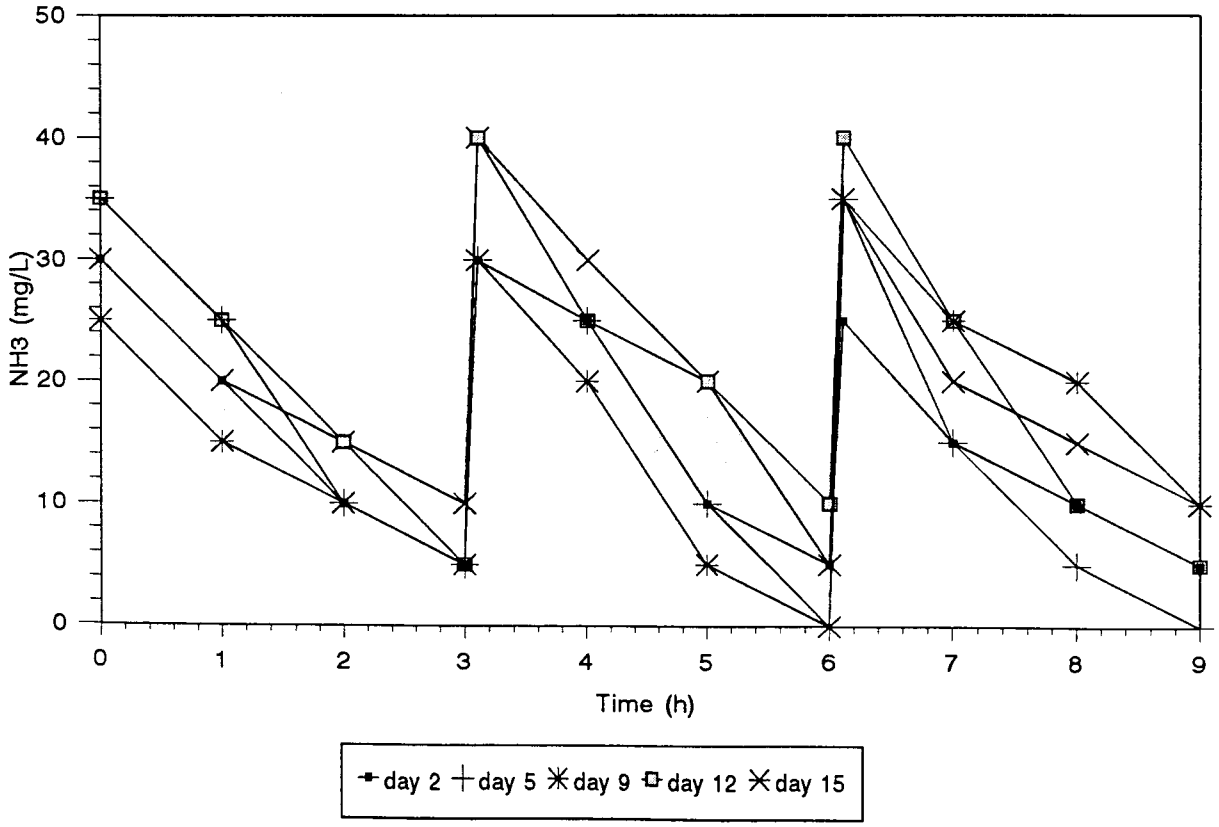


Figure 4.18 ammonia time profile for extended aeration experiment conditions as for Fig 4.16

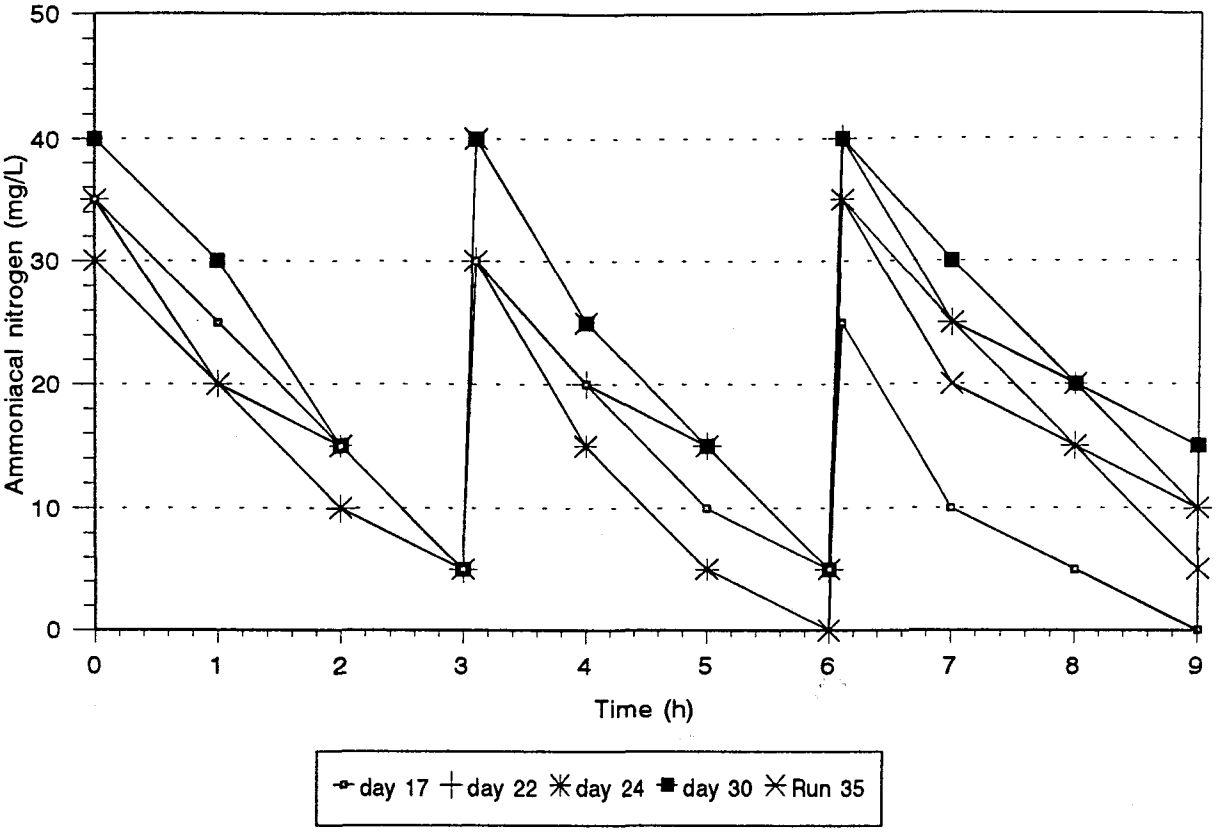


Figure 4.19 ammonia time profile for extended aeration experiment conditions as for Fig 4.16

days time	2	5	9	12	15	17	22	24	30	35
1	0	0	0	1	5	1	1	5	1	5
2	1	5	1	5	3	3	0	1	0	0
3	5	5	3	5	5	5	5	0	1	1
4	3	3	1	1	1	5	1	3	1	1
5	5	1	1	5	3	3	1	0	3	1
6	3	3	0	3	1	1	0	5	0	3
7	5	3	1	1	5	5	5	0	1	0
8	1	5	3	3	3	5	1	1	3	3
9	3	3	1	0	1	1	1	1	0	1

Fig 4.20 Nitrite concentrations following 3 hourly effluent additions, conditions were the same as those in Fig 4.16. Aeration was maintained throughout. Time is in hours

Days time	2	5	9	12	15	17	22	24	30	35
1	0	0	5	10	10	5	5	0	0	5
2	10	10	10	15	10	15	20	15	10	25
3	5	15	20	25	25	20	25	25	10	15
4	15	10	25	25	15	15	15	10	15	20
5	10	20	20	15	20	25	10	25	25	15
6	15	15	25	10	15	25	25	15	15	25
7	25	20	25	25	25	20	20	25	10	15
8	20	10	15	15	25	10	20	20	25	25
9	20	20	25	20	10	25	20	25	25	25

Fig 4.21 Nitrate concentrations following 3 hourly effluent additions, conditions were the same as those in Fig 4.16. Aeration was maintained throughout. Time is in hours.

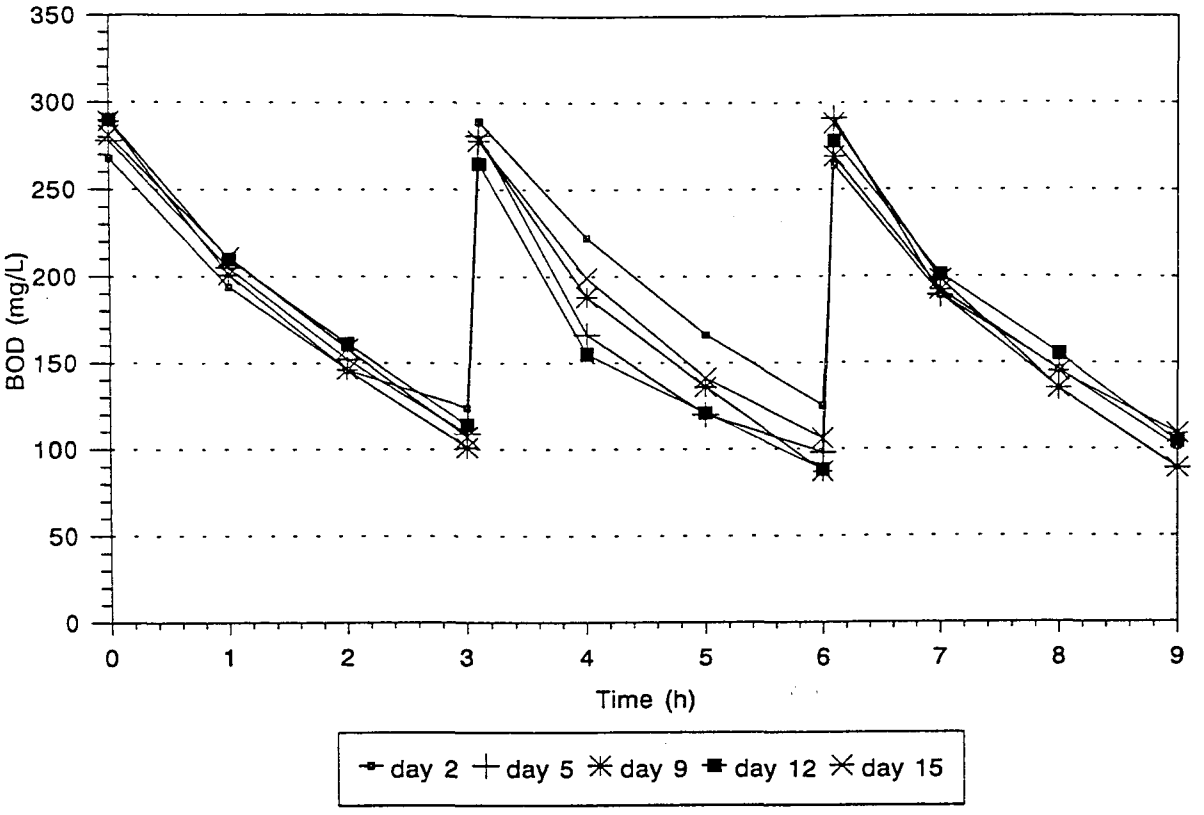


Figure 4.22 BOD time profile for extended aeration tests conditions as for Fig 4.14.

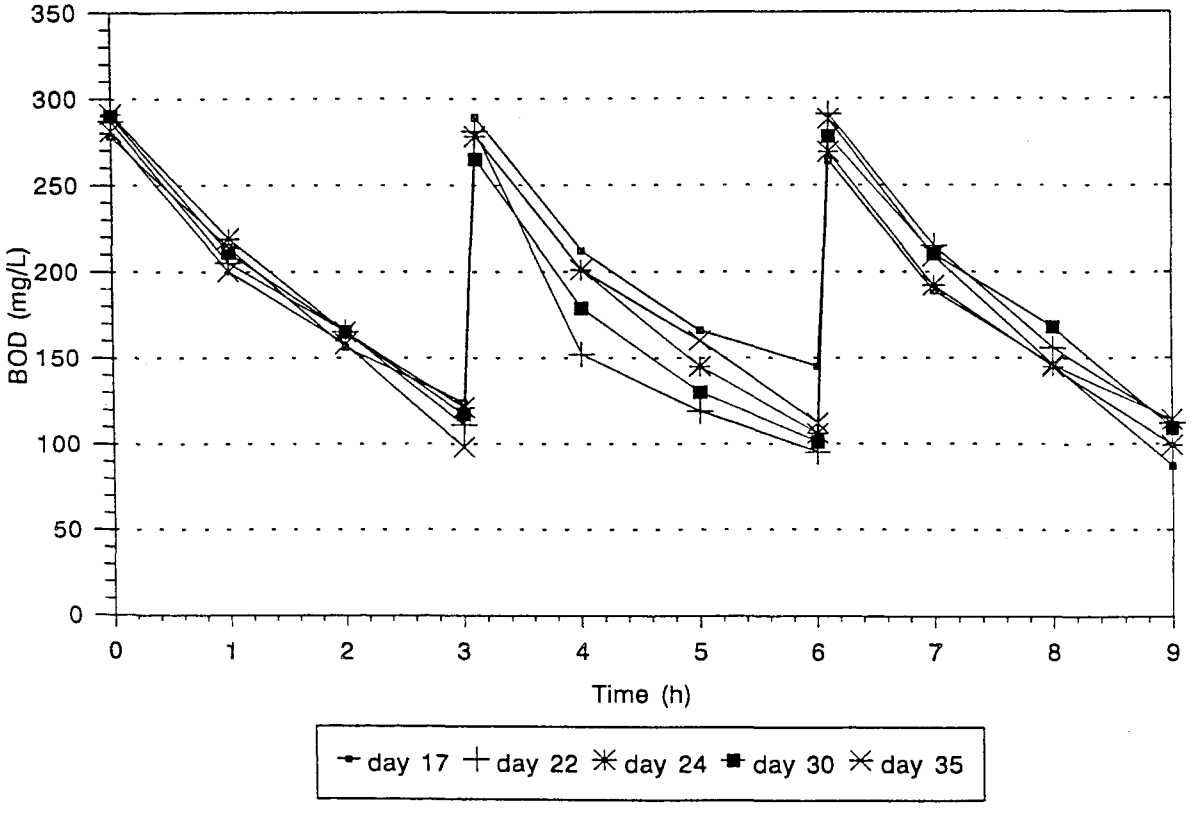


Figure 4.23 BOD time profile for estended aeration tests conditions as for Fig 4.14.

CHAPTER 5 ANAEROBIC PRE AND POST TREATMENT FOR INCREASED BACTERIAL STABILITY AND DENITRIFICATION

SUMMARY.

It has been widely recognised in the water treatment sector that packed bed substrates confer advantages for oxidative treatment by stabilising the biology as an attached bio-film. For aerobic conditions it is both the mobility and increased surface area of a substrate that confers advantages, in particular for mass transfer of nutrients and dissolved oxygen to the bacterial population. The advantages of bio-films as used in trickling filters, can extend not only to aerobic and anoxic conditions but also to anaerobic processes. Anaerobic processes are often held to be more robust in withstanding toxic and shock loadings and many integrated treatment processes employ anaerobic pre-treatment as well as anoxic/anaerobic denitrification as final stage. Taking these various considerations together and recognising that carbon oxidation of a certain level is required for nitrification; the use of a four-stage system seemed appropriate. This system would incorporate anaerobic pre-treatment, two aerobic oxidation zones one for BOD reduction, the other for ammonia oxidation. Finally an anoxic/anaerobic denitrification zone could be used as a polishing step.

The study reported here evaluates a segregated sequential four-stage system incorporating both packed and mobile substrate zones. As packing we retained Lytag, a micropourous heavy cut slag, to provide an adequate substrate for biomass support for the anoxic/anaerobic zones. The mobile substrate was sourced from the few candidates

possessing the commercial advantage of cheapness, and was based around the reticulated foam used in the "Captor process". Following extensive and largely inconsequential evaluation of candidates, a nylon mesh matrix commonly used as scouring pads was identified as a solution. The pads were utilised as $1cm^2$ sections cut in to 2-3mm squares, with the preliminary work using shake flasks verifying their suitability for bacterial colonisation. Whilst satisfactory performance was delivered on carbon and ammonia oxidation (in line with previous work), significant denitrification cannot be claimed from the studies undertaken so far.

5.1. INTRODUCTION

Whilst anaerobic effluent treatment units have been widely evaluated, their utilisation as a pre-treatment is considerably scarcer. One of the few examples is that of Goronszy *et al.*, (1987) who utilised a calcium carbonate reactive growth support media for the anaerobic pre-treatment of potato processing waste water. In this process the calcium carbonate support matrix allowed a 70% reduction in COD, at a loading of 3kg COD m^{-3} d^{-1} with a retention time of around 20h (initial starting concentration of 3000-4000mg L^{-1}). The amount of calcium carbonate used was some 1.4g for every g COD reduced, with the alkalinity this provided minimising the impact of short term pH fluctuations (Goronszy *et al.*, 1987). Following this anaerobic treatment the wastewater was passed to a conventional activated sludge plant.

There are of course problems associated with anaerobic treatment of municipal effluent, compared to aerobic treatment. First the organic strength of sewage is relatively weak (typically 250-400mg L^{-1}), which will limit cell growth, since the uptake of carbon substrate is the rate limiting step (Jewell, 1985). Also, there is the question of temperature, as most sewage entering a treatment works is fairly constant between 10-20°C (Stanibridge, 1977) depending on climate, and this does not favour anaerobic treatment, which usually demands an operating temperature at least 10°C higher (Jewel, 1987). There has been some work on low temperature anaerobic treatment although this is still a relatively new process (Sanz *et al.*, 1990). Next there is the question of throughput times, as most conventional aerobic treatments have throughput times in the 4-8h range (Metcalf and Eddy, 1992), with pressure on this to be reduced further to around 2h. Unfortunately anaerobic processes need much longer residence times of up to 20h (Fang *et al.*, 1995; Johnson *et al.*,1994) as the microbial populations involved have slow growth rates and longer metabolic and respiration times (Krogulska *et al.*, 1984). Apart from these problems, there is also the possibility of aerobic bacterial inhibition due to oxygen-deficient liquor passing into the aerobic zone (Laanbroek *et al.*, 1994).

Among the processes that have been tried (Heijnen *et al.*, 1991; Goronszy *et al.*, 1987) there have been no obvious problems of aerobic inhibition. However, Heijnen *et al* (1984) also have claimed that for the anaerobic zone to be of practical value in such a configuration, the population would need to be supported, allowing a high biomass concentration, increased sludge age and corresponding increase in throughput times . On the other hand, there are advantages to be gained. Inclusion of an anaerobic pre-

treatment reduced energy consumption (reduced aeration for complete treatment) reduced sludge production and increased settling speed of the flocs produced (Heijnen *et al.*, 1991).

As with aerobic packed bed systems (chapter 4), one of the main concerns with utilising anaerobic packed bed pre-treatment is the blocking of the matrix, and subsequent cleaning requirement. This was more fully discussed in chapter 4, with the conclusion that blocking would not become a problem, until after several years of continuous operation; however, there would be some reduction in plant performance over this period (Miyahara *et al.*, 1994). Further concerns centre on the two main bacterial species involved and there differing environmental requirements. The two species involved in anaerobic degradation are *acidogens* and *methanogens* (Metcalf and Eddy, 1992). The *acidogens*, as their name suggests, produce acidic conditions, and are more oxygen intolerant than the *methanogens* (McGhee, 1991). These differing environments mean that if the system were well mixed, there would be a need for two tanks, one for each species to proliferate in. Alternatively a period of stratification would be required if using one tank. However, under conditions where there is natural stratification, either from supply of nutrients or dissolved oxygen, one tank would suffice. Essentially taking this natural stratification as the inspiration and recognising the relatively few attempts to capture such symbiotic processes in intensive treatment plant, we undertook to explore the advantages in an integrated four-stage system.

Plant denitrification performance is coming more to the fore as nitrate discharge limits are decreased (Roberts, 1995), to improve water quality. Denitrification can occur either by assimilation (reduced to ammonia, incorporated into cells) or dissimulation (reduction to gaseous nitrogen) but is essentially an anoxic process. The incorporation of a denitrification zone into the system raises several concerns, some of which are similar to those of the anaerobic pre-treatment, such as substrate blockage or methane production. There is, however, the control of carbon levels entering the denitrification zone, which will affect the denitrification rate (Nurse *et al.*, 1980).

Currently for effluent denitrification there are several methods available. These can be segregated according to their carbon source (either methanol or endogenous carbon), and how the bacteria are supported (attached or suspended). Of the two substrate types, methanol used to be by far the most commonly used (Nurse, 1980), with its main drawback being cost. However, it does provide rapid denitrification (25% to 10%) when compared with that obtained using an endogenous carbon source. The use of support matrices is also used to increase sludge age with the most common forms being the Sludge Blanket (Klapwijk *et al.*, 1981), and fluidised bed reactors. Both of these processes use sand (Sanz *et al.*, 1990) as their support matrix although there are biofilm reactors utilising polypropylene fibres (Takamizawa *et al.*, 1993). Typical performance data for these systems are nitrate removals of up to 85-90% at hydraulic loadings of approximately 20 L h^{-1}. However, due to their intensive nature, the running costs are high. Also there will be a requirement for back-washing for accumulated solids removal, to prevent sand blockage.

Our study concentrated on the batch operation of half-metre units of increasing depth. This was to investigate the concept of an anaerobic pre-treatment zone and whether oxygen penetration of the Lytag layer would be a problem. Finally, once the ideal depth of Lytag had been established the system was re-configured to incorporate a fourth zone. This was then used as a denitrification and final polishing step. The report is split into four sections with the next section outlining the materials and methods used. In section three we will present the results, with a focused comparative discussion, comparing and contrasting these results with earlier work. Finally the last section will give a broader account of how this work links to previous findings and also to the overall goal of implementing this multi-stage system.

5.2. MATERIALS AND METHODS.

The work presented here is split into two sections. First, a single tank, two-stage system was utilised to test if an anaerobic/anoxic pre-treatment was a viable option. Second, this two stage tank was converted to a four-stage system by the use of shelves, to test if this integrated system would provide the performance required. To utilise a four stage system an aerobic mobile substrate was needed. The mobile matrices were placed in the agitated upper layer, itself segregated according to findings established earlier, to provide protection for the nitrifiers from undue competition for oxygen by carbonaceous organisms associated with BOD levels in excess of 100mg L^{-1} (recall earlier findings,

chapters 2and 3). The nitrification zone was then split into two zones with a view to look at the scope for denitrification within the lower layer of packed Lytag.

5.2.1. Single tank two-stage system.

In this system a tank of height 1.5m and diameter 550mm, had a distribution plate placed in the bottom of it (Fig 5.1), on top of which a layer of Lytag (washed and sized maximum diameter 10mm) was placed, to a depth of 550mm. Into this tank the aeration system of two snorkel aerators (350mm separation) was placed, on a central shaft. The new aerators were 350mm long (base to air inlet) with a triangular cross section, with a front angle of 30°, and side lengths of 10mm; the back surface was made of perforated (50 cm^2) brass. The aerators were submerged to 320mm, and were rotated at 120rpm (2m s^{-1}) using a 2kW motor. This produced K_La values of 0.65min^{-1}, and overall power to oxygen values of around 1kg O_2 kWh^{-1}.

The screened sewage was added in two stages. First some 70L of primary screened sewage (obtained from Severn Trent Water Reclamation works, Minworth; BOD 400-350mg L^{-1}, ammonia 50-55mg L^{-1}, negligible nitrite and nitrate) was poured into the tank, then approximately 3L of anaerobic granular sludge (from the Civil Engineering department at Birmingham University), was added as an anaerobic bio-catalyst. Next a further 70L of primary screened sewage was introduced over a 24h period (at a uniform rate) of around 3L h^{-1} bringing the total volume to approximately 143L, which filled the tank to a depth of around 1.2m. The top aerobic layer was seeded with 3L of activated sludge poured directly into the top of the tank. Aeration was maintained (120rpm), for 4

weeks to allow bacterial colonisation of the Lytag, and to replace evaporation, further daily additions of sewage (1-2L) were used. To increase the temperature of the anaerobic layer the bottom of the tank was insulated with 80mm thick fibre-glass loft insulation, which allowed the temperature in the Lytag to be maintained at around 27-30°C for the majority of the experiment.

After a four-week stabilisation period the speed of the aerators was varied, to assess how the depth of the boundary layer between aerobic and anoxic layers might alter with aerator speed. The three speeds used were 60rpm (1m s^{-1}), 90rpm (1.5m s^{-1}) and 120rpm (2m s^{-1}). For each speed setting the plant was allowed to operate for one day before sampling. Then 30L of new effluent (concentrations as above) were added and 50ml samples taken from four depths from and including the surface. These were 0mm, 400mm, 800mm and 1200mm, with these samples being tested for ammonia (Quantofix test strips see previous chapters), BOD (Winkler method see previous chapters), oxygen and nitrate (Quantofix test strips see previous chapters). The sample from 1200mm was taken from inside the Lytag layer, by the use of a fixed sample point, the purpose being to assess the conditions into the Lytag. Oxygen measurement was by four fixed calibrated galvanic oxygen probes (using the static gassing out method, (Ahmad *et al.*, 1988)), which was recalibrated every 2-3 days to eliminate drift problems. The probes were fixed at the previous sample heights, by the use of clips, which facilitated their easy removal for cleaning, and re-calibration.

5.2.2 Co-Axial Twin tank arrangement.

An inner tank of diameter 300mm (Fig 5.2) was introduced co-axially with the tank described above (diameter 550mm). This would thereby reproduce the configurational layout studied previously for two stage oxidation system. The inner tank (Fig 5.2) was 1200mm deep compared with 1500mm for the outer tank. The outer annular zone was filled with Lytag to a depth of 450mm, above which a shelf was introduced at a height of 550mm above the tank base, and bevelled to encourage methane egress from the lower region. This avoids the potentially hazardous accumulation of combustible methane. The purpose of the shelf was to confine mobile support pads to the upper nominally aerobic agitated layer. Eight transfer tubes (25mm diameter) were located symmetrically around the shelf annulus, to aid transfer of effluent from anaerobic to aerobic zone. These were fitted with coarse mesh tops to prevent the pads from falling down them.

A two zone strategy was also adopted for the inner tank as indicated in Fig 5.3 with a Lytag layer of some 400mm being placed in the bottom of the tank. A shelf, 500mm from tank base, was then placed in the inner tank. This tank was again bevelled to prevent methane retention and contained a transfer tube similar to those mentioned above in its centre. The mobile supports used in the upper layers of each tank comprised nylon mesh pads cut to $10mm^2$ (as supplied by Key Plastics Ltd, Maidenhead) and used in a 10% volume fraction in both aerobic sections. These pads were adopted, following successful results at shake-flask scale (see appendix 1; Angel and Heath, 1995). Motor powered snorkel aerators (described above) again provided aeration, however four were used, and

two in each zone, with separation of 250mm (inner tank) and 350mm (outer tank). Rotational speed was 120rpm (2m s^{-1}), providing a similar oxygen input (K_La values of 0.35min^{-1}, and overall power to oxygen values of around 0.8kg O_2 KWh^{-1}). Liquor transfer from outer to inner tank was facilitated by 14 x 10mm diameter holes located symmetrically 30mm from the top edge.

The system was seeded as before with some 60L of primary screened sewage (Minworth) being added via the distribution plate to the outer annulus, together with 6L of granular anaerobic sludge (Civil Engineering Dept, University of Birmingham) as an anaerobic bio-catalyst. The inner Lytag layer was seeded via the exit pipe (Fig 5.3) by the addition of 6L of granular anaerobic sludge, supplemented with 30L of primary screened sewage. Both of these zones were presumed to be anaerobic, as there was no equipment available to prove whether it was or not. These additions filled the outer and inner annulus to the level of the shelf. The level was then increased after 1 week by the addition of a further 80L of primary screened sewage via the distribution plate, following which 15L of activated sludge was poured into the top section of the outer annulus as an aerobic catalyst. Into this section the nylon pads were added to a 10% annulus working volume fraction, and the aeration unit as described above was turned on (120rpm, four aerators). Further sewage additions of between 2-3L were made daily over a three week stabilisation period, when BOD levels in the top aerobic outer annulus dipped below 100mg L^{-1}.

For the four stage system, samples were taken from the inner tank just below the surface,

from the top of the outer Lytag layer, and from the exit pipe. To take a sample from just above the surface of the outer Lytag layer, a 5mm radius tube was placed down one of the transfer tubes. Suction was applied, and once flow had started the first 150ml was retained then a 100ml sample taken and the retained 150ml was then returned to the tank.

In all cases hourly 100ml samples were taken and tested immediately for BOD, suspended solids, ammonia, nitrite and nitrate. BOD was estimated by the Winkler (1982) method (described in detail in chapter 3). Ammonia, nitrate and nitrite were tested using Quantofix Test Strips (Camlab), which are sensitive down to 1mg L^{-1} (nitrite) and 5mg L^{-1} (ammonia and nitrate). The sensitivity of these strips was increased as outlined in previous chapters. Suspended solids were measured by filtering a known volume of liquid (25ml) through a pre-weighed filter paper, which was re-weighed after being placed overnight in an oven at 105°C, to remove moisture.

5.3. RESULTS OF SINGLE TANK TWO-STAGE SYSTEM

5.3.1 Aerator speed 60rpm (1mm s^{-1})

Fig 5.4 shows the dissolved oxygen levels against time at the four sample depths, for the two-stage system with the aerators rotating at 60rpm (1m s^{-1}, K_La 0.07min^{-1}). The graph clearly shows that, as expected, the surface layer was well oxygenated (30-35% saturation); however this aerated layer did not extend to the next sample point of 400mm, even though the aerators were submerged to 320mm. Below 400mm the dissolved oxygen level was consistently below 15% saturation, and only 33% of the surface value.

The concentration at 1200mm, the bottom of the Lytag layer, was below 10% (equal to approximately 0.6mg L^{-1}; saturation constant for water taken from Perry, 1992). It is interesting that there is still so much dissolved oxygen at this depth. This dissolved oxygen presence will severely affect any obligative anaerobes present (Kemp *et al.*, 1990), which cannot tolerate dissolved oxygen. It was a pity that electrode potential readings were not taken, as they would have told us just what type of conditions we had, whether they were anoxic or anaerobic. Moreover, it should be noted that the incoming influent dissolved oxygen levels were not taken and will also have contributed to this residue.

Fig 5.5 shows the BOD time profiles for this aerator speed (60rpm), and it is clear from the graph that there is good BOD removal above 400mm in the tank, at a rate of approximately 30mg L^{-1} h^{-1}, consistent with earlier work (chapters 2-4). The traces for depths 400mm and 800mm also show BOD degradation however at a slower rate of 12-15mg L^{-1} h^{-1}, consistent with the dissolved oxygen time profiles and indicate that there is some stratification within the lower reaches of the tank at this aerator speed. The BOD concentration in the Lytag layer (1200mm) also showed a decrease although only slight at 100mg L^{-1} (a 25% reduction) over 8h; this compares with 50-80% reduction in many anaerobic treatment systems (Goronszy, 1987; Johnson *et al.*, 1994) and greater than 98% in some intensive plant (Harada *et al.*, 1994).

Fig 5.6 shows the ammonia time traces for the four depths, at the above conditions (influent ammonia concentration 50-55mg L^{-1}). This graph shows that there was little ammonia oxidation in the system except in the top 400mm. Here oxidation takes place at

a rate of around 5mg L^{-1} h^{-1}, although there was a slight increase in this rate towards the end of the test from 4mg L^{-1} h^{-1} (over first 4h) to 6mg L^{-1} h^{-1} (over the last 4h). These results are consistent with previous results (chapter 2-4) and support those from the dissolved oxygen graph (Fig 5.4), that only the top 400mm of the tank was sufficiently aerated for nitrification. Fig 5.7 shows the nitrate time profiles for this test, which combined with Fig 5.6 indicates that nitrification in the top 400mm is progressing well, and that there is limited nitrification in the remainder of the tank.

5.3.2. Aerator speed 90rpm (1.5m s^{-1})

Fig 5.8 shows the dissolved oxygen levels against time for the four depths, at an aerator rotational speed of 90rpm (1.5m s^{-1}, k_La 0.25min^{-1}). This graph when compared to Fig 5.4 (60rpm), shows that the boundary between aerobic and anoxic conditions has moved down the tank to around 800mm, from under 400mm with this increase in aerator speed. Also the percentage dissolved oxygen saturation of the waste water has also increased by around 10-15% due to this increase in aerator speed. It should, however, be noted that the oxygen concentration reaching the deepest sample point has not increased at all, indicating that the Lytag layer at this depth is acting as an anoxic/anaerobic layer, although there was still some dissolved oxygen present. This dissolved oxygen could be due to its influent concentration as the sample point was only a couple of mm off the base of the tank. This raises the question of how far into the Lytag layer from the base is it before the Lytag becomes fully anaerobic. This now becomes a design criterion, as the minimum Lytag depth for anaerobic conditions is now constrained from both above and below.

Fig 5.9 shows the BOD time profiles for this test, and clearly shows that BOD is removed throughout the tank depth. The rate achieved in the top 800mm of the tank varies between 30-35mg L^{-1} h^{-1}, with the higher rate at the surface. This is a slight increase over the previous results (60rpm) and can be attributed to the increase in aeration. Over the entire aerobic zone there is a reduction of 85%, which compares favourably with the previous results (at 60rpm) and with other plant such as RBC's at around 80-90% reduction in BOD (Wantabe *et al.*, 1993).

When the rate of BOD reduction in the Lytag is looked at more closely it has increased to 25mg L^{-1} h^{-1} (150-200mg L^{-1}) from 12mg L^{-1} h^{-1} before the increase in aeration speed. This gives an overall decrease in BOD of some 50% in the Lytag layer, now giving reduction rates much nearer those of established anaerobic processes which achieve upwards of 80% (Goronszy, 1987; Johnson *et al.*, 1994) and is consistent with the slow doubling time of anaerobes (Prescott and Dunn, 1984).

Fig 5.10 shows the ammonia concentrations against time, and shows that there is significant ammonia removal down to 800mm, or effectively for the whole of the liquid layer. Ammonia removal rates for the top 800mm vary from 4.5-5.5mg L^{-1} h^{-1} (gradient of line), and are consistent not only for those achieved at 60rpm (Fig 5.6, top 400mm) but also for previous work (chapters 2-4). At 1200mm there was no noticeable ammonia oxidation, consistent with the lack of dissolved oxygen as shown by Fig 5.8, and are compatible with the notion of an anoxic zone. Fig 5.11, shows the nitrate time profiles for this test, and as expected there is a steady rise in nitrate as ammonia is oxidised for the

upper tank levels. Also there was no nitrate detected in the Lytag layer, as expected, as there was no ammonia oxidation in this layer.

5.3.3 Aerator speed 120rpm (2m s^{-1})

Fig 5.12 shows the dissolved oxygen concentration time profiles at 120rpm (k_La 0.65min^{-1}, 1kg O_2KWh^{-1}). Again oxygen remains depleted within the Lytag layer although higher now than at the lower speeds, with concentrations ranging from 15-25 percentage saturation or so, and probably unacceptable for anoxic performance. Fig 5.13 shows the BOD concentration time profiles for this rotational speed. In all cases there is considerable BOD removal, with levels decreasing in the upper reaches to below 100mg L^{-1} from approximately 400mg L^{-1} in the 8h (35-40mg L^{-1} h^{-1}) comparable with previous results.

Fig 5.14 shows the ammonia time profiles for this test and it demonstrates that there is considerable ammonia oxidation at a rate of 4-5mg L^{-1} h^{-1}, comparable with the previous results. Also the final value of 5-10mg L^{-1} shows that the plant is performing satisfactory for a loading of 50mg L^{-1} on an 8h period. Again there is no significant ammonia oxidation within the Lytag layer. Fig 5.15 shows the nitrate time profiles for the test, and as expected there was an increase in nitrate concentrations in the upper tank as ammonia was oxidised (Fig 5.14). Nitrate levels also increased in the Lytag layer, without any noticeable ammonia oxidation, indicating that there was some diffusion of nitrate between the upper bulk liquid and Lytag layers. Again there was no noticeable reduction in the nitrate concentration, which suggests that no denitrification occurred.

5.4. RESULTS OF COAXIAL TWO TANK SYSTEM.

Fig 5.16 shows the BOD time profiles for two consecutive (batch wise) tests in the two tank model, incorporating three distinct zones (Fig 5.2) an anaerobic/anoxic pre-treatment, aerobic (carbonaceous removal), and an aerobic (nitrification) zone. It is clear from the graph that there has been significant BOD removal in both the zone 1 (anaerobic/anoxic) and zone 2 (aerobic), with levels in the aerobic zone being reduced from 250mg L^{-1} to 50mg L^{-1} in 6h. This gives a rate of 30mg L^{-1} h^{-1} comparable with previous results (chapters 3 and 4), and with RBC's (20-30mg L^{-1} h^{-1}, Wantabe *et al.*, 1993). For the anaerobic/anoxic zone BOD is removed at 15mg L^{-1} h^{-1}, which is similar to previous results, but still lower than those rates found in purely anaerobic systems (Goronszy, 1987; Miyahara, 1994). Also the exit value (zone 3) showed no significant BOD removal; however there was only a very low value entering the system of 50-60mg L^{-1}, so the lack of removal is not unexpected.

Fig 5.17 shows the SS time profiles for this test, and the graph clearly shows that the majority of the suspended solids were removed in the first two zones, as expected. The anaerobic zone reduced SS levels from around 120mg L^{-1} to 75-80mg L^{-1}, with the aerobic zone removing a further 65mg L^{-1} at a rate of approximately 10mg L^{-1} h^{-1}. The final zone reduced the SS levels to around 5mg L^{-1}, which compares favourably with literature expectations where removal of SS down to discharge limits (5mg L^{-1}, Roberts,

1995) without sedimentation can be considered an achievement (Zachritz *et al.*, 1993; Netzband, 1994).

Fig 5.18 shows the ammonia time profiles for two consecutive (batch wise) tests in the two tank model (Fig 5.2), and the graph clearly shows that as expected there was no significant ammonia oxidation occurring in the anaerobic zone throughout the test. However, there was significant ammonia oxidation in the aerobic zones where it was reduced from 35mg L^{-1} to 10mg L^{-1} in 6h (4mg L^{-1} h^{-1}) comparable with the previous results (chapters2-4). Fig 5.19 shows the nitrate time profiles for this test and it is clear from the rise in nitrate levels from 10 to 40mg L^{-1} in 6h, coupled with a reduction in ammonia (Fig 5.17) that nitrification is occurring.

5.4.1. Twin tank with partitioned inner tank.

Figs 5.20 and 5.21 show the BOD time profiles for the nominal four stage system over a five day period. It is clear from the graphs that the majority of BOD removal is occurring in zone 2 (aerobic outer zone) where BOD is reduced from 250 to 50mg L^{-1}, in 6h (rate of around 30mg L^{-1} h^{-1} comparable with previous findings (chapters 2-4)). Zone 1, the anaerobic pre-treatment stage also demonstrates BOD removal throughout the test, with levels being decreased by 25% (100mg L^{-1}) comparable with previous results (see above) but still lower than industry standards (Goronszy, 1987; Miyahara, 1994). There was little or no BOD removal in the other zones predominantly because of the low initial concentration entering these zones (less than 80mg L^{-1} (zone 4 aerobic nitrification) and around 20mg L^{-1} in zone 3). Even here there was still some BOD removed with final

effluent quality being less than 20mg L^{-1}. However it was noticed that due to the low transfer BOD concentrations other activities were suffering such as final denitrification so initial loading was increased at 78h from 400 to 1000mg L^{-1}. This increase in loading caused an increase in removal rates in the zone 1, from 15 to 30mg L^{-1} h^{-1} (anaerobic/anoxic), whereas they remained unchanged in Zone 2. Zone 3 (aerobic, nitrification) also showed an increase in removal rate with a doubling from 10 to 20mg L^{-1} h^{-1}. However, the final effluent quality did not alter, and still maintained a discharge quality of 20mg L^{-1}.

The corresponding suspended solids time profiles are shown in figs 5.22 and 5.23. It is clear from the graphs that again the majority of SS (70%) is removed in zones 1 (anaerobic) and 2 (aerobic outer zone). The final value of around 5mg L^{-1} is consistent with earlier work and is of discharge quality (Roberts, 1995) and is comparable with other treatment systems (Zachritz *et al.*, 1993; Netzband, 1994) with some 95% removal. Also once the loading was increased at 78h, from 130mg L^{-1} to 250mg L^{-1}, there was no change in the final quality of around 5mg L^{-1}, a 98% removal. The majority, some 80%, of this removal were again in the first two zones.

Figs 5.24 and 5.25 show the ammonia time profiles for the nominal four stage process, with the latter incorporating an influent concentration increase at 78h from 55 to 90mg L^{-1}. It is clear from Fig 5.24 that nitrification is occurring mainly in zone 2 (aerobic outer annulus) where it decreased at around 5mg L^{-1} h^{-1} which is consistent with previous results (chapters 2 and 3) and trickling filters (Wik *et al.*, 1995). Zone 1, the anaerobic

pre-treatment section, showed little ammonia oxidation over the period of the test as expected due to the anaerobic conditions. There was also very little ammonia removed in Zone 4, the nominal denitrification stage; however there was less than 10mg L^{-1} entering this section so this is not unexpected. Zone 3, which was the inner annulus aerobic (nitrification) section also, had a low ammonia oxidation rate over the initial 78h, again due to the reduced influent levels of ammonia (less than 15mg L^{-1}). However the rate did increase once the ammonia load was increased at 78h, to around 5mg L^{-1} h^{-1}. Zone 2 also showed a decrease in rate at this stage to 3mg L^{-1} h^{-1} (Fig 5.25). When this is compared to the BOD time profiles it is clear that the reduced nitrification rate in the outer annulus coincides with an increase in BOD levels (300 to 450mg L^{-1}) while the level in the inner annulus is below 275mg L^{-1}. Previously carbon competition presented an obstacle to effective ammonia oxidation when oxygen was depleted whereas here, in contrast, we see that there is effective concurrent oxidation of carbon and ammonia. With excess oxygen available, nitrification can occur simultaneously with BOD removal. The final ammonia level from both graphs varies only slightly between 0 and approximately 5mg L^{-1}, well within NRA discharge limits.

Figs 5.26 and 5.27 show the corresponding nitrate time profiles for this test and show an increase in nitrate levels in zone 2 from 0 to around 30mg L^{-1} consistent with the ammonia oxidation shown in Fig 5.24. The nitrate concentration increases further in zones 4 to approximately 40mg L^{-1}, again consistent with the ongoing ammonia oxidation shown in Fig 5.24. In Zone 4 there are occasional decreases in the nitrate level of some 10mg L^{-1}, but nothing to indicate any significant denitrification. Once ammonia influent

concentrations were increased (55 to 90mg L^{-1}) at 78h there was a corresponding rise in the nitrate levels as it was oxidised, which gave an influent level to Zone 4 of around 65mg L^{-1}. This increase in the nitrate levels was accompanied with a rise in BOD to approximately 100mg L^{-1}. This seemed to have the combined effect of reducing the effluent nitrate value by some 30-40%. This denitrification was consistent with earlier work (see chapter 4) but was still lower than comparable processes which achieve 80% reduction or better (Takamizawa *et al.*, 1993; Timmermans *et al.*, 1983). It is noted that further extended work on denitrification is required before any claims on consistent denitrification can be made.

5.5. BROADER DISCUSSION.

Although seemingly sustained effective oxidative treatment of BOD and ammonia and reductive removal of nitrate products has been demonstrated in the tests reported here, perhaps the major remaining issue relates to possible accumulation of solids within the system. In particular attention is drawn to the head space above the Lytag layer and below the partitioning shelf which was investigated following completion of these tests when the system was stripped down for cleaning and a substantial layer of solids was observed. This begs a question on how long it would be before this solids accumulation would break through to the aerobic layer, causing the reduction in performance found in chapter 4. The layer of solids found was some 10mm deep, in a space of 100mm between shelf and Lytag, and had taken some 3 months to develop. Assuming that there would be

no bacterial degradation of these solids, and a continuing uniform deposition of these solids a period of some 30 months would need to elapse before breakthrough would occur. At this point solids removal of some description would be required, either as an integral part of the plant or by its manual strip down and clean.

The anaerobic pre-treatment section gave a reasonable performance with BOD removal throughout the tests being around 25% (over all stages reduced to around 20mg L^{-1}) which is in-line with industry standards (Goronszy, 1987; Miyahara, 1994). Suspended solids, however, removed in line with discharge limits (95% removed over whole plant) with up to 70% being removed in the anaerobic first stage comparable with other purely anaerobic systems (Zachritz *et al.*, 1993; Netzband, 1994). It should also be noted that there was no inhibition of the aerobic bacteria by the anaerobic liquor being passed to it as expected due to their sensitivity to low dissolved oxygen levels (Laanbroek *et al.*, 1994), as the aerators were able to provide sufficient oxygen to the system (1kg O_2 kW h^{-1}).

Denitrification was achieved in this evaluation but it is interesting to ask to what extent it might be increased by controlled enhanced supply of carbon substrate (e.g. methanol). Although denitrification was observed to some extent in the inner Lytag layer, it was only after the carbon loading entering this zone had been boosted indicating that the denitrifiers were carbon limited. As carbon is the limiting substrate then boosting the carbon concentration to meet this demand for the plant as currently configured will require careful control to ensure that the residual BOD is not excessive. If the discharge level does, perhaps, become too high then the liquor could be recycled into the pre-treatment

zone for further treatment. This would also allow the nitrate levels to be further reduced in this anaerobic zone.

A plant of given size and internal layout with defined volume fractions of packed and mobile supports with aerators running at defined efficiencies at a defined speed will meet a defined oxidation duty and also a defined reduction duty with regard to post treatment denitrification, so that the whole thing can be balanced up because there are sufficient degrees of freedom with regard to sizing and intensity in each of these zones. However, this does not necessarily fix the rate at which BOD is degraded, since this depends on other factors, such as initial loading and the precise concentration of active volatile solids available, though over the range of initial BOD values we tested, in most cases BOD was degraded to near discharge levels. Because we are not controlling precisely the concentration of residual oxidisable carbon, in other words the BOD level entering the final anoxic zone, this is working by more hap and hazard than by tightly controlled regime. So, although we were lucky enough to be able to demonstrate that denitrification was taking place, the amount of electron acceptor (BOD) was not matched by electron-donor (nitrate) and therefore denitrification was incomplete. There seems to be no reason why tighter control regimes (resulting in virtually complete denitrification) should not be applied.

The feasibility of using wind power for this system does raise questions on performance with regards to wind intermittency (see chapter 2). As a guideline, approximately a 3 day down-time needs to be tolerated as a design consideration. This of course always

recognises that if one were to restrict the utilisation of this plant to windier areas such as coastal strips and hilly zones, one could guarantee variations which were merely diurnal (12h down time). At present there are no data on the nominal four stage system utilising wind power, but tests will need to be carried out to highlight any deficient areas or control problems. One such area already known to be doubtful is that of ensuring a sufficient supply of carbon to the denitrifiers in the final zone, while not overloading the preceding nitrification zone with BOD under intermittent aeration conditions. As stated previously there are strategies around at the moment for providing an adequate supply of carbon for the denitrifiers via recycle without any recourse to expensive methanol. However, if the system has been designed properly there is a window of opportunity in that chapter 2 shows that nitrifier intolerance occurs only for BOD values in excess of 100mg L^{-1}. This means that one can provide for sufficient carbon coming over that there is a residue for the denitrifiers following nitrification but not so much as to seriously inhibit the reactivation of the nitrifiers.

5.6 CONCLUSIONS

The presence of a presumptive anaerobic activity in the pre-treatment zone has provided adequate suspended solids removal and significant BOD removal. This Lytag layer has shown that it will retain solids not only in the matrix but also in the nominal four stage system between the support surface and the base of the shelf. This solids accumulation will require some form of removal but not until after several years of operation. Both of

the aerobic zones (BOD removal and nitrification) provided good removal rates when compared both to previous work and to published literature, and the only question concerns control of flows between these zones. The denitrification zone did not show reliable denitrification until the carbon level entering it was increased; however, once the carbon level had been raised, the presence of some denitrifying activity was noted. The unreliable nature of this denitrification again centres on the control of carbon levels entering each zone; as mentioned before, a more adequate form of control strategy should cure this reliability problem. Further work will be needed not only on control but also on the use of wind power, which will once again raise the question of intermittency.

Future work will, therefore, centre on further development work, particularly on scale-up of the half-metre and 2m units to produce a larger, more-practical plant. There will also have to be optimisation work to see if increasing the volume fraction of the mobile support pads would increase plant performance and reduce throughput times. There is also the question of pad life-time under normal working conditions, and whether cleaning the pads will be necessary, and if so, will this reduce the pad life-time. As stated above the control of this process will need to be looked at more closely, particularly if the full four stage system is adopted, as the control of carbon loading in each of the stages is critical. At present, very little has been done in this area, and is very much the next step required, as it could again increase performance, but more importantly could stabilise the complete process making it an even more attractive proposal/alternative to existing plant.

FINAL CONCLUSIONS

1 Competition between heterotrophic and nitrifying bacteria, for dissolved oxygen, following a period of oxygen starvation keeps the nitrifying bacteria "dormant" for several hours.

2 Defined recovery times for nitrifying bacteria following oxygen starvation, with defined BOD loadings. The optimal being 100mg L^{-1}.

3 Compartmentalization of the system so that the heterotrophic and nitrifying have separate zones removes this competition.

4 The use of various bacterial support matrices (mobile and fixed film submerged) improves plant overall performance.

5 Several years of operation, with no solids blockage problems encountered with the submerged fixed film support matrix.

6 The overall plant performance (four-stage system) is similar or betters that of conventional plant such as Rotating Bacterial Contactors and Trickling Filters.

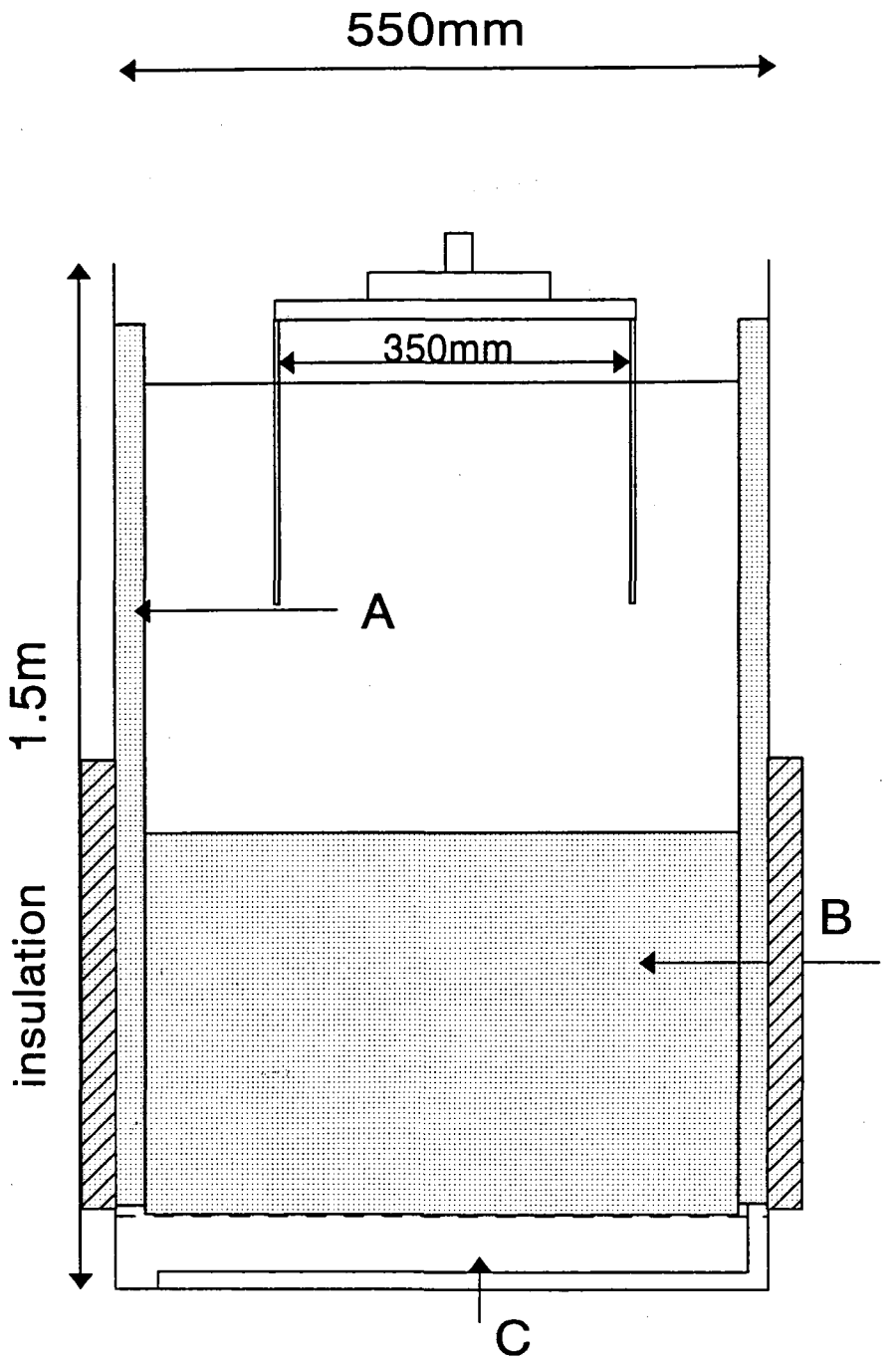


Fig 5.1 shows the single tank configuration used. A=baffle; B=Lytag (550mm); C=distribution plate. Tank was constantly aerated using two snorkel aerators (dimensions in text).

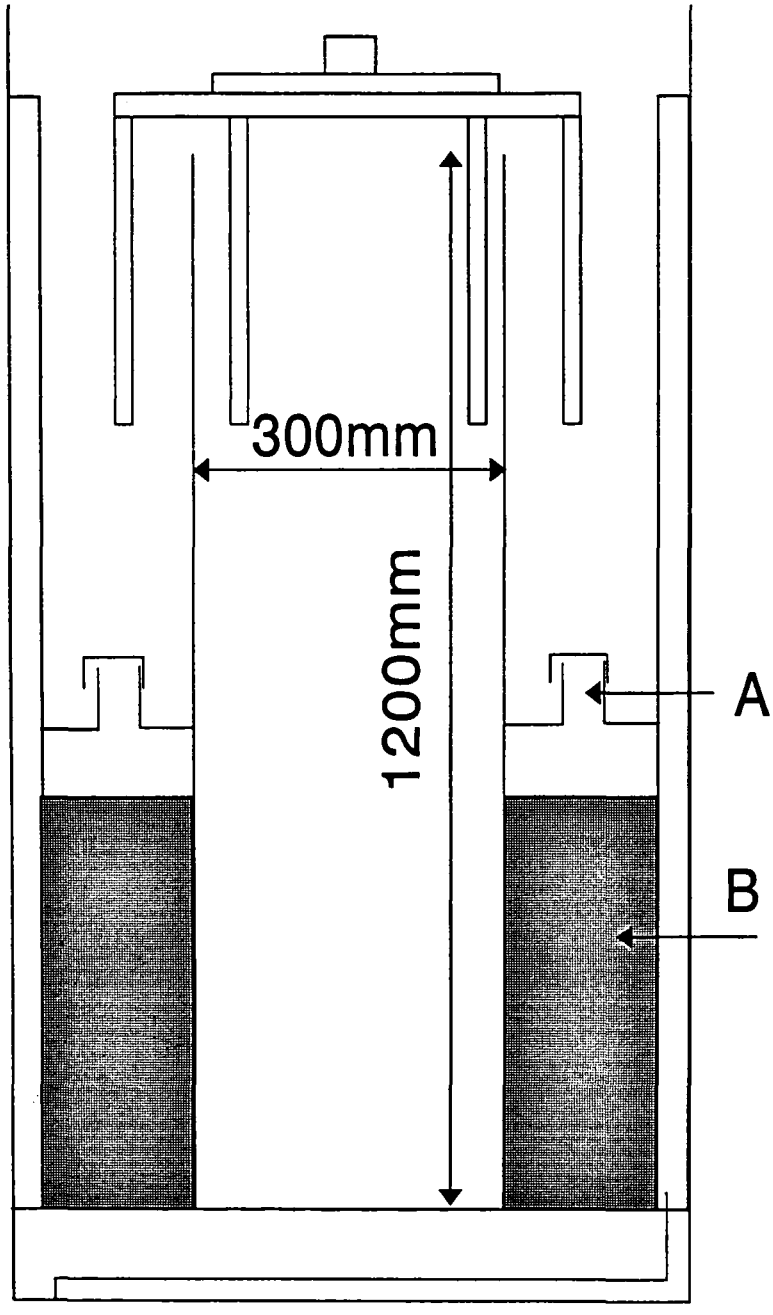


Fig 5.2 shows the three stage system including inner tank. A=retaining shelf for mobile support; B=Lytag layer(450mm). as before top layer aerated constantly using 4 snorkel aerators, 2 in each cylinder.

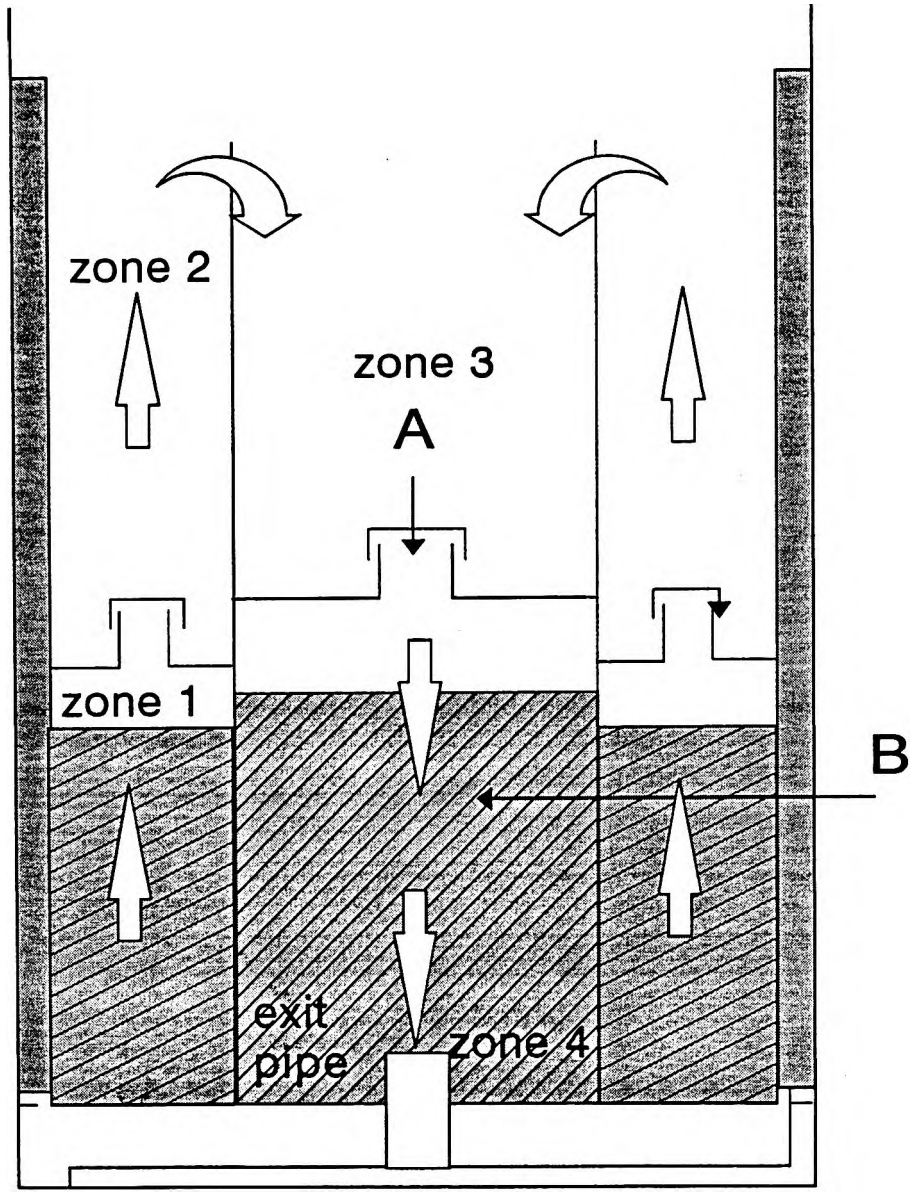


Fig 5.3 shows twin tank configuration each incorporating upper and lower zones for nominally a four stage system anaerobic, aerobic (BOD), aerobic (ammonia) and denitrification the inner cuylinder was partitioned by the addition of shelf A, to retain aerobic support pads. Lytag layer B (500mm) was for denitrification support. All tank sizes, are as in Figs 1 and 2. Flow is indicated by arrows.

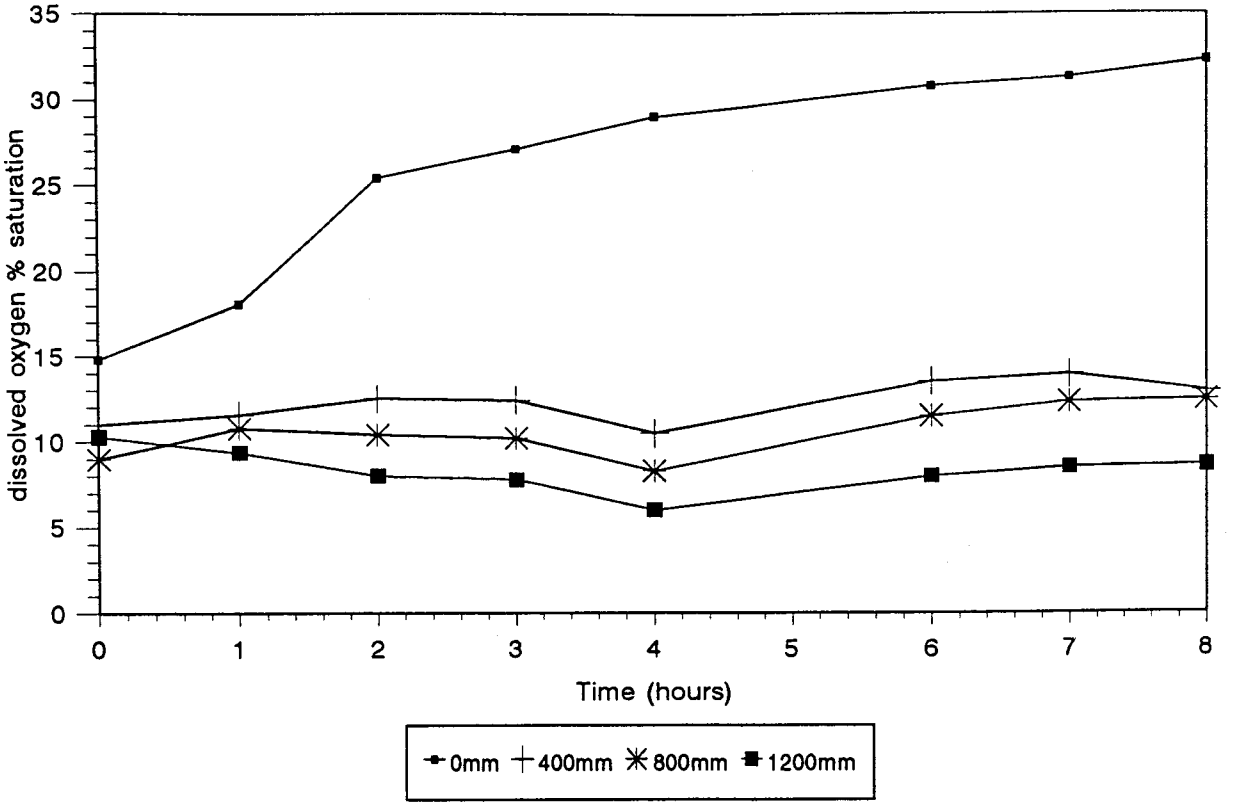


Fig 5.4. Dissolved oxygen levels against time, for the two stage system. sample depths were as shown. Aeration was constant throughout test with with tube speed 60rpm (1m/s)

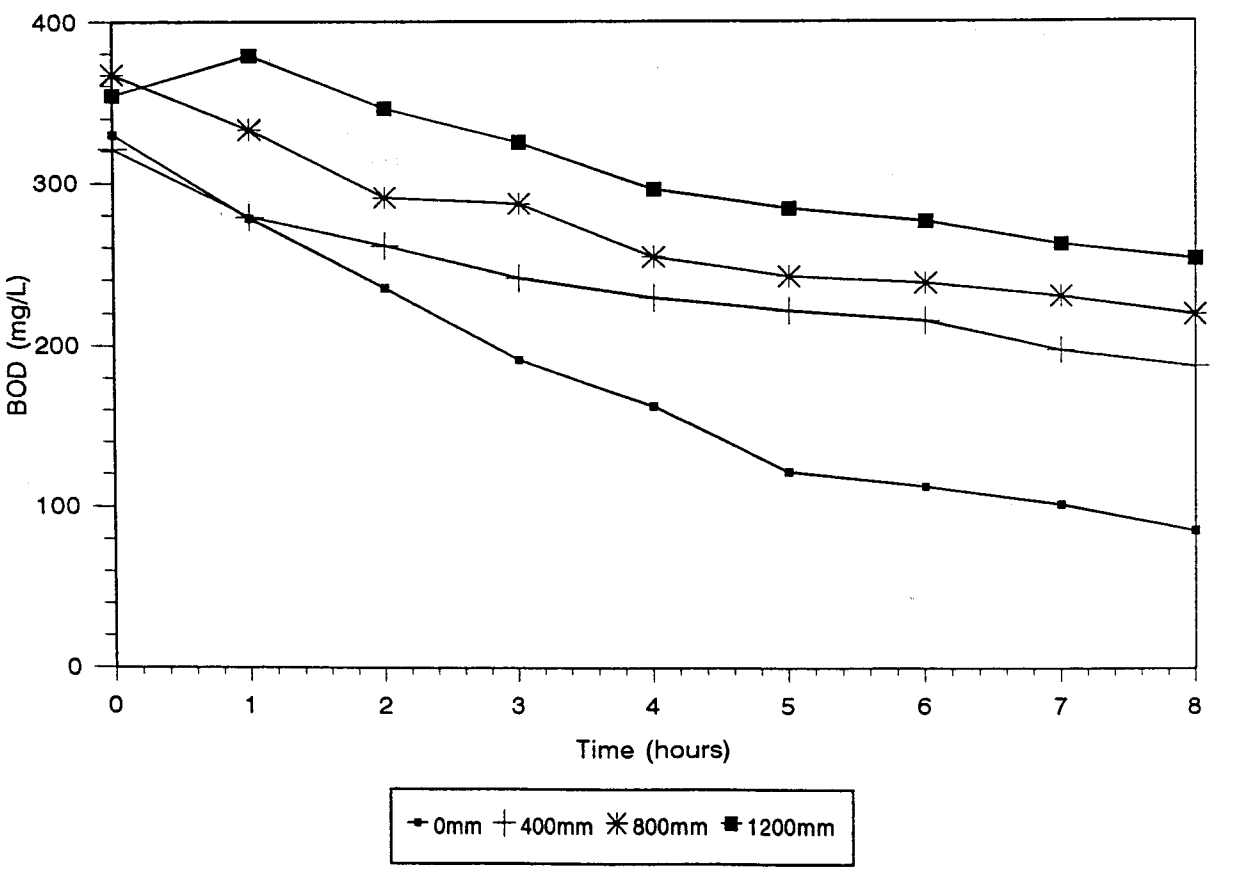


Fig 5.5. BOD levels against time conditions as for Fig 5.4

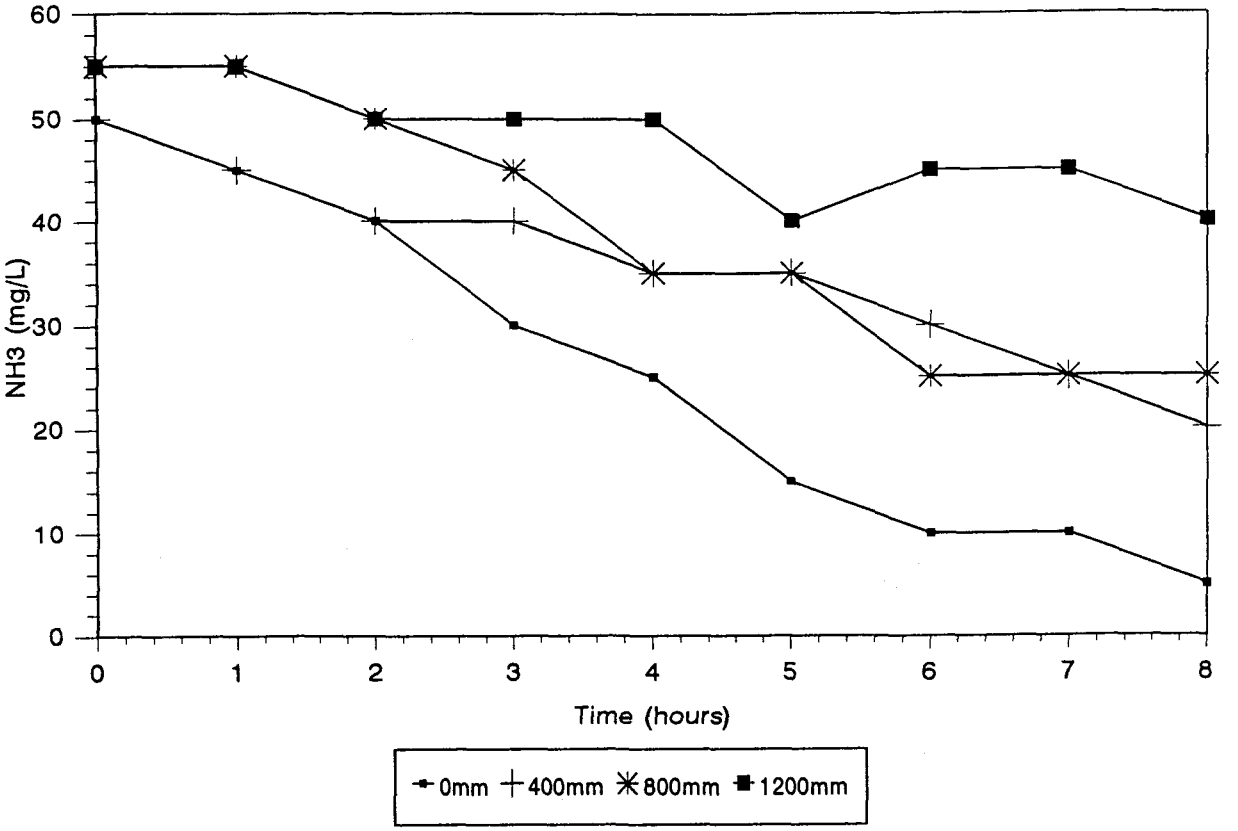


Fig 5.6. Ammonia concentrations against time for the four sample heights. conditions as for Fig 5.3.

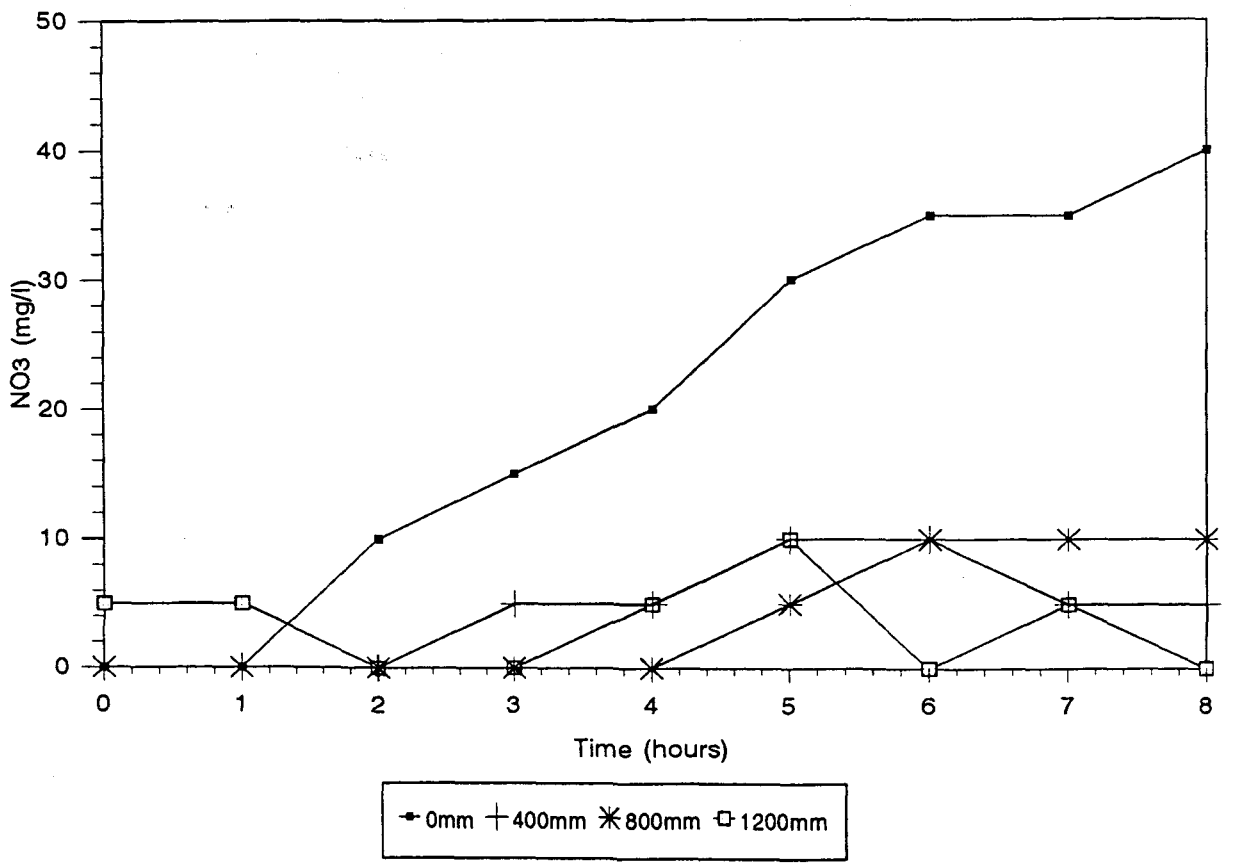


Fig 5.7. Nitrate concentrations against time for conditions described in Fig 5.4

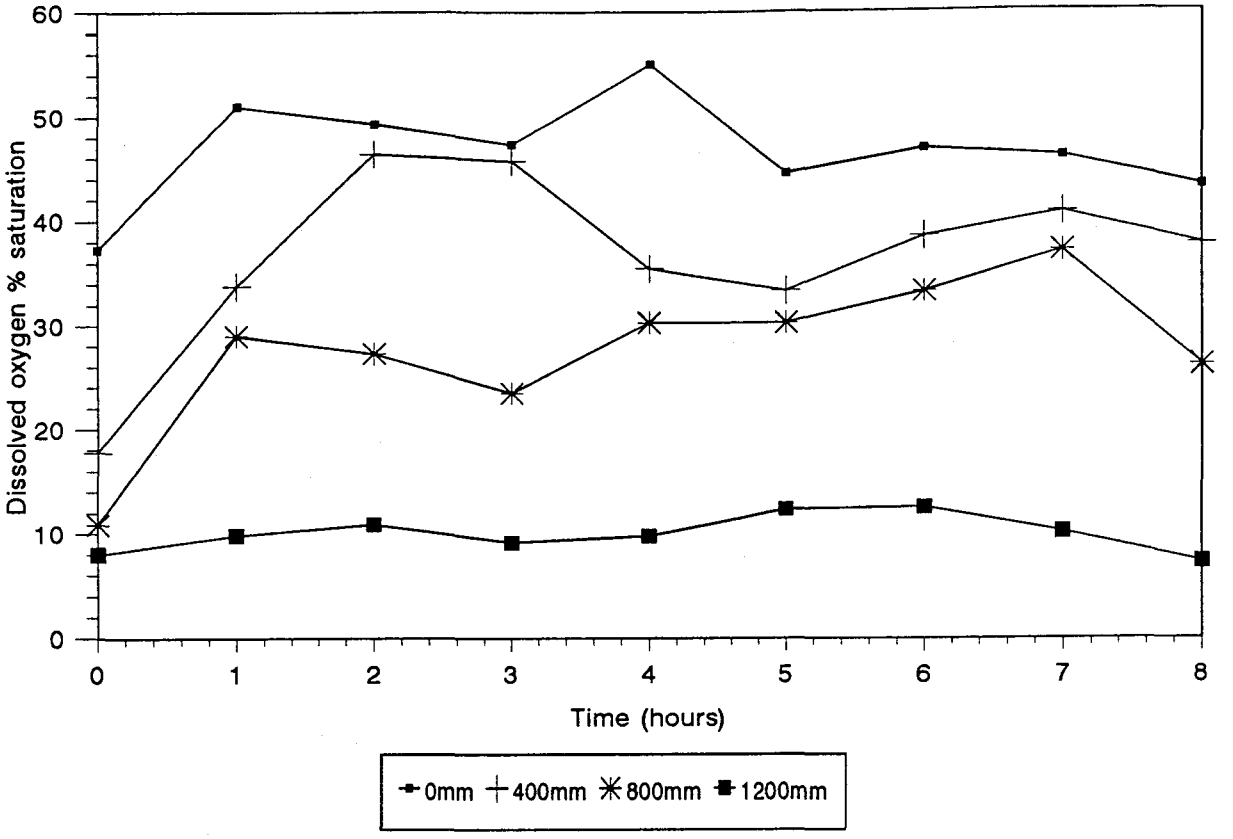


Fig 5.8. Dissolved oxygen levels against time for maintained aeration at 90rpm (1.5m/s). Sample depths were as shown

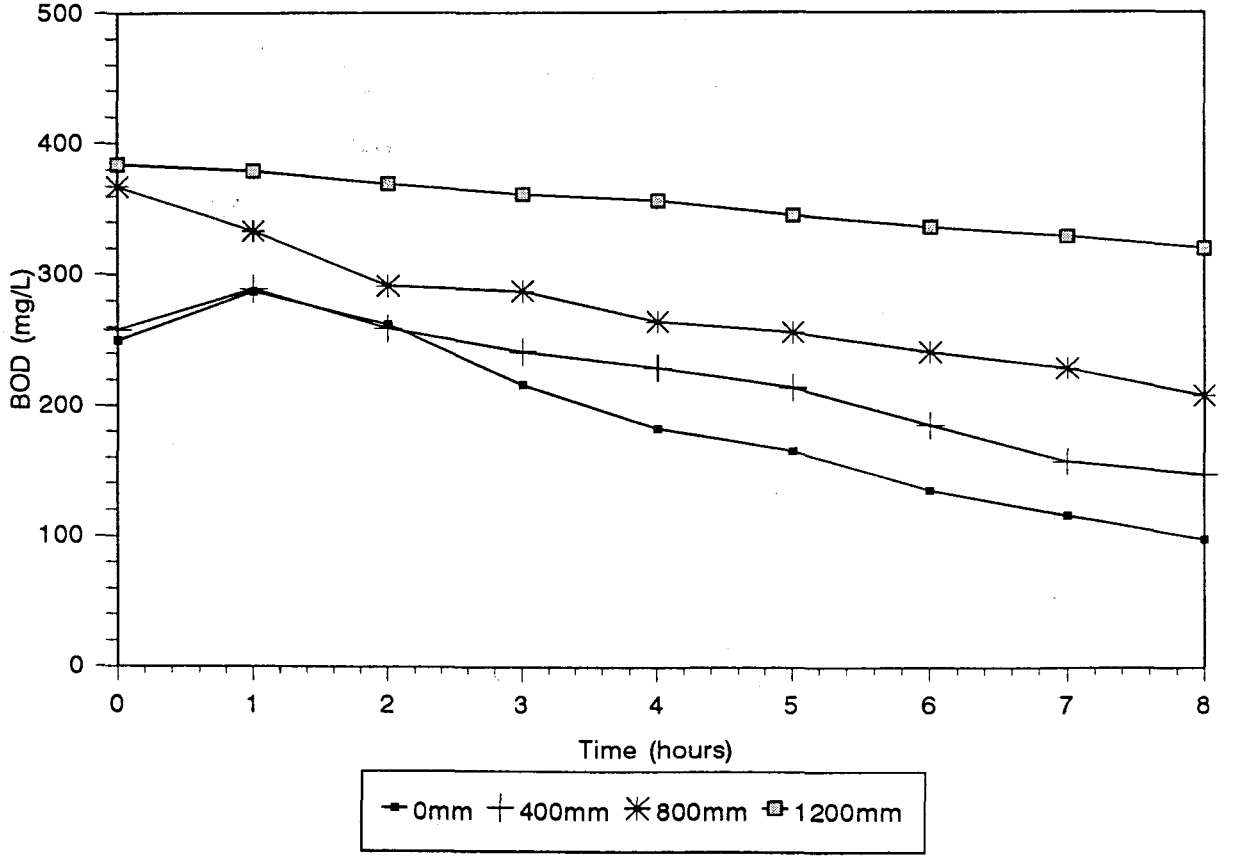


Fig 5.9. BOD time profiles at 90rpm, conditions as for Fig 5.8

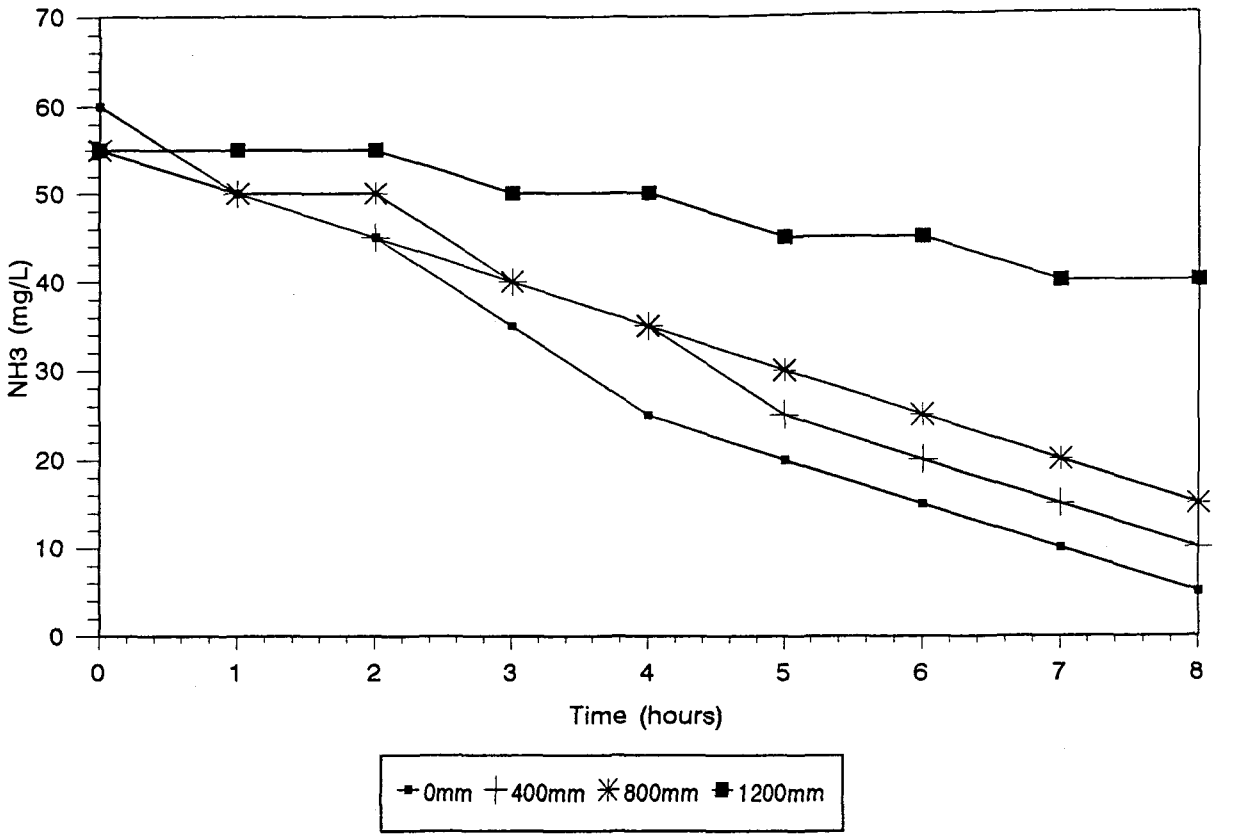


Fig 5.10. Ammonia concentrations against time for 90rpm. conditions as for Fig 5.8.

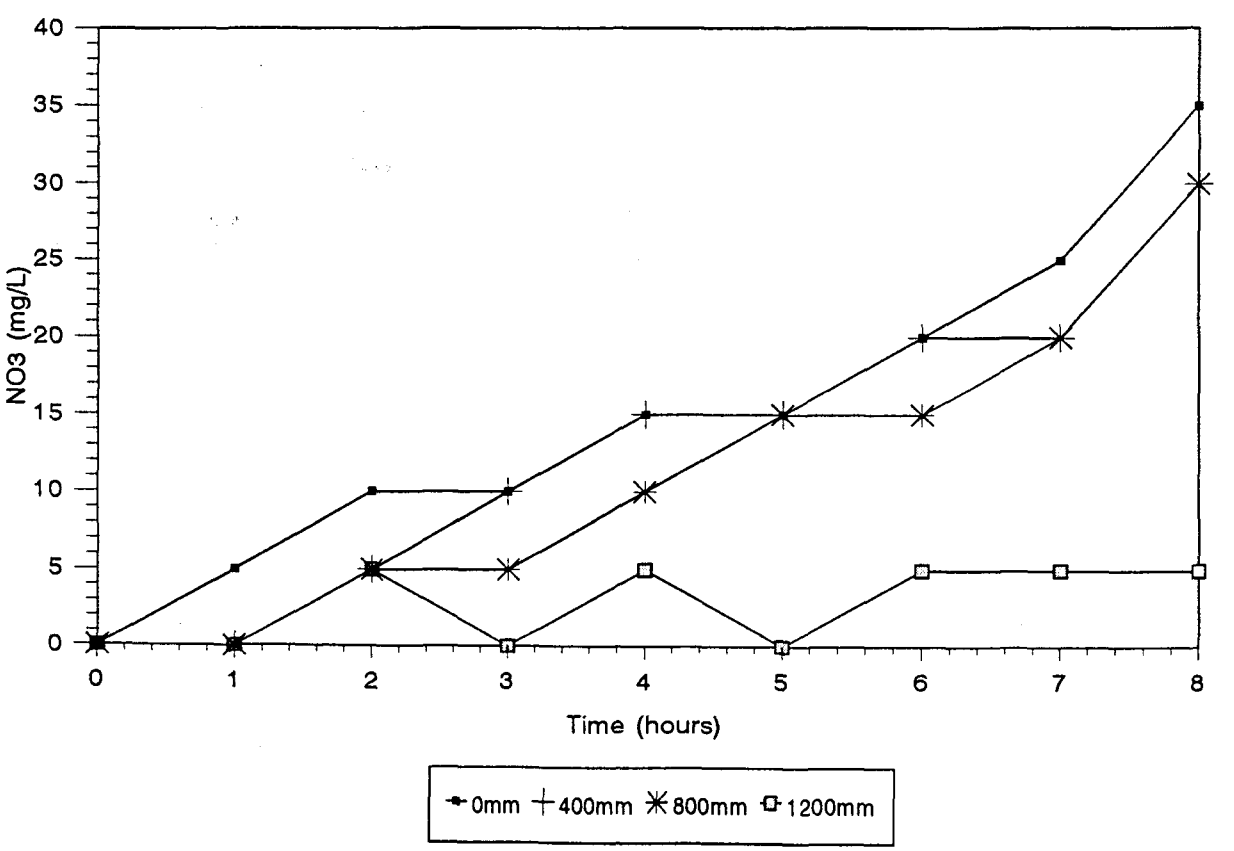


Fig 5.11. Nitrate concentrations against time for 90 rpm conditions as for fig 5.8

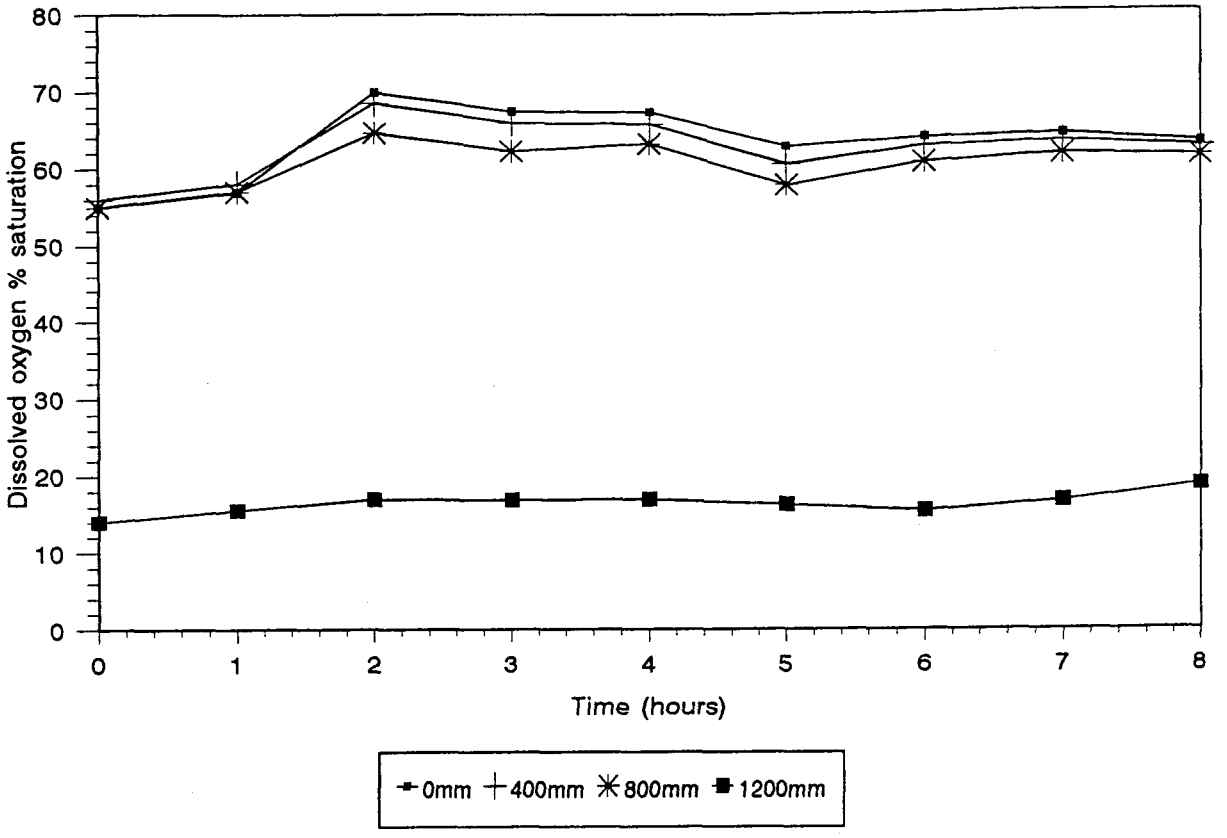


Fig 5.12. Dissolved oxygen time profiles for mainatained aeration at 120rpm (2m/s) Sample depths were as before.

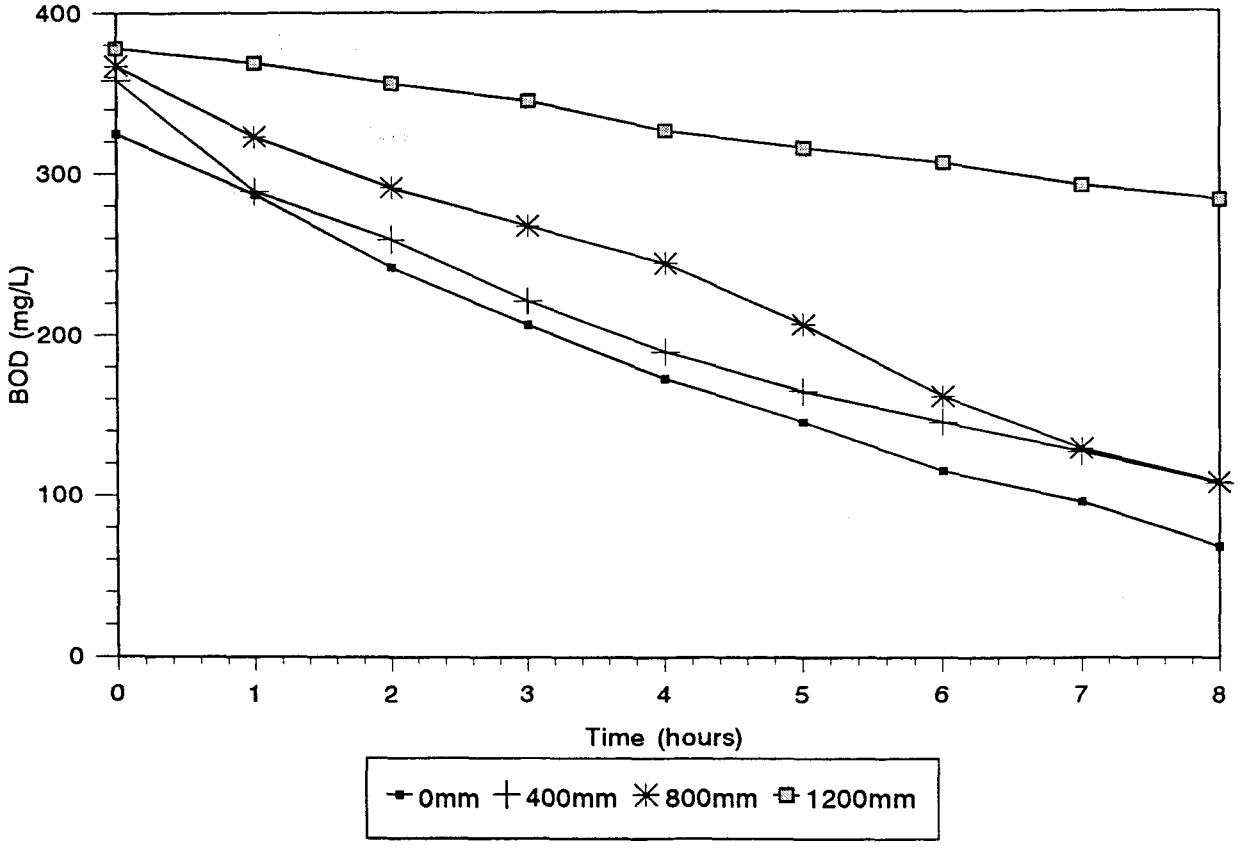


Fig 5.13. BOD time profiles for 120rpm conditions as for Fig 5.12

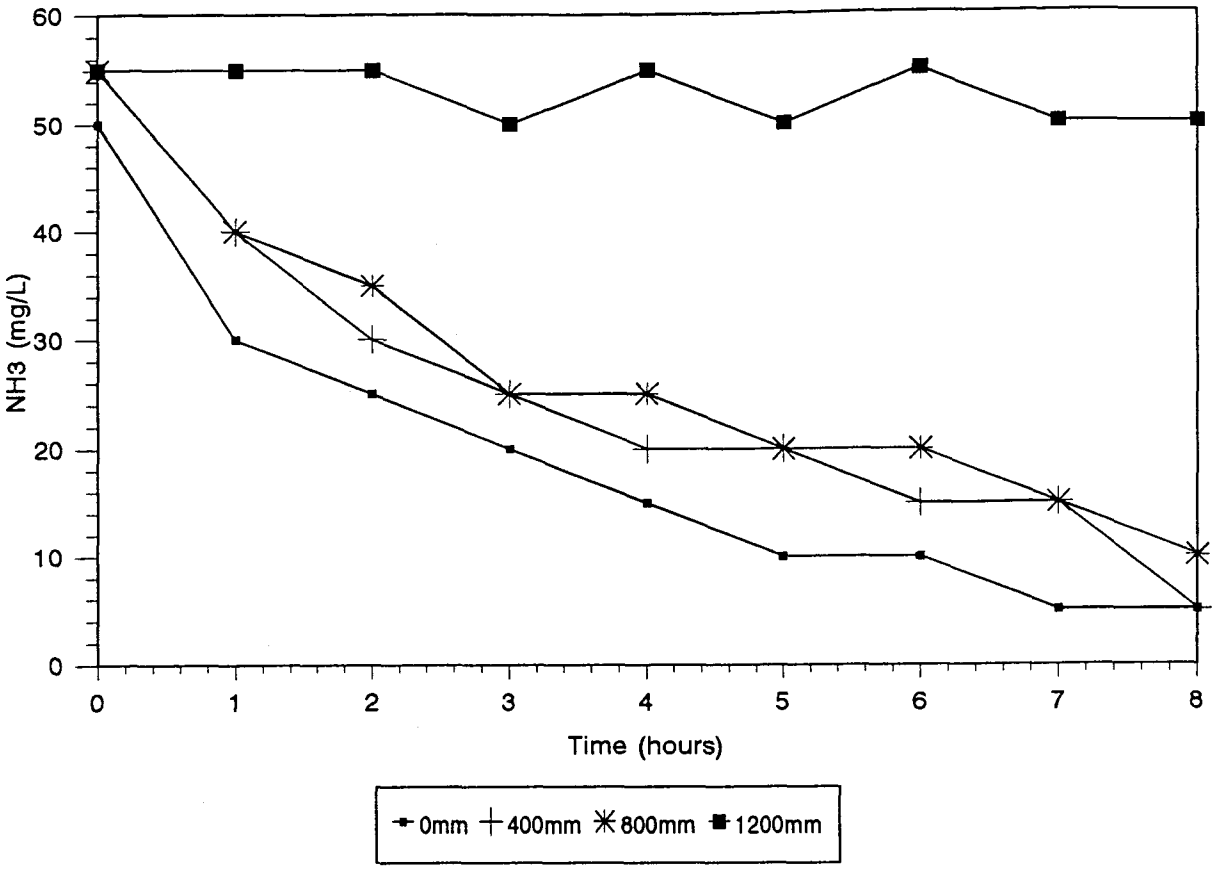


Fig 5.14. Ammonia time profiles at 120rpm conditions as for fig 5.12

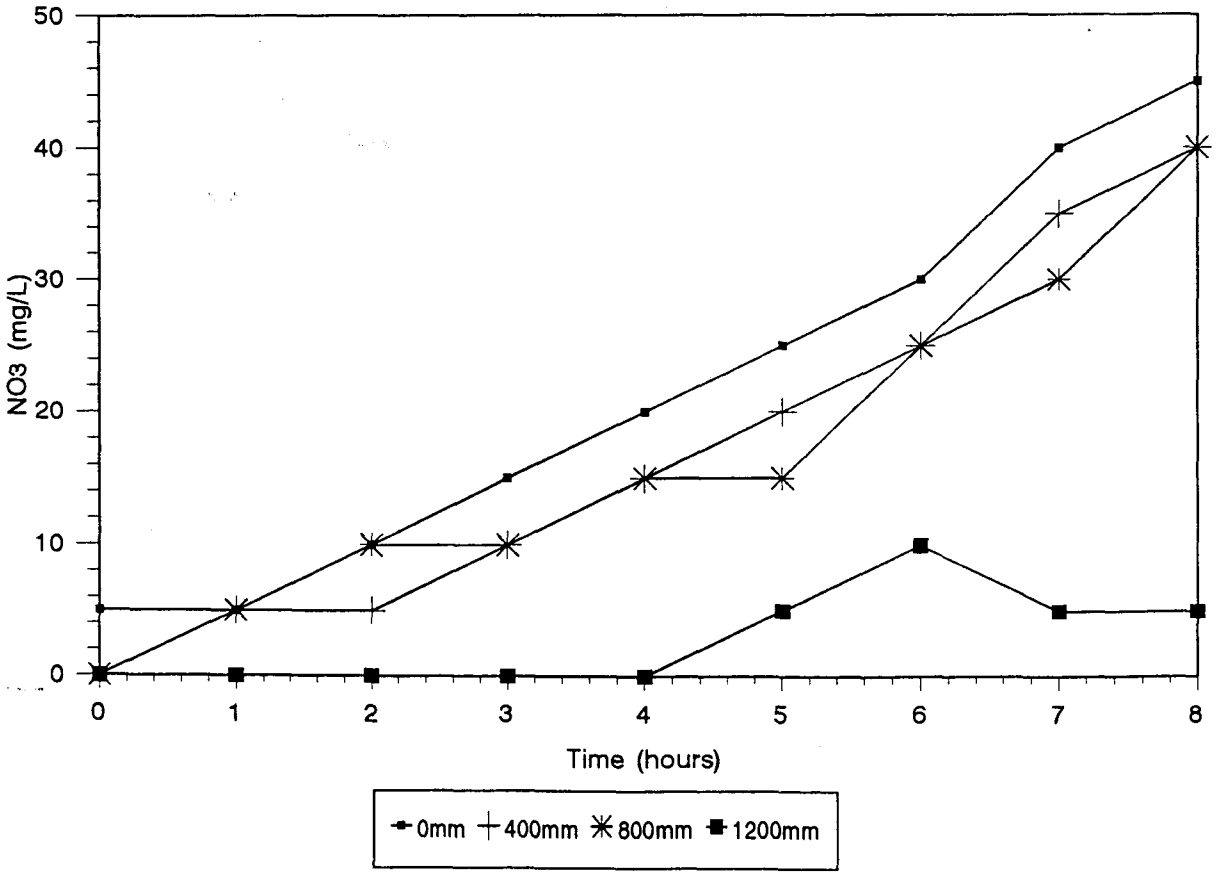


Fig 5.15. Nitrate time profiles at 120rpm conditions as for Fig 5.12

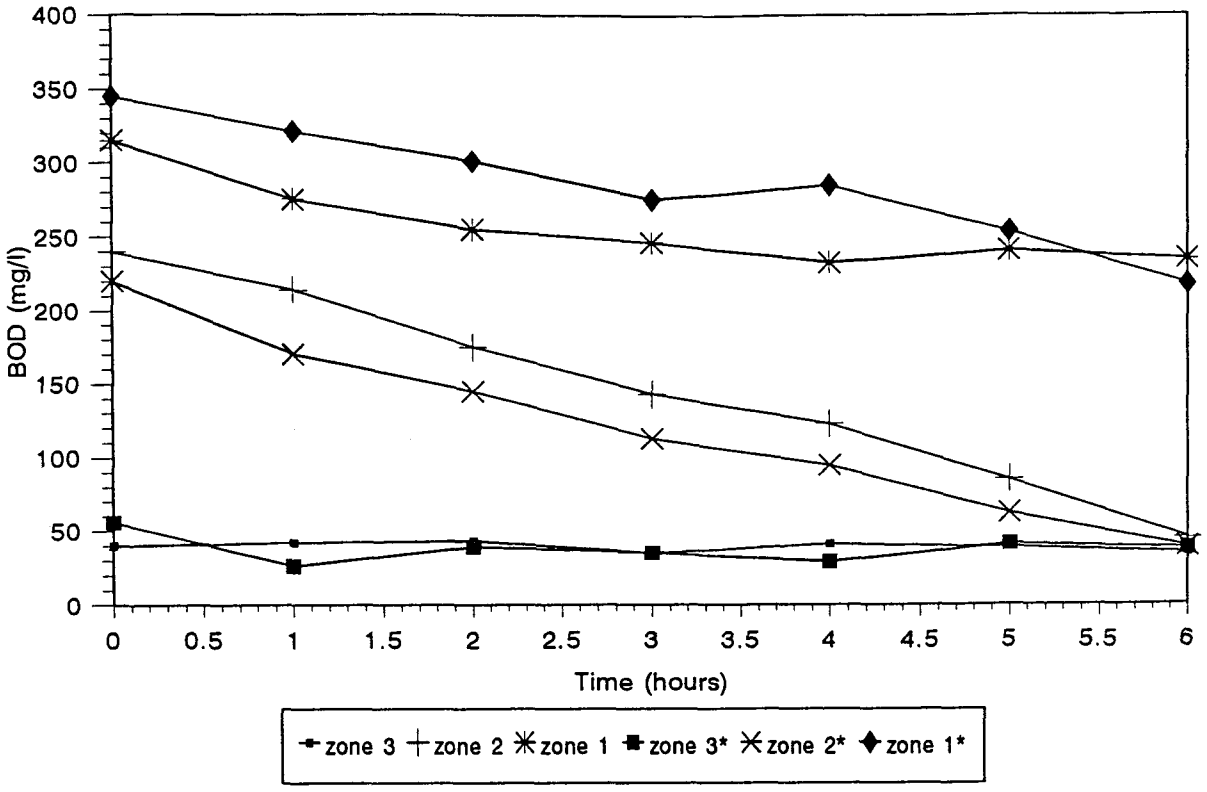


Fig 5.16. BOD time profiles for two tank configuration with 4 zones maintained aeration (125rpm 2m/s) results for two consequtive tests * denotes second Zone 3 = effluent exit, Zone 2 = outer tank surface, Zone 1 = outer lytag layer

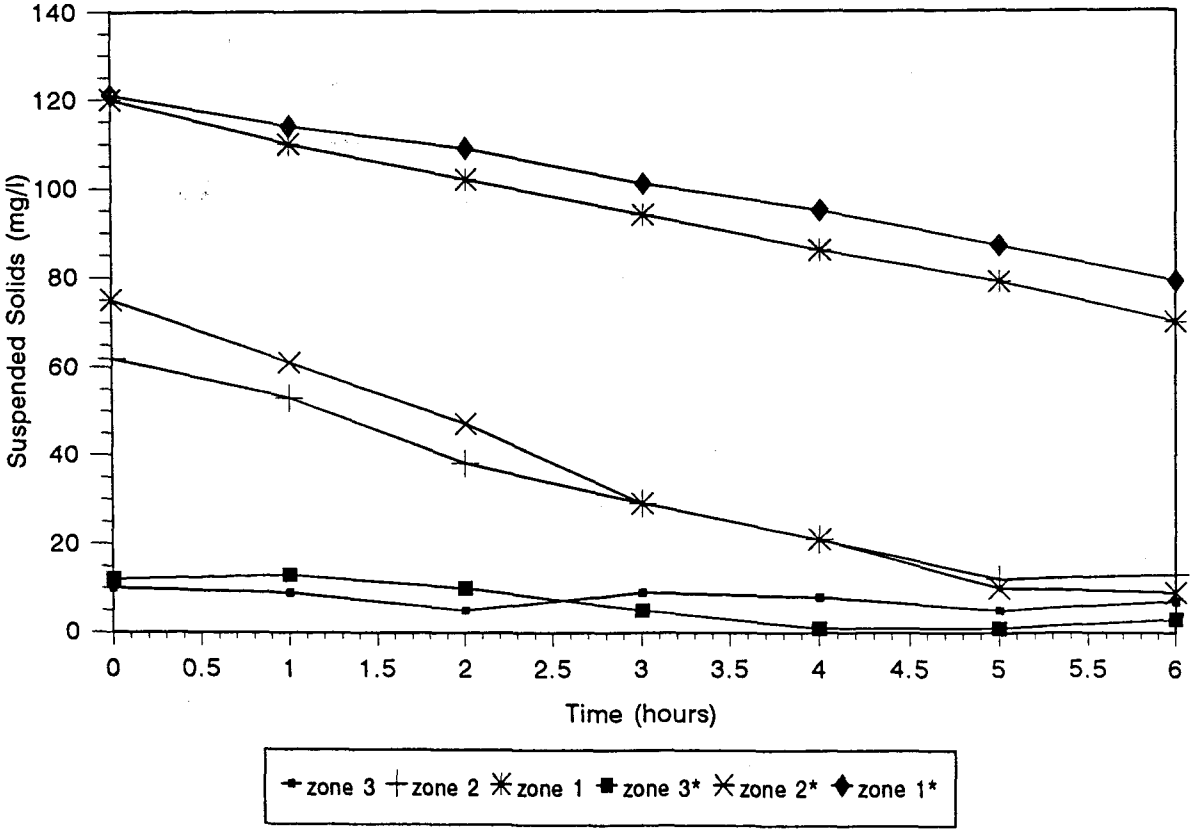


Fig 5.17. Suspended solids time profiles for three stage work conditions as for Fig 5.16, * denotes second consecutive test

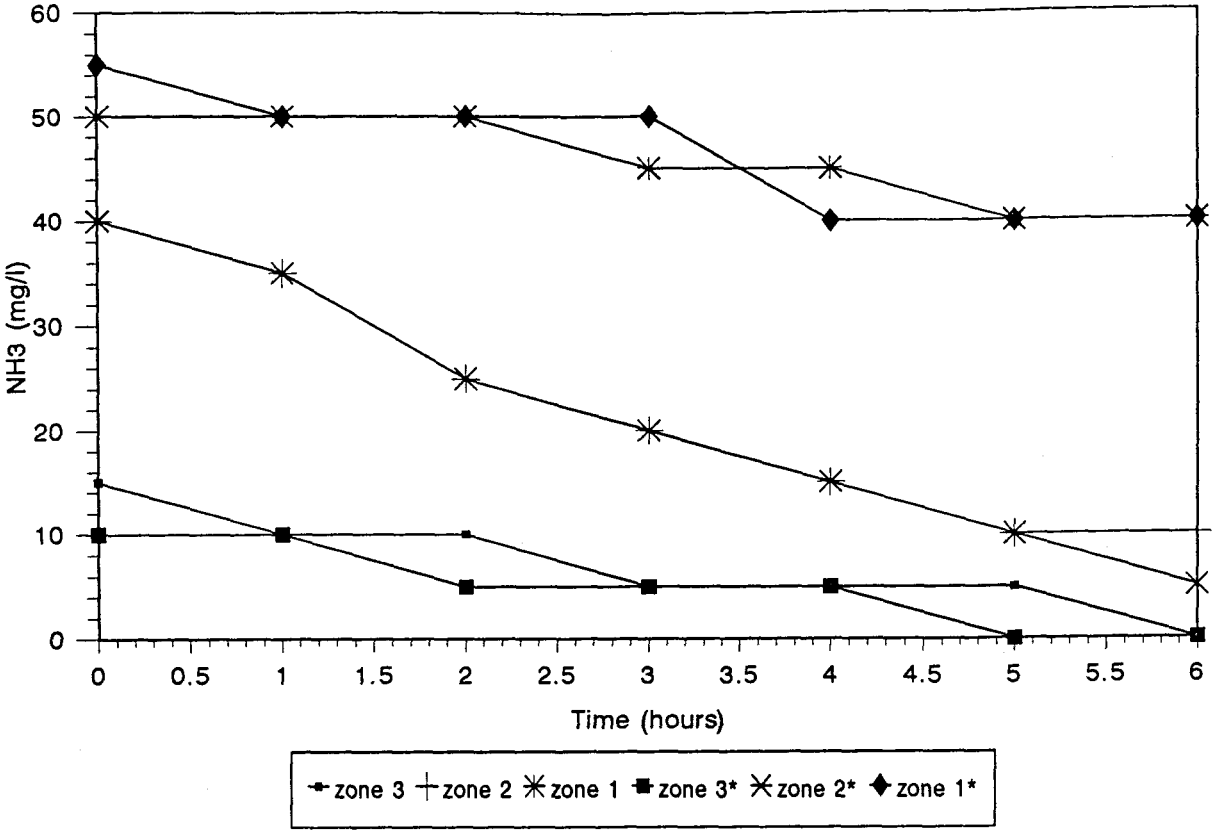


Fig 5.18. Ammonia time profiles for two consecutive three stage tests. conditions as for Fig 5.16, * denotes second test

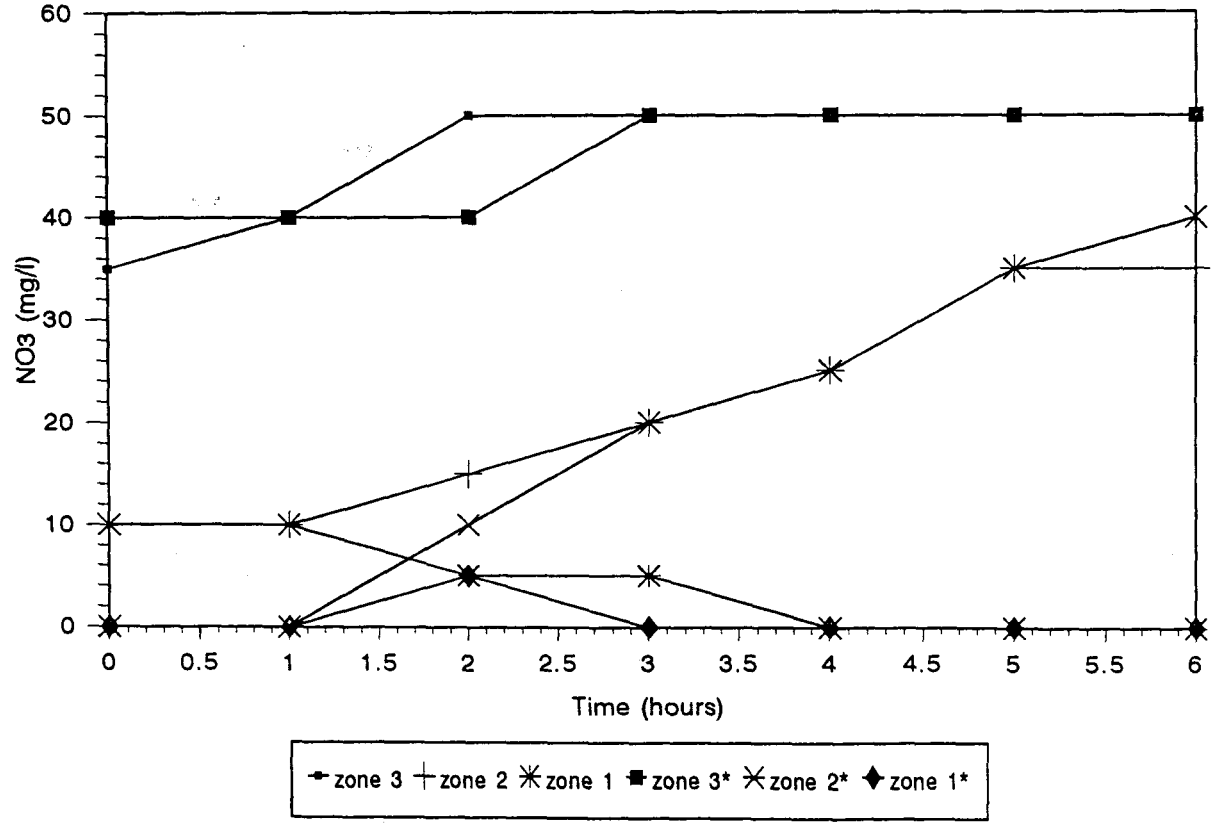


Fig 5.19. nitrate time profiles for two consecutive three stage tests. Conditions as for Fig 5.16, * denotes second test

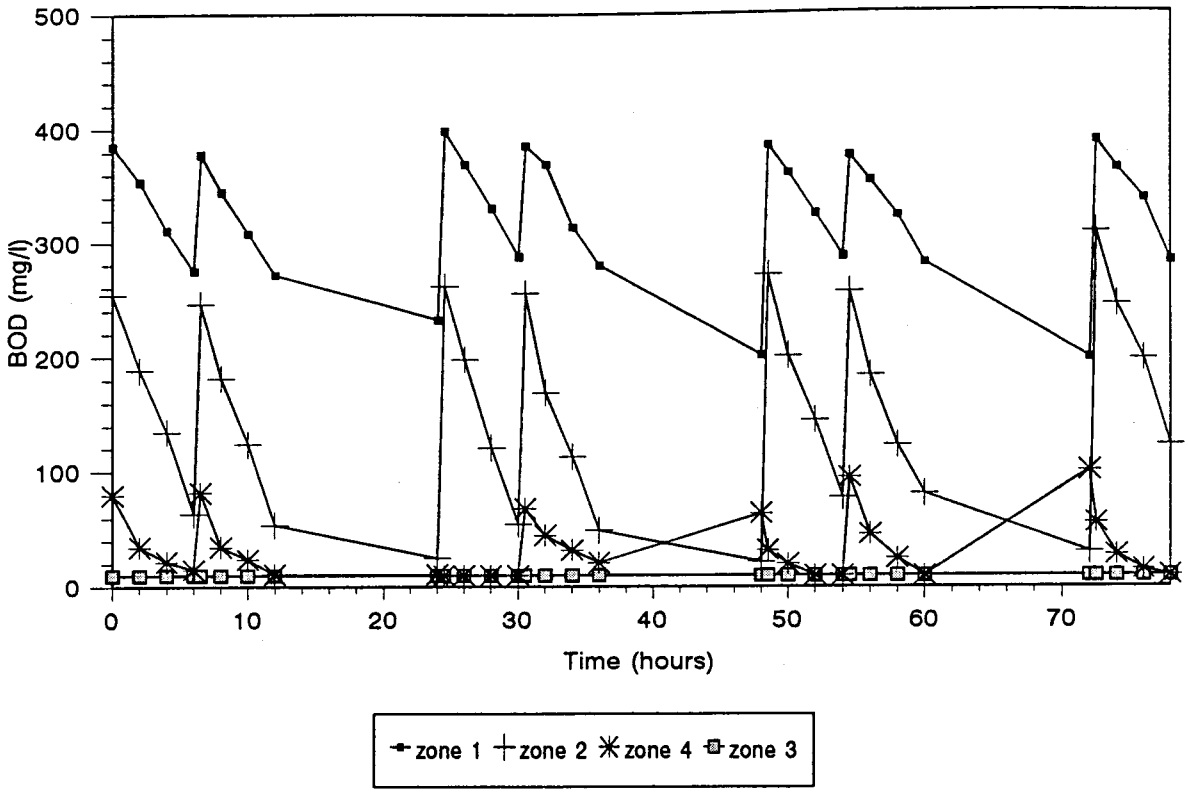


Fig 5.20. BOD time profiles for nominal 4 stage system with maintained aeration (120rpm 2m/s). influent addition at 6 hourly intervals over the first 12h of 30L (400mg/L).

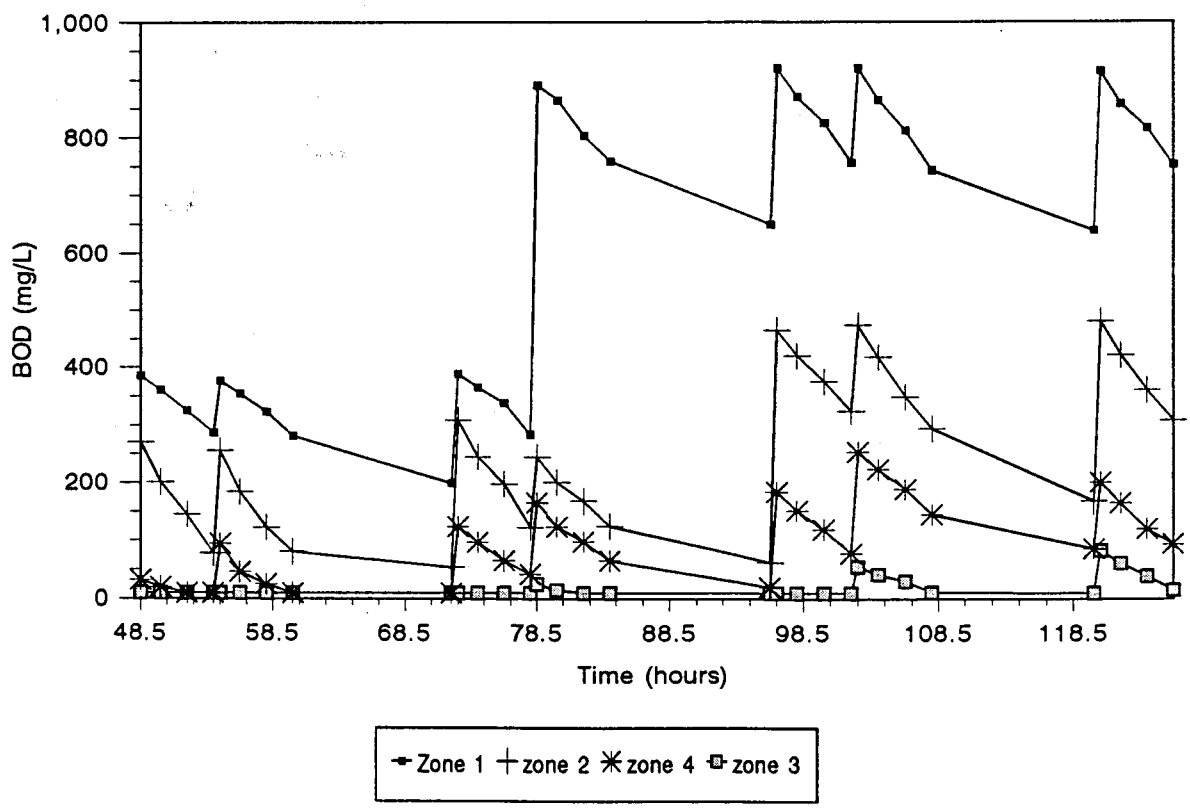


Fig 5.21. BOD time profiles for nominal 4 stage system conditions as for Fig 5.20. Influent concentration changed at 78h from 400 to 1000mg/L (30L).

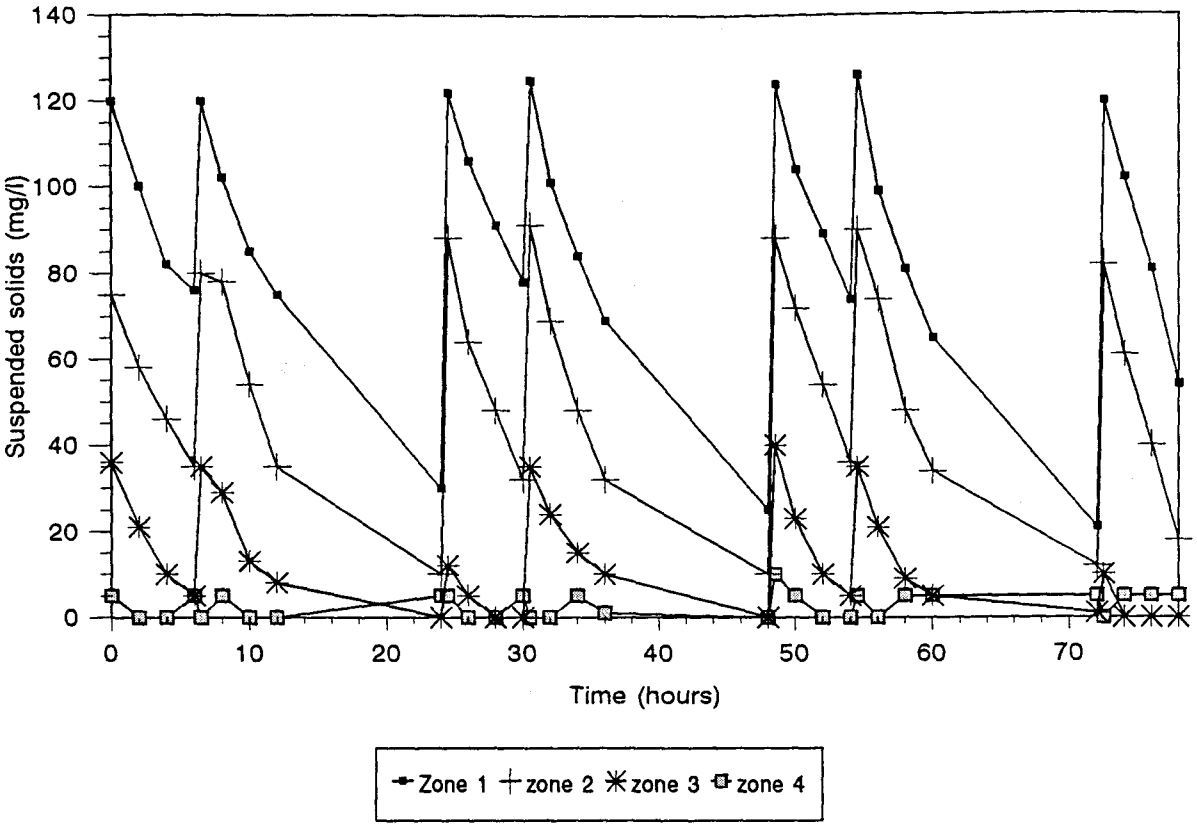


Fig 5.22. suspended solids time profiles for nominal 4 stage system. conditions as for Fig 5.20

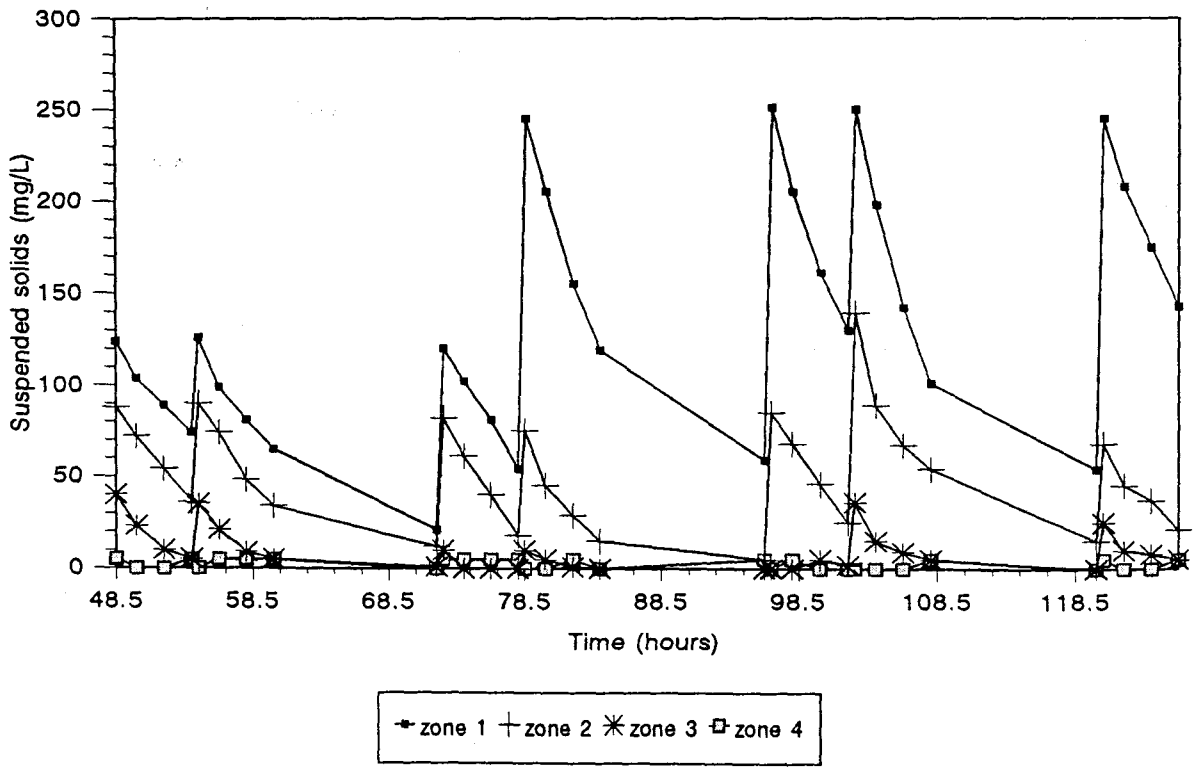


Fig 5.23. Suspended solids time profiles for nominal 4 stage system conditions as for Fig 5.20. Influent concentration changed at 78h from 130mg/L to 250mg/L (30L).

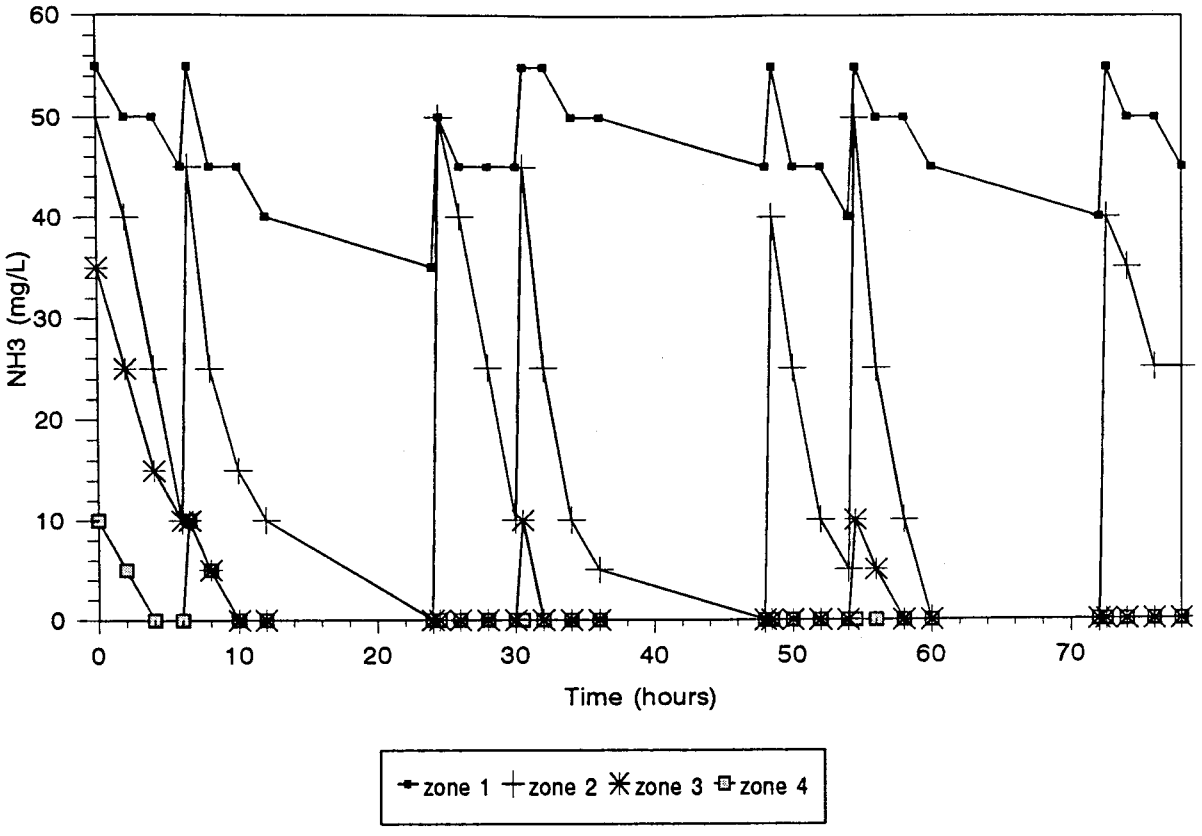


Fig 5.24. Ammonia time profiles for nominal four stage system Conditions as for Fig 5.20.

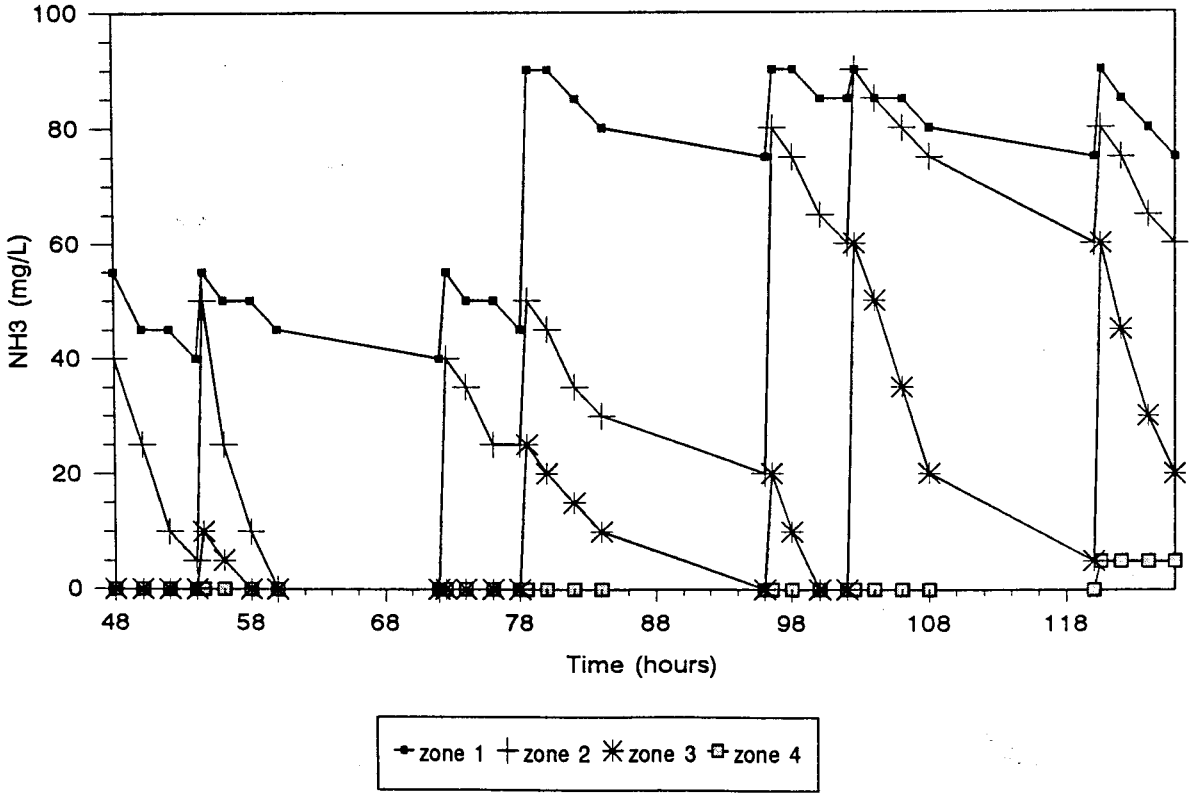


Fig 5.25. ammonia time profiles for nominal four stage system conditions as for Fig 5.20. Influent concentration increased at 78h from 55mg/L to 90mg/L.

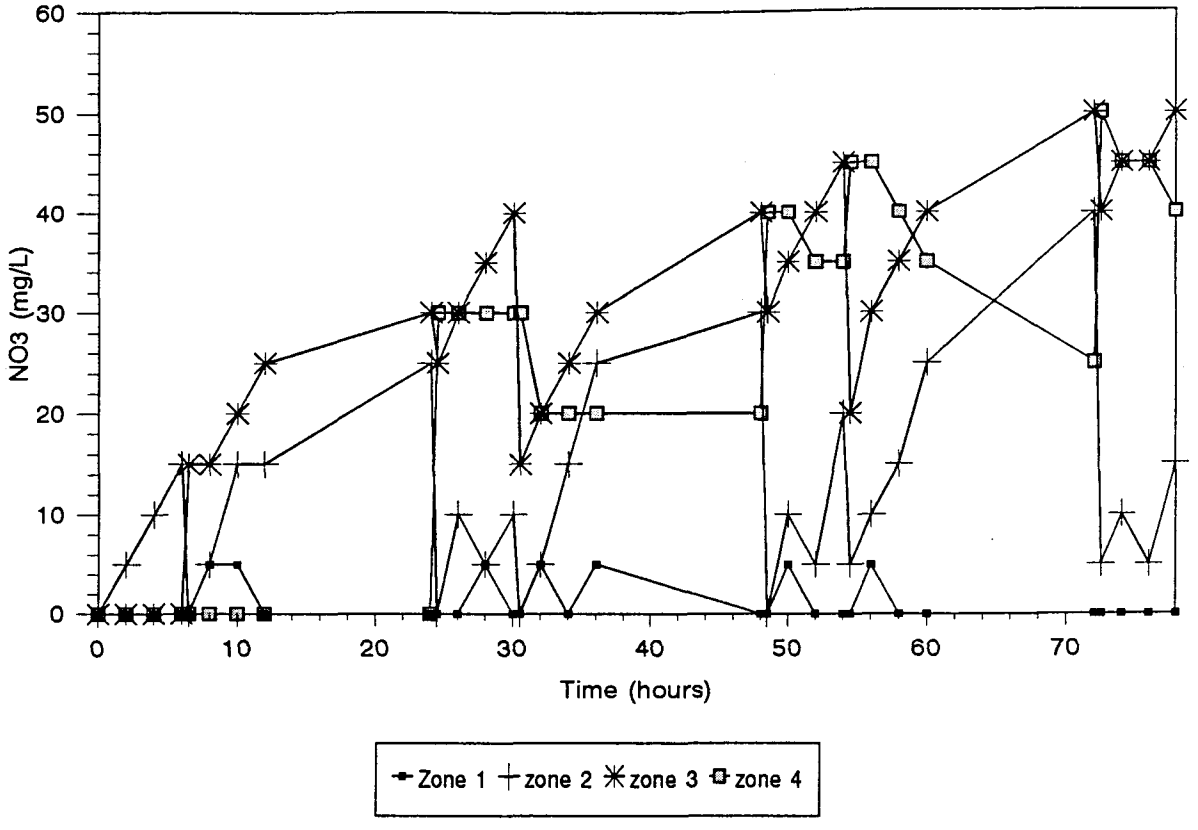


Fig 5.26. Nitrate time profiles for nominal four stage system. conditions as for Fig 5.20.

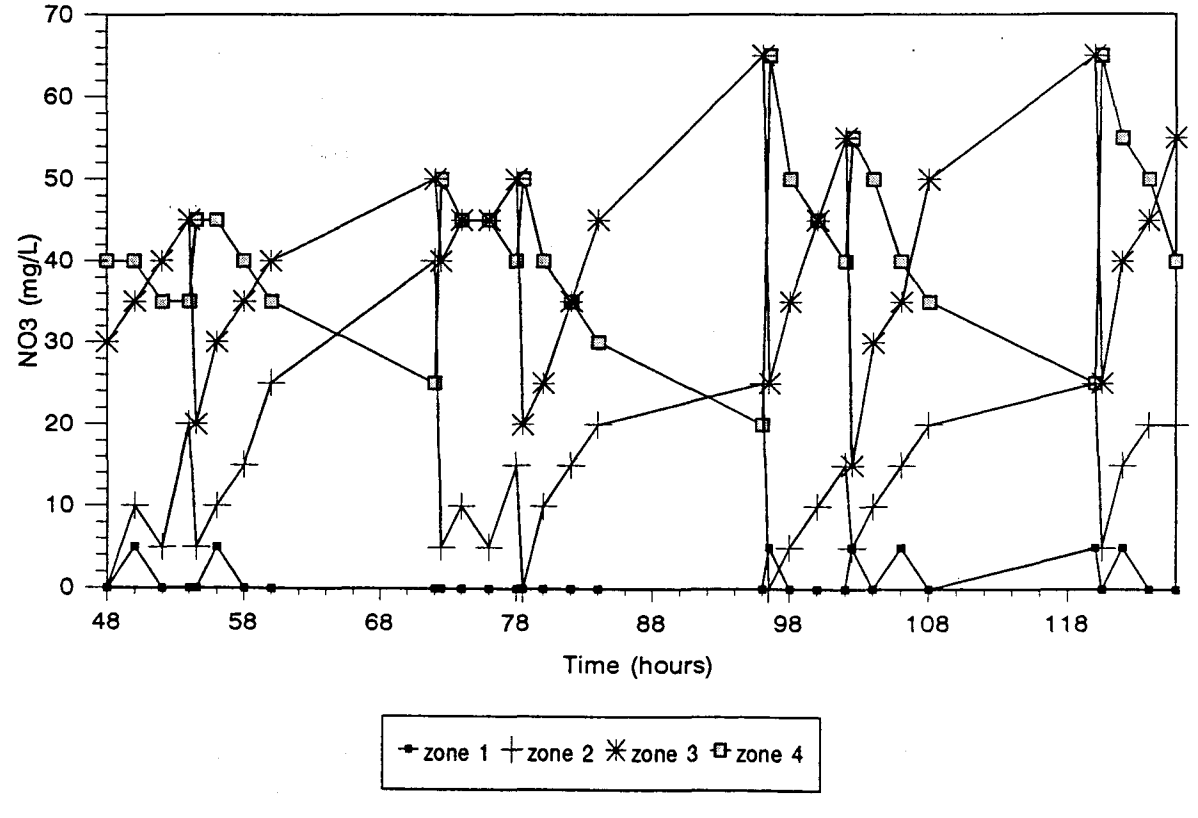


Fig 5.27. Nitrate time profiles for nominal four stage system. Conditions as for Fig 5.20. Influent concentration increased at 78h from 55mg/L ammonia to 90mg/L.

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