Full-Scale Chemical Control of Sludge Bulking

GB Saayman • J van Leeuwen • CF Schutte

Report to the Water Research Commission
by the
University of Pretoria

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SYNOPSIS

Bulking sludge is a complex biological problem affecting a large number of activated sludge plants in South Africa. Filamentous sludge bulking is caused by the proliferation of filamentous organisms. In the last decade important progress was made in bulking sludge control technologies. There are two approaches to bulking control. Specific control measures aimed at eliminating the conditions which favour the growth of filamentous organisms and non-specific control measures involving the use of chemicals to inhibit the growth of, or to selectively kill the filaments. From the evidence available on specific bulking control in nutrient removal activated sludge removal systems it seems that it is impossible to design and operate an activated sludge plant that would not bulk with low food to micro-organism (F/M) filaments. Most of the reports on non-specific control of sludge bulking in biological nutrient removal systems in South Africa deal with laboratory-scale and pilot plants. In spite of all the progress that has been made operators of full-scale plants are hesitant to employ preventive non-specific control measures, because of the potential detrimental effect of the chemicals on the nutrient removal process.

The aims of this study were:

- To investigate the effectiveness of preventative non-specific bulking control measures in a full-scale biological nutrient removal plant, and to demonstrate the feasibility of applying chlorine, ozone and hydrogen peroxide on full-scale.
- To determine the effect of hydrogen peroxide, ozone and chlorine on biological nitrogen and phosphorus removal, and on the filamentous species composition.

The final conclusions of this research are:

- Chlorination of activated sludge is the most economical non-specific method of controlling bulking sludge in a nutrient removal activated sludge plant. With the necessary precaution such as a daily trend plot of SVI values to follow the effect of chlorination and dosing at the minimum effect dose, bulking can be controlled by chlorination with only a marginal effect on phosphate removal.
- Hydrogen peroxide controls bulking during the initial stage of treatment. Regrowth of the low F/M filaments occurs even while H_2O_2 was being dosed.
- Ozone improves sludge settleability consistently and stabilises nutrient removal. However, the dose of 1,42 g $O_3.kg^{-1}$ MLSS.d^{-1} was too low to effect a dramatic improvement in sludge settleability, a prerequisite to demonstrate the success of the method.

SAMEVATTING

Slykuitdying is 'n ingewikkelde biologiese verskynsel wat 'n probleem in baie Suid-Afrikaanse geaktiveerde slykaanlegte is. Filamentiese slykuitdying is die resultaat van die florering van draadvormige of filamentagtige organismes. In die laaste dekade is daar heelwat vordering in die beheer van filamentiese uitdy-slyk gemaak. In die beheer van uitdy-slyk is daar twee benaderings. Spesifieke beheermaatreëls wat daarop gemik is om die oorsake van uitdy-slyk te identifiseer en uit te skakel teenoor nie-spesifieke metodes waarin chemikalieë gebruik word om die groei van die filamente te beheer of om die organismes selektief te dood. Uit die verslae oor spesifieke beheermetodes vir uitdy-slyk in nutriënt-verwyderings geaktiveerde slyk stelsels, wil die voorkom asof dit nie moontlik sal wees om 'n stelsel te ontwerp of te bedryf waarin lae substraat tot mikro-organismes (S/M) filamentagtige organismes nie kan floreer nie. Die meeste verslae oor nie-spesifieke beheermetodes in nutriënverwyderingsstelsels is gebaseer op werk wat op laboratorium- en loodsskaal gedoen is. Ten spyte van die vordering wat op die gebied gemaak is, is operateurs van volskaalse aanlegte huiwerig om nie-spesifieke beheermetodes in die praktyk toe te pas, weens die potensiële nadelige uitwerking daarvan op die proses.

Die oogmerk van hierdie navorsing was :

- Om voorkomende nie-spesifieke beheermetodes te ondersoek en om die toepasbaarheid daarvan in 'n volskaalse aanleg te demonstreer.
- Om die invloed van waterstofperoksied, osoon en chloor op biologiese stikstof- en fosforverwydering en op die samestelling van die filamentagtige organismepopulasie te bepaal.

Die finale gevolgtrekking van hierdie ondersoek is:

- Chlorering is die mees ekonomiese nie-spesifieke beheermetode. Met die nodige voorsorgmaatreëls soos om daagliks die tendens in die slykvolume-indeks te bepaal om die effek van chlorering op slykbesinking te volg en om chloor teen die laagste effektiewe dosis toe te dien, kan uitdy-slyk beheer word met slegs 'n marginale beïnvloeding van fosfaat-verwydering.
- Dat waterstofperoksied uitdy-slyk slegs tydelik tydens die aanvanklike stadium beheer. Lae S/M filamente het weer begin vermeerder selfs terwyl die H_2O_2 nog gedoseer is.
- Osoon het slyk-uitdying deurlopend beheer. Die dosis van 1,42 g $O_3.kg^{-1}$ MLSS.d^{-1} was egter te laag om 'n dramatiese verbetering in die besinkbaarheid van die geaktiveerde slyk, wat nodig is vir die aanvaarding van die tegniek, te weeg te bring.

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LIST OF ABBREVIATIONS.

BNR	biological nutrient removal
BOD	biological oxygen demand
C	concentration at the dosing point, mg.l^{-1}
COD	chemical oxygen demand
DO	dissolved oxygen
DSVI	diluted sludge volume index
F	frequency of exposure d^{-1}
F/M	food to micro-organism ratio
MLSS	mixed liquor suspended solids
Q	daily average influent
PHB	poly-β-hydroxybutyrate
RAS	recycle activated sludge
S	substrate concentration
SSV	settling sludge volume
SSVI	stirred specific volume index
SVI	sludge volume index
T	local mass dose at the dosing point, g.kg^{-1} MLSS
TEFL	total extended filament length
T_m	overall mass dose rate, g .kg^{-1} MLSS.d^{-1}
TSS	total suspended solids
τ	hydraulic retention time
UCT	University of Cape Town
VSS	volatile suspended solids

LIST OF FIGURES.

CHAPTER 1
INTRODUCTION AND OBJECTIVES

1.1 INTRODUCTION

The Commission of Enquiry into Water Matters (1970) (reprinted by the Department of Water Affairs, 1986) stated that an increase of 7 % per year in the demand for urban and industrial water will lead to a deficit by the turn of the century. That fact that the increase of demand has eased off to 4,4 % per annum (Department of Water Affairs, 1986) will extend this period by a few years only. Because of this increase in demand on fresh water resources, the direct and indirect reuse of wastewater will have to form an integral part of the national water economy to effect a balance between supply and demand. This demands stricter water quality criteria for water discharged into the water environment. The two major problems caused by the discharging of wastewater resulting from domestic, industrial and agricultural use of water for indirect reuse is salinisation and eutrophication of the receiving waterbody.

In the "end-of-pipe" treatment technology the activated sludge processes and in particular the nutrient removal processes are the most suited for the reclamation of wastewater. There are several reasons, or benefits, for utilising the biological nutrient removal (BNR) processes for the treatment of wastewaters. The most important environmental benefit is the control of eutrophication without causing salinisation of the receiving water. Eutrophication is the excessive enrichment of the water environment with plant nutrients, such as phosphates and nitrates, which results in the undesirable proliferation of algae and water plants. The abundant growth of algae and water plants gives rise to a number of problems, such as increased water purification cost, interference with the recreational and irrigational usage of the impoundment, aesthetic problems, loss of livestock and possibly sublethal effects on humans (Department of Water Affairs, 1986). The elimination or reduction of the addition of chemicals for phosphorus removal is an obvious direct economical benefit. The reduction in chemical addition also reduces the amount of waste sludge generated. This leads to a further economic benefit by reducing sludge disposal cost.

If an activated sludge system is properly designed and operated it can meet or exceed most effluent quality requirements (Andrews, 1992). This must be attributed to the high organic carbon as well as nutrient removal capability of the activated sludge process. The quality of the effluent from the BNR process is even higher. Wilson *et al.* (1990), cited by Randall *et. al.*, (1992) have shown that the BNR process can produce effluent lower in biological oxygen demand (BOD), total suspended solids (TSS), and phosphorus than a conventional activated sludge process using alum for phosphorus removal. To achieve this the process must meet two quite

distinct requirements:

1) Removal of organic carbon and nutrients by bacteria.
2) Separation of the bacterial flocs from the purified wastewater. This is mostly performed in settling tanks.

The first requirement presents few problems. In most cases it demands only the adaptation of the bacteria for a specific wastewater. In a review of biological nutrient removal, and in particular phosphorus removal processes, Marais (1987) concludes that the future of the processes no longer depends on a better understanding of these phenomena, but on how to deal with important ancillary problems that have been identified in operating nutrient removal plants. One of these problems is separation of the activated sludge solids from the treated effluent. Activated sludge consists of a mixture of dispersed floc-forming and filamentous bacteria. When filamentous bacteria prevail in this mixture a filamentous micro-structure is formed which reduces the sludge settleability. This phenomenon is known as bulking sludge. When the settleability of activated sludge deteriorates, incomplete separation in the settlement stage can cause high concentrations of suspended solids in the final effluent. Eventually this can result in a loss of bio-oxidation capacity, further deterioration of effluent quality, and in severe cases, a complete breakdown of the process.

Bulking sludge is a complex biological problem affecting a lot of wastewater treatment plants. In a survey (Blackbeard and Ekama, 1984 and Blackbeard *et al.* 1986) of 111 plants bulking and foaming was identified as a problem of considerable magnitude in activated sludge plants in South Africa. In another survey (Blackbeard *et al.* 1988) of nutrient removal plants in South Africa filamentous bulking was found to be a problem of considerable proportions. Of 33 plants surveyed, 27 had bulking sludge. In 1988 there were about 45 biological removal plants in South Africa in operation.

In the last decade or so, important progress was made in bulking sludge control technologies. There are two approaches to bulking control. Specific control measures aimed at eliminating the conditions which favour the growth of filamentous organisms and non-specific control measures involving the use of chemicals to inhibit the growth of, or selectively kill the filaments. Most of the reports on non-specific control of sludge bulking in biological nutrient removal systems in South Africa deal with laboratory-scale and pilot plants. In spite of all the progress that has been made, operators of full-scale plants are hesitant to employ preventive non-specific control measures, because of the potential detrimental effect of the chemicals on the nutrient removal process.

1.2 AIM AND OBJECTIVES

The aim of this study is to improve sludge settling characteristics in biological nutrient removal plants using a simple, economical technique to control bulking sludge, the main cause of poor settleability. The principal non-specific bulking control method in fully aerobic activated sludge systems is by chlorination. This procedure is well documented and is also recommended in the Manual on the Causes and Control of Activated sludge Bulking and Foaming by Jenkins *et al.* (1986). Although chlorination is also effective in controlling bulking in biological nutrient removal plants, it causes loss of nitrification and phosphorus removal and results in turbid effluent in full-scale plants (Van Leeuwen, 1992). This may be due to over dosing because operators of full-scale plants tend to chlorinate only when bulking has reached crisis proportions. Another disadvantage of chlorination is the potential risk of uncontrollable formation of chlorinated organic compounds which are discharged into the receiving waters.

Two more promising oxidants for sludge bulking control in nutrient removal systems are hydrogen peroxide and ozone. Hydrogen peroxide was used in full-scale fully aerobic activated sludge plants for bulking control (Jenkins, *et al.*, 1986) but not in biological nutrient removal activated sludge systems. In bench scale experiments on a biological nutrient activated sludge system hydrogen peroxide controlled bulking without detrimentally influencing nutrient removal (Van Leeuwen, 1992). In pilot scale investigations ozone was effective in combating filamentous and non-filamentous sludge bulking(Van Leeuwen and Pretorius, 1988; Van Leeuwen, 1988b; Van Leeuwen, 1989). Improved nutrient removal was observed (Van Leeuwen 1988a).

At this stage, there is a need to investigate these non-specific sludge bulking control methods in a full-scale plant in order to prove that these technologies are successful and for technology transfer to consultants and operators of full-scale plants. Consultants will not easily recommend the application of capital intensive technologies, such as ozone, which have not been proven in full-scale plants. There is also a need to demonstrate that these technologies can be applied in full-scale plants as preventative bulking control measures and not as shock treatments only.

The biological nutrient removal plant at the Daspoort Works of the City Council of Pretoria consists of three parallel, identical units with a common feed making it ideally suitable for the comparison of sludge bulking control chemicals. One of the units can serve as a control while the chemicals, hydrogen peroxide, ozone or chlorine are dosed in the other two.

The objectives of this study are:

1) To investigate preventative non-specific bulking control measures in a full-scale biological nutrient removal plant, and to demonstrate the feasibility of the techniques on full-scale.
2) To determine the effect of hydrogen peroxide, ozone and chlorine on biological nitrogen and phosphorus removal, and on the filamentous species composition.

CHAPTER 2
LITERATURE REVIEW

2.1 THE ROLE OF FILAMENTOUS ORGANISMS IN ACTIVATED SLUDGE

Many activated sludge solids separation problems can be related to the nature of the activated sludge floc. In a typical activated sludge there is a wide range of particle sizes. Parker *et al.* (1971) suggests that the smaller sized particles are individual micro-organisms or small aggregates of micro-organisms that either have not flocculated or have been sheared off from larger flocs. The larger sized particles are flocs of which the peak size is controlled by their strength and the turbulence level of the environment in which they exist.

On the basis of visual observation and some physical measurements it has been suggested that there are two levels of structure in the activated sludge floc. These have been termed the microstructure and the macrostructure (Sezgin *et al.*, 1978). The microstructure is imparted by the process of microbial aggregation and bioflocculation. This process is the basis for floc formation in activated sludge. Without the ability to stick to one another, the larger aggregates that exist in activated sludge would never form. The macrostructure of activated sludge flocs is provided by filamentous micro-organisms. These organisms form a network within the floc on to which the floc-forming micro-organisms adhere. When activated sludge contains filamentous micro-organisms, larger flocs are formed, because the presence of the filamentous organism backbone strengthens the floc so that it can survive the turbulent environment of the aeration basin.

In the absence of filamentous micro-organisms the flocs will only have microstucture. These type of flocs are small, spherical, compact and relatively weak. They can be easily sheared in the turbulent environment in the aeration basin of an activated sludge plant. The resultant smaller flocs or aggregates settle slowly and create a turbid supernatant. The presence of a filamentous backbone is essential for proper floc formation, however excessive filamentous growth causes sludge bulking. When filaments are present in large numbers they form weblike structures extending into the bulk liquid, leading to large diffuse flocs and bridging between flocs. The filamentous organisms extending from the flocs into the bulk of the liquid interfere with the settling and the compaction of the settled sludge on the floor of the settling tank. Interference with settling causes excessive solids carry over in the overflow of the settling tank and the lack of compaction may lead to denitrification and sludge flotation and difficulties in achieving the return activated sludge concentration and maintenance of the required recycle ratio's (Jenkins *et al.*, 1986). For the proper operation of the activated

sludge system a balance between floc-formers and filaments should be maintained. It is therefore clear that the aim of all bulking sludge control strategies must be the reduction of the number of filamentous organisms and not the complete elimination of the filaments.

The effect of filamentous organisms on floc structure is represented in Figure 1.

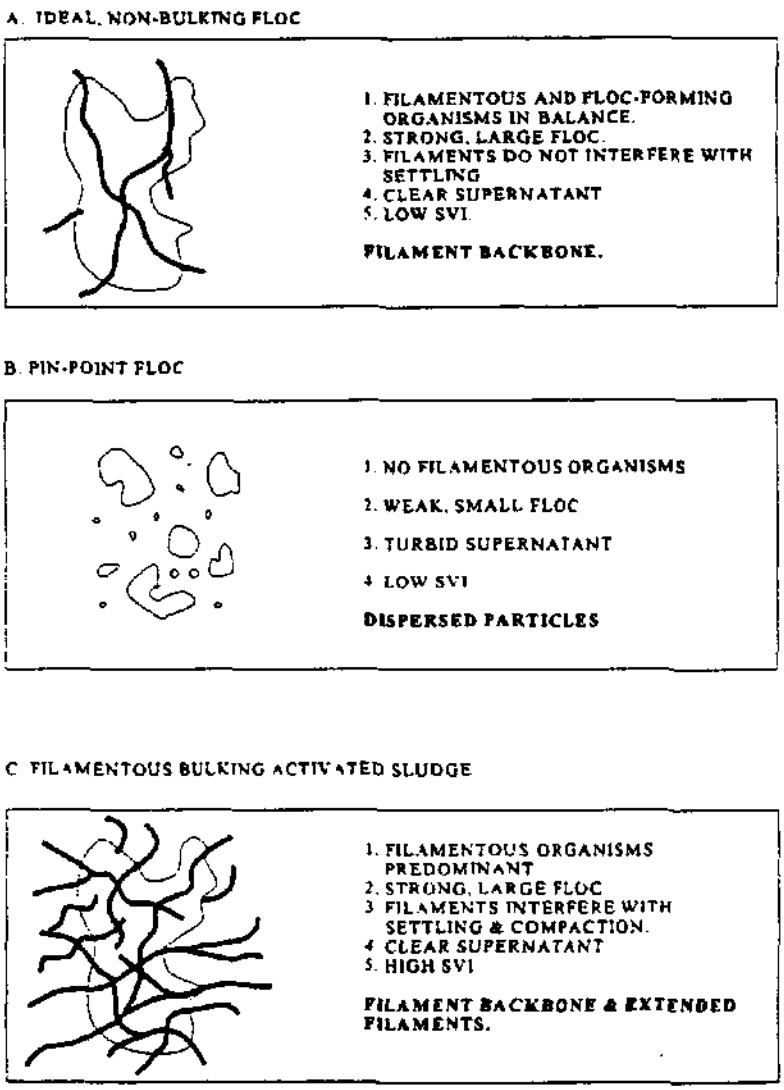

Figure 1 The effect of filamentous organisms on the activated sludge floc structure. (Jenkins, *et al.*, 1986).

2.2 ACTIVATED SLUDGE SEPARATION PROBLEMS

Activated sludge with the desired settling properties are of preeminent importance to the success of the activated sludge system. Sludge with the desired character,

- settles fast, usually with velocities in excess of 1 $m.h^{-1}$,
- does not occupy an excessive volume after settling,
- after settling leaves a clear supernatant, and
- does not rise within a reasonable period, usually 2-3 hours, after settling.

In wastewater treatment five major problems related to biomass quality lead to deterioration in treatment effectiveness of activated sludge plants.

2.2.1 Dispersed Growth

Dispersed growth is caused by a micro-structure failure (Jenkins *et al.* 1986) in which, under certain circumstances, bacteria of activated sludge do not aggregate into flocs and grow as individual cells or small clusters. There is no zone sedimentation in the secondary settling tanks. The individual cells and small clusters are carried over causing high turbidity in the effluent.

2.2.2 Pinpoint flocs

Pinpoint floc formation is due to a macrostructure failure (Jenkins *et al.* 1986). Small, compact, weak, roughly spherical flocs are formed. A considerable range of floc sizes can be observed. The larger flocs settle very quickly. The smaller flocs, with settling velocities almost zero, remain in suspension. These flocs cause the turbidity in the final effluent and poor control over sludge age. Pinpoint flocs are, *inter alia*, the products of the disintegration of initially firm flocs due to:

- Insufficient production of glycocalyx, the extracellular material that contributes to bioflocculation, or its consumption by bacteria inside the flocs as a result of a low organic loading (systems with high sludge age) (Eckenfelder and Grau, 1992),
- Absolute absence of filamentous micro-organisms necessary for the so-called filamentous backbone of the flocs (Jenkins *et al.* 1986).
- The disintegration of flocs by shearing effects, for instance inappropriate mechanical aerators (Konicek and Burdych 1988).

2.2.3 Rising sludge

Certain types of "bulking sludge" are caused by the formation of microscopic bubbles of nitrogen gas, the result of endogenous denitrification in the secondary clarifier, causing flotation of sludge solids. Others types come from rising sludge blankets. Endogenous denitrification does occur in the secondary settling tank if the effluent from the aeration basin contains high nitrate concentrations and denitrifying bacteria are present. The latter may result from exceeding the sludge transport capacity of the clarifier scrapers, inadequate recycle activated sludge (RAS) rate, excessive floor loading, or various combinations of these factors. These observations inflate the reported level of bulking sludge problems (Randall *et al.* 1992). Sludge rises if the density of the flocs is lower than that of water. There are two reason for low density of the floc:

- The adsorption of oils and fats
- The occlusion of small gas bubbles.

2.2.4 Zoogloeal Bulking

Zoogloeal or non-filamentous bulking is caused by a microstructure failure in which too much of the extracellular material that contributes to bioflocculation is produced (Jenkins *et al.* 1986). In this case a viscous sludge with a high sludge volume index (SVI) is formed. Zoogloeal organisms may also proliferate in activated sludge to such an extent that they may cause sedimentation problems. This type of bulking is uncommon and the only occurrence in South Africa was reported in fuel synthesis wastewater treatment (Van Leeuwen, 1989). This type of bulking has been referred to as hydrous or viscous bulking due to the slime or jelly-like characteristics of the sludge solids.

2.2.5 Filamentous Bulking and Foaming

In filamentous bulking and foaming filamentous organisms that provide the macro-structure in the activated sludge floc are present in excess. They interfere with the settling and compaction of the activated sludge either by producing a very diffuse floc structure or by growing in profusion beyond the confines of the floc into the bulk medium causing bridging between flocs. Filamentous micro-organisms, however, belong to the natural components of activated sludge biocenosis. If the specific environment provides improved opportunity for filamentous organism growth versus that of the preferred organisms, they will proliferate. Filamentous organisms need not be the dominant mass in sludge to cause bulking. A change of 5 - 10 % of the sludge

mass from non-filamentous to filamentous species can produce a bulking condition (Randall, *et al.* 1992)

In the presence of adequate nutrients, the cause of bulking is generally linked to the design of the activated sludge reactor and the environment maintained within the reactor. Specific environmental factors have been linked to the presence of specific filamentous organisms. Jenkins *et al.* (1986) identified five conditions that lead to filamentous organism proliferation. They are low dissolved oxygen levels (DO), low food to micro-organism ratio (F/M), nutrient deficiency, septic influent and low pH.

2.3 THE IDENTIFICATION OF THE CAUSES OF ACTIVATED SLUDGE SEPARATION PROBLEMS

In the rational diagnosis of activated sludge solids separation problems a microscopic examination and filamentous organism characterization is necessary. The conventional method of identification of bacteria is based on their isolation and subsequent exhaustive tests of their morphological, biochemical and physiological features. The results of these tests are then compared with standard references. For the purpose of isolating the filamentous micro-organisms from the diverse biocenosis of activated sludge, the conventional methods are too cumbersome, time consuming and mostly unnecessary or even inappropriate.

The identification and classification of types by Eikelboom and van Buijsen (1981) represented a great breakthrough in the classification of activated sludge filamentous bulking and foaming organisms. The techniques are based on phase contrast microscopic observations of morphology, the relationship to other organisms and staining characteristics. The filamentous organisms are classified according to cell shape dimensions, presence of sheath, morphology of the filaments, staining characteristics and the presence or absence of polyphosphate, sulphur and poly-β-hydroxybutyrate (PHB) granules. The identification of types according to Eikelboom and van Buijsen (1981) with modifications by Jenkins *et al.* (1986) is fast and can be performed by trained non-microbiologists.

Specific environmental conditions have been linked to the presence of specific filamentous organisms. Table 1 (Jenkins *et al.* (1986)) gives a listing of causative factors and the dominant filamentous organism found.

Table 1. Dominant filament types as indicators of conditions causing activated sludge bulking.

Suggested causative factor	Indicative filament types.
Low DO	type 1701, *Sphaerotilus natans, Haliscomeno-bacter hydrossis*
Low F/M	*Microthrix parvicella, H. hydrossis, Nocardia* sp. types 021N, 0041, 0675, 0092, 0581, 0961, 0803
Septic wastewater/sulphide	*Thiothrix* sp., *Beggiatoa* and type 021N
Nutrient deficiency	*Thiothrix* sp., *S. natans* , type 021N and possibly *H. hydrossis* and types 0041 and 0675
Low pH	Fungi

Source Jenkins *et al* 1986.

The five most frequently dominant filamentous organisms in the mixed liquor of nutrient removal plants in South Africa were (i) Type 0092 (82 %). (ii) Type 0675 (45 %), (iii) Type 0041 (39 %), (iv) *Microthrix parvicella* (33 %) and (v) Type 0914 (33 %) (Blackbeard, *et al.*, 1988). These five organisms were also the most frequently dominant ones in foams on 18 of the plants . Except for type 0941, these filaments are classified by Jenkins *et al.*. (1986) as follows:

- Type 0092 and *M.parvicella* low F/M ratio (long sludge age):
- Types 0675 and 0041 low F/M and nutrient deficiency.

The consistent dominance of Type 0914 in South Africa's long sludge age plants suggests that it possibly should be classified as a low F/M filamentous organism (Blackbeard, *et al.* 1988).

2.4 DEFINITIONS AND MEASUREMENTS

The term "bulking" lacks scientific clarity. Lacking a comparative measure of the settling characteristics of activated sludge early practitioners and researchers referred to, what is now recognized as bulking sludge, as the instability of the process or the unsettleability of the sludge. In order to quantify bulking and make comparison between plants possible a number of parameters were defined. The more useful ones are the following.

2.4.1 Sludge Volume Index

The volume occupied by sludge after settling, is mainly determined by the interorganism spaces. The SVI of activated sludge is defined as the volume occupied by one gram of sludge after settling. SVI is internationally used as a measure of sludge settleability. Activated sludge is voluminous, 1 gram of a nonbulking sludge can easily occupy 100 m*l* after quiescent settling. Jenkins *et al* (1986) rated an activated sludge with a SVI > 150 m*l*.g^{-1} as a bulking sludge. To an operator who is used to coping with a sludge with a SVI of 300-400 m*l*.g^{-1}, a sludge with a SVI of 150 m*l*.g^{-1} will be regarded as a non-bulking sludge. However, plant operators who are used to maintain stable operations at SVI's of 60 - 80 m*l*.g^{-1} will become concerned when the SVI exceeds 100 m*l*.g^{-1}. The SVI of bulking sludges easily reaches values of 200 to 300 m*l*.g^{-1} or even higher. Settlers operating at an upflow rate of 1 m.h^{-1}, a design parameter for secondary settling tanks recommended by the Water Institute of Southern Africa (1985), can hardly cope with sludge with an SVI of 150 m*l*.g^{-1}. This leads to spilling over of sludge from the settler and consequently a poor quality effluent and uncontrollable sludge age.

The term SVI has many different meanings in the international community since data have been reported as SVI_{30} (and SVI_{60}), $SSVI_{30}$, $SSVI_{2.0}$, $SSVI_{3.5}$, $DSVI_{30}$ and perhaps other definitions: A brief description (Randall *et al.* 1992) of each of these tests follows:

2.4.1.1 SVI_{30}

The activated sludge in the mixed liquor is allowed to settle, without agitation, in a one litre graduated measuring cylinder or a two litre settlometer for thirty minutes. The settling sludge volume (SSV) is recorded as millilitres per litre. The recorded 30-minute settling volume (SSV_{30}) is converted to SVI as a function of the mixed liquor suspended solids (MLSS).

$$SVI = \frac{SSV}{MLSS} \times 1000$$

When the SSV_{30} exceeds 250-300 m*l*.*l*$^{-1}$, the relative value of this test is limited because bridge formation between the sludge flocs prevents the sludge from settling. According to Randall *et al.* (1992) values above 500 m*l*.*l*$^{-1}$ may be misleading and of little value. Sometimes the sludge in this test, and the following tests, is allowed to settle for sixty minutes, and reported as SVI_{60}.

2.4.1.2 Stirred specific volume index ($SSVI_{30}$)

The settling test is conducted in a settlometer equipped with a stirrer. The stirrer operating at about

1 rpm is used to disrupt floc particle bridging and assist the consolidation of the sludge. This test will generally produce a smaller 30-minute sludge volume than the unstirred test. It is considered to be more representative of full scale sludge compaction, but not of solids-liquid separation rates.

2.4.1.3 $SVI_{2.0}$ and $SSVI_{2.0}$

The settling in these tests is conducted at a MLSS concentration of 2 000 mg.l^{-1} (2,0 g.l^{-1}) and represents an effort to standardize the procedure for comparison. The $SVI_{2.0}$ has the same limitations as the SVI_{30} test. The $SSVI_{2.0}$ test is also an attempt to eliminate the effects of varying MLSS concentrations on both the settling rate and solids-liquid separation rate. Because of standard conditions maintained in this test, it has an advantage over the $SSVI_{30}$ test for plant-to-plant comparisons.

2.4.1.4 $SSVI_{3.5}$

This test is similar to the $SSVI_{2.0}$ procedure. It is conducted at a MLSS concentration of 3 500 mg.l^{-1}. It is, however, more directed to the final concentration characteristics of the sludge than the solids-liquid separation rates at the interface. The $SSVI_{3.5}$ values will increase at a higher rate than the $SSVI_{2.0}$ as the sludge volume index of the sludge increases.

2.4.1.5 Diluted sludge volume index $DSVI_{30}$

In this procedure the mixed liquor is diluted until the resulting 30-minute settled volume is 200 m$l.l^{-1}$ or less. A tolerance of 40 m$l.l^{-1}$ does not have a serious effect on the resulting DSVI. Due to minimization of bridging between flocs and MLSS concentration effects, this procedure will provide a more representative value of the solids-liquid separation rate and a better projection of the potential concentration characteristics of the settled solids (Randall, *et al.*, 1992).

The relationships between these various test are not constant, and plant-to-plant comparisons are difficult and sometimes impossible at high SVI's.

2.4.2 Filament Levels in Activated Sludge

The filamentous organism level in activated sludge is extremely important in determining the settling and compaction characteristics of the sludge. Finstein and Heukelekian (1967) determined a relationship between filamentous organism levels and sludge settling properties. They showed that the SVI of activated sludge could be related to the total filament length.

Pipes (1979) counted the filamentous organisms extending from the floc and found that at low filament numbers (10^2 filaments/mg volatile suspended solids (VSS)) the SVI was below 100 $ml.g^{-1}$, At high filament counts (>10^2 - 10^3 filaments/mg VSS) the SVI increased markedly.

Sezgin *et al.* (1978) developed a method for measuring the total extended filament length (TEFL). Using this method Sezgin *et al.* (1978), Palm *et al.* (1980) and Lee *et al.* (1982) found correlations between various measures of activated sludge settling, such as SVI and zone settling velocity , and TEFL for activated sludge grown in the laboratory on domestic sewage. Sezgin (1980) found that these relationships were valid for activated sludge taken from several full-scale plants.

Lee *et al.* (1983) investigated the relationship between TEFL and the various settleability tests for the control of activated sludge plants. Their data showed that the most consistent correlation was obtained between TEFL and DSVI. On the strength of this conclusion they proposed the replacement of the standard SVI test with the DSVI test. Further justification for this proposal was provided by Koopman and Cadee (1983), Rachwal *et al.* (1982) and Pitman (1984) who showed that the parameters relating activated sludge settling velocity and MLSS concentration were well correlated with the DSVI. Thus the DSVI test, a simple test that can be done on a routine basis, can be used as a indication of the level of filamentous organisms present in activated sludge.

2.5 CONTROL OF BULKING AND FOAMING FILAMENTOUS MICRO-ORGANISMS

Since the evolution of the activated sludge reactor design from fill-and-draw or batch reactors to continuous flow design, sludge bulking has plagued designers and operators. Due to a lack of a comparative measure of sludge settling characteristics early workers referred instead to the instability of the process or the unsettleability of the sludge. This is now recognized as bulking sludge caused by an excessive growth of filamentous micro-organisms.

In spite of much progress, bulking sludges still cause serious problems in wastewater treatment plants. However, technology has slowly progressed to control organisms responsible for bulking sludge. The control theory and practice of controlling an excessive growth of filamentous micro-organisms can be divided into two groups. The measures can be either directed towards creating environmental conditions in the plant such that the growth of the filamentous organisms is inhibited or suppressed (specific control), or involve the use of oxidants which selectively control the excessive growth of the organism(s) causing the bulking or the use

of flocculants that increases the settleability of the sludge (non-specific control). Specific growth control methods are of a preventative nature and should always be considered in the design of plants. On the other hand, non-specific methods are remedial and cannot guarantee a long term effect.

2.5.1 Specific bulking control

Specific control of bulking focuses on identifying and eliminating the conditions that promote the proliferation of the specific filaments causing the bulking problem. Through the identification of the types of filaments present in the sludge the conditions promoting the growth of the filaments can be identified. Once these conditions are identified, it may be possible to create environmental conditions in the activated sludge reactor which should inhibit or suppress the growth of filamentous organisms.

The conditions which lead to the proliferation of filamentous organisms in activated sludge have been the subject of a large amount of work. A number of conditions may singly or in combination alter the microbial population in a reactor from one dominated by floc forming organisms to one dominated by filamentous organisms. Growth environments blamed for filamentous bulking, summarised by Chiesa and Irvine (1985), include the following:

- Low aeration basin DO concentration.
- High aeration basin DO concentration.
- Low organic loading rates.
- High organic loading rates.
- Completely mixed reactor configurations.
- Conventional plug flow reactor configurations.
- Inadequate micronutrient concentrations.
- Elevated metal concentrations.
- Low pH.
- Elevated sulphide concentrations (septic influent)
- Low relative influent nitrogen and phosphorus content (high chemical oxygen demand (COD):P and COD:N ratio's at high organic loadings).
- Over-grazing by protozoa.

Many of the reasons reported to cause bulking appear to be contradictory.Chiesa and Irvine (1985) therefore, proposed a more fundamental approach to the problem. According to them an explanation for the contradictions can be based on the fact that due to the wide range of operating conditions that can exist at

activated plants the growth of different filamentous organisms can be favoured in different plants. Filamentous organisms can therefore easily proliferate on the substrate available at almost any plant if the operating conditions favour the filaments rather than the floc formers. From this Chiesa and Irvine (1985) came to the conclusion that bulking can be expected at the same treatment plant at different times throughout the year unless a strong selective pressure that favours floc formers over filaments is imposed.

Chiesa and Irvine (1985) summarized (Table 2) the key physiological characteristics of the major groups of microbes from literature exploring the ecology of activated sludge systems. The non-filamentous bacteria include micro-organisms such as *Pseudomonas* and *Arthrobacter*; the high organic filaments, *Sphaerotilus natans*, type 1701 and type 021N; and the low organic filaments, *Haliscomenobacter hydrossis*, *Microthrix parvicella* and *Nocardia*. They point out, however, that the characteristics of individual species or strains may vary from the general characteristics noted. The filament groups in Table 2 are responsible for bulking either due to low organic loading conditions or oxygen conditions.

Table 2. Relative physiological characteristics of activated sludge bacteria.

Traits	Commonly encountered nonfilamentous bacteria.	Filaments associated with high organic loading rates	Filaments associated with low organic loading rates
Substrate affinity	Low	High	High
Maximum growth rate	High	High	Low
Endogenous metabolic rate	Low	High	Low
Resistance to starvation	High	Low	Very high
Potential to accumulate classical storage compounds	High	High	Varies
Maximum rate of substrate uptake/storage product formation.	Very high	High	Low
Decrease in metabolic activity at low dissolved oxygen tension.	High	Slight	-

Source Chiesa and Irvine (1985)

In the conventional kinetic hypotheses filamentous micro-organisms have been treated as a uniform group of organisms with the same kinetic and metabolic properties. The activated sludge process is idealized by Chiesa and Irvine (1985) as consisting of three classes of model organisms. The organisms include a fast growing non-filamentous floc forming bacterium, a slow growing, starvation resistant filament exhibiting high substrate affinity (a low Michaelis-Menten constant K_m) and a fast growing starvation susceptible filament with a high affinity for oxygen. The comparative growth rate versus substrate concentration, incorporating the physiological traits detailed in Table 2 are presented in Figure 2. If a steady state substrate

concentration (S) in a continuously fed system is maintained above the critical value S*, indicated in Figure 2, selection for a non-filamentous population would be expected. These critical values are more characteristic for conventional activated sludge loadings.

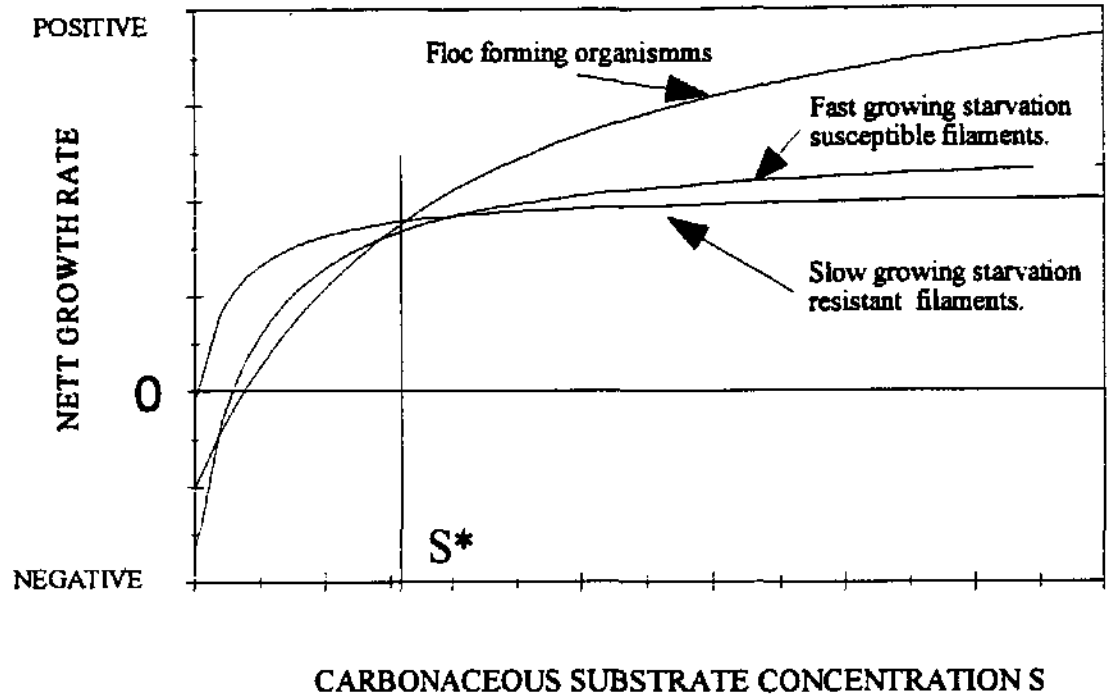

Figure 2 Microbial growth and competition including starvation induced death.
Source Chiesa and Irvine (1985).

The growth relationships depicted in Figure 2 assume that substrate concentration is the only growth limiting factor. Figure 3 includes the effects of potential oxygen deficiencies. In a reactor with high substrate (above S**) but low dissolved oxygen levels, fast growing filaments with a high affinity for oxygen would be expected to proliferate. This would affect the sludge settling adversely. An increase in oxygen concentration in a system bulking under these conditions would most likely reduce the problem and prevent its reoccurrence unless the loading to the system increases.

The dominant class of micro-organism selected in continuously fed activated systems will be a function of the reactor substrate concentration or organic loading. To date, in most of the specific bulking control methods, reactor configuration and operating conditions are employed that result in differential specific growth rates caused by substrate gradients. Low substrate concentration conditions could in part of the processes be prevented by avoiding completely mixed conditions in which the organic substrate concentration in the reactor equals that of the effluent. Chuboda, *et al.*, (1973) (as cited by Van Leeuwen, 1990) were the first to establish the selector principle, the maintenance of a high F/M ratio at the beginning of the process

allowing the floc forming organisms to utilise the major portions of the available substrate rapidly and outgrow the filamentous organisms. The remaining, but smaller, food fraction would still encourage filament growth during further processing. Increasing the F/M ratio during part of the retention time in the process could be achieved by:

1) compartmentalisation of the aeration reactor to approach a plug flow pattern,
2) installation of small aerated reactors ahead of the main aeration reactor in which the recycled sludge and incoming wastewater are mixed, and
3) semi-batch operation where intermittent high feed loads create temporary high F/M ratios.

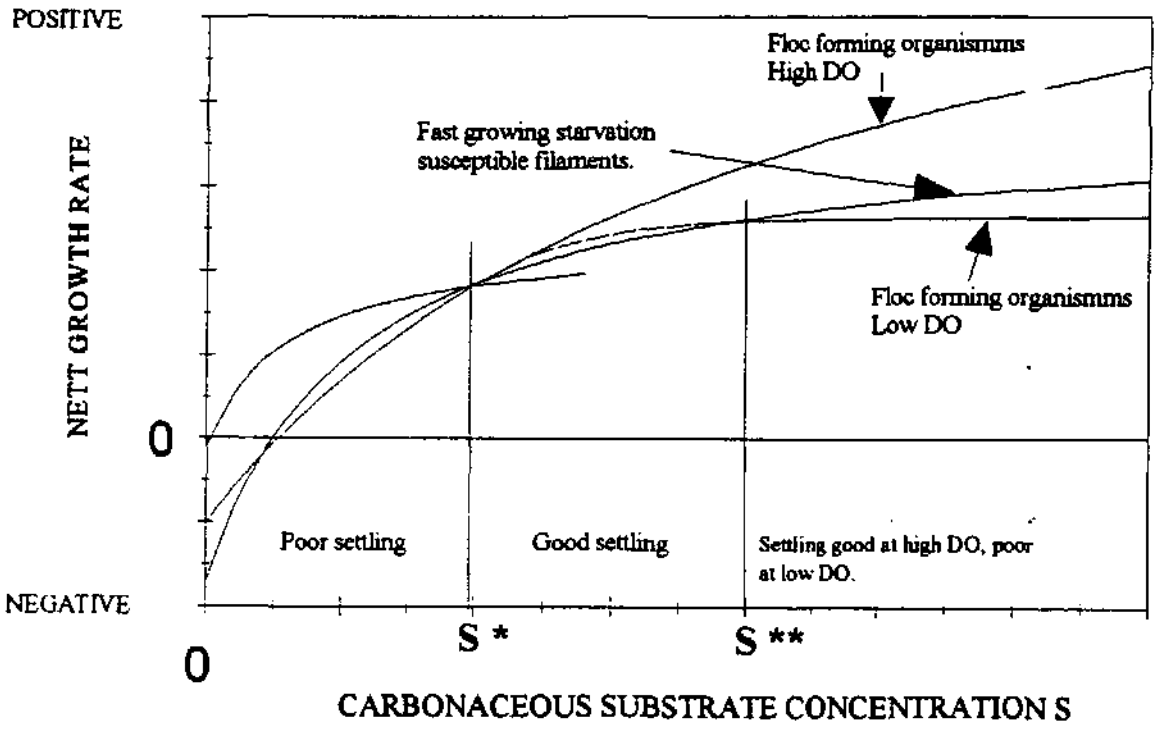

Figure 3 Microbial competition indicating potential dissolved oxygen limitations.
Source Chiesa and Irvine (1985)

Gabb, *et al.*, (1988) proved that the forementioned measures were effective against those filaments which are characteristic of purely aerobic low F/M plants only. Filaments of the low F/M group associated with bulking in long sludge age BNR activated sludge systems do not proliferate under purely aerobic conditions irrespective of whether or not a selector is present. The presence of unaerated zones or intermittent aeration, however, promotes their proliferation. Under these conditions the installation of an aerobic selector does not ameliorate the bulking (Gabb, *et al.*, 1991). This finding placed the research on specific sludge bulking control in biological removal activated sludge systems back into an exploratory phase.

From operating laboratory N and N & P removal systems under different conditions Casey, *et al.*, (1993) concluded that a major factor influencing low F/M bulking was the continuous alternation between anoxic and aerobic conditions in the system. This gives the low F/M filaments a competitive advantage over the floc-formers. From an examination of their experimental data and survey of denitrification pathways in the microbiological and biochemical literature, it was hypothesized that this advantage arises because the filaments reduce nitrate to nitrite whereas floc-formers reduce nitrate to nitrogen gas (with possible intermediates nitrite, nitric oxide (NO), and nitrous oxide N_2O)) and in the process accumulate NO which exerts an inhibitory effect on their aerobic oxygen utilization rate in the subsequent aerobic conditions.

If this hypothesis that intermediates of the denitrification process favour the growth of filamentous organisms is true, it will not be possible to design and operate a nitrogen or nitrogen and phosphorus plant that does not bulk with low F/M filaments, because denitrification takes place in the aeration zones too. Mass balances on nitrogen in the Baviaanspoort and Rooiwal works at Pretoria have shown that denitrification takes place in the aerobic zone of the reactor too (unpublished report). This has also been observed in Australia. (P. Griffiths and Sinclair Knight Merz, personal communication). The Baviaanspoort works is equipped with surface acrators and the Rooiwal plant with a fine bubble aeration system. Furthermore Randall *et al.* (1992) also reported that some degree of denitrification takes place in the aeration zones of biological nutrient removal plants, varying from 20 % of the total nitrogen in the influent to all the nitrogen not captured in the sludge. The explanation for the observed denitrification may be that at the DO levels at which BNR plants operate, anaerobic and anoxic conditions may exist inside large flocs. The build up of NO in the aerobic zone may occur even if all the nitrate returned to the anoxic zone is fully denitrified before the sludge goes back to the aerobic zone. It is, therefore, clear that there will always be a need for non-specific bulking control methods.

2.5.2 Non-specific bulking control

With non-specific bulking control methods the problem caused by the presence of filamentous micro-organisms is treated. In these methods the settleability of the activated sludge is improved by chemical additions or by the selective killing of the filamentous organisms with an oxidant such as chlorine, ozone or hydrogen peroxide. They have a short-term impact and should be applied repeatedly.

2.5.2.1 Bulking control without Damaging the Filamentous Micro-organisms

In an emergency case when activated sludge escapes from the secondary settling tanks the settleability of the

sludge can be improved by weighting of the activated sludge flocs or by the addition of synthetic polymers.

2.5.2.1.1 Weighting of the Sludge

The settling velocities of bulking sludge can be increased by increasing the density of the flocs. It is a known fact that activated sludge treating raw sewage suffers less frequently from bulking. One of the reasons is the addition of better settleable solids to the flocs. These settleable solids are enmeshed in activated sludge flocs and improve settling. In some instances inorganic coagulants such as lime and ferrous or ferric salts are added into the aeration basin. The heavier precipitates sweep down the bulking sludge flocs and improve the settling.

The inhibition of biological phosphate removal by the addition of ferrous sulphate into a nutrient removal activated sludge system was investigated by Boyd (1993). Boyd came to the conclusion that the formation of iron hydroxide 'uses' the hydroxyl ions that are closely associated with the bacteria found in the flocs, thus interfering with the hydroxyl mediated transport process. According to Wentzel *et al.* (1986) the translocation of acetate, phosphate and cations across the cytoplasmic membrane under anaerobic conditions is a hydroxyl mediated process. Based on the selection principle it is also reasonable to accept that to establish a population of phosphorus accumulating bacteria a phosphate rich environment is a prerequisite. The addition of phosphate removing chemicals such as iron and aluminium salts will in all probability reduce the specific growth rates of these organisms. Over the long term this will lead to the washout of these organisms from the system, at the normal sludge retention times maintained in a biological nutrient removal plant. The use of ferrous, ferric and aluminium salts is, therefore not recommended for sludge bulking control in nutrient removal systems.

It should however be noted that iron is an important nutrient. The iron determined analytically in wastewater may not be easily available to micro-organisms due to the formation of various precipitates. In some cases small doses of iron improve settling. However, in such instances weighting of the flocs is not the most probable mechanism, but rather the elimination of a growth-limiting condition that favours the floc-forming micro-organisms.

2.5.2.1.2 Polymer Addition

Synthetic polymers have been used to improve the settleability of the activated sludge (Jenkins *et al.*, 1986). The dose and type of polymer (cationic alone or in combination with anionic) should be determined in jar test.

The dosing point should be determined by trial and error keeping in mind the requirements for optimum flocculation (rapid mixing followed by mild agitation for flocculation). Dosing into the centre well or flocculation zone of the secondary settling tank is often the best dosing point. Polymer addition generally improves clarifier efficiency at lower temperatures when the density difference between the water and the flocs is smaller and the viscosity of the water higher.

According to Jenkins *et al.* (1986) it is wise to frequently estimate the polymer dose in jar tests, because the optimum dose can change and overdosing can lead to deterioration in performance.

2.5.2.2 Bulking control by Inhibiting the Filamentous Micro-organisms with Oxidants

The filaments protruding from the flocs into the bulk of the solution can be selectively damaged or their growth can be inhibited by strong oxidants. Because the filamentous organisms extend beyond the flocs into the liquid, they are more exposed to the oxidant and are selectively killed. In contrast the floc formers are not seriously affected because they are protected inside the sludge flocs. Due to their larger surface to mass ratio it is also possible that the filamentous organisms may be more susceptible to the oxidative action of oxidants than the typical floc-formers . Due to the selective killing of the filaments, their number and lengths are reduced and bulking is ameliorated. The oxidant affects all filaments irrespective of type and for this reason this method of bulking control is called non-specific.

2.5.2.2.1 Oxidants and their reactions

2.5.2.2.1.1 Hydrogen Peroxide

In nature, hydrogen peroxide, H_2O_2, occurs only in traces in the free state, since it decomposes into oxygen and water. Because H_2O_2 is unstable, owing to the reaction

$$2\,H_2O_2 \rightarrow 2\,H_2O + O_2\,(g)$$

it is difficult to store. The decomposition is slow but is catalyzed by impurities such as dust and dissolved compounds. It is also accelerated in the presence of light. For these reasons, solutions of H_2O_2 are stored in vented dark containers with various chemicals added which destroy catalysts.

Hydrogen peroxide is often manufactured by methods which involve the intermediate preparation of

perdisulphuric acid, $H_2S_2O_8$, or its salts. Thus, in one process, 50 % sulphuric acid is electrolysed under high current density with platinum anodes and graphite rods for the cathodes. Hydrogen gas is formed at the cathode and perdisulphuric acid at the anode. The resulting solution of perdisulphuric acid, and unchanged sulphuric acid, is distilled under reduced pressure, when the following hydrolysis takes place

$$H_2S_2O_8 + 2H_2O \rightarrow 2H_2SO_4 + H_2O_2$$

and an aqueous solution of hydrogen peroxide distils over. This contains up to 40 % of hydrogen peroxide and may be concentrated up to 90 % by fractional distillation in vacuo. Pure hydrogen peroxide can be obtained by fractional distillation of aqueous hydrogen peroxide under reduced pressure. The pure compound is very much like water in many of its physical properties. Anhydrous hydrogen peroxide is a colourless liquid of density 1 460 $kg.m^{-3}$, but it is blue in bulk, like water. Hydrogen peroxide is commercially available usually as a 50 % solution. This solution is stabilised and has a relatively long shelve life.

Hydrogen peroxide has been used for bulking control in conventional activated sludge systems (Caropresso *et al.*, 1974). Jenkins *et al.*, (1986) reported on the results of several investigators using hydrogen peroxide. The doses and period of application for effective filamentous organism reduction and bulking vary from case to case. Doses vary from 9 to 400 $mg.l^{-1}$. Hydrogen peroxide has been successful when dosed both continuously and on a batch basis. Dosing into the aeration basin, the RAS line and the mixed liquor channels between the aeration basin and the secondary clarifier was successful. Excellent initial mixing of the peroxide with the activated sludge is necessary for the effective destruction of the filamentous organisms. One of the cases (Anon., FMC Corp., 1976 cited by Jenkins, *et al.*, 1986) suggests that a greater contact time is required for hydrogen peroxide than for chlorine. In this case dosing of 20 $mg.l^{-1}$ at a point which was 5 minutes flow time ahead of the secondary settling tank in the mixed liquor channel resulted in foam on the clarifier and no bulking control. When the dose point was relocated to the overflow of the aeration basin 15 minutes flow time ahead of the settling tank, a dose of 12 mg $H_2O_2.l^{-1}$ reduced the SVI from 580 $ml.g^{-1}$ to 178 $ml.g^{-1}$ within 2 days.

The action of H_2O_2 on the filamentous organisms is reported to be one of attacking the sheath, thus destroying the filamentous form (Caropresso *et al.*, 1974). The effect observed, regardless of the mechanism, apparently is similar to that of chlorine, the filaments break up and become shorter and cells within the filaments show signs of lysis.

Hydrogen peroxide has an advantage in it that does not leave any harmful residuals since it decomposes to

water and oxygen. It therefore not only kills the filamentous organisms causing bulking, but in the oxidation/reduction reactions (that accompany disinfection) oxygen is produced that is available for supplementing the dissolved oxygen level:

$$2H_2O_2 \rightarrow H_2O + O_2$$

Should the cause of bulking be due to low DO, then this oxygen should also contribute in the amelioration of the problem. However, should the hydrogen peroxide degrade prior to its availability for destroying the filamentous organisms, its effectiveness for controlling bulking may be reduced.

Bench scale tests performed at the University of Pretoria with nutrient removal activated sludge indicated that a continuous dosage of 5 - 10 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} directly into the aeration basin controlled bulking and did not appear to interfere with the nutrient removal process (van Leeuwen, 1990). Even massive doses of 100 $g.kg^{-1}$ MLSS.d^{-1} did not affect the sensitive nutrient removal organisms. It was further found that a dose of 8 $g.kg^{-1}$ MLSS.d^{-1} was required for bulking control. It was observed during the periods with Types 0092 and 0041 dominating that such doses of H_2O_2 could reduce the numbers of both organisms but that they remained the dominating filaments (van Leeuwen, 1992).

2.5.2.2.1.2 Ozone

Ozone, O_3, is a pungent gas often smelled in the vicinity of discharging high-voltage apparatus and sometimes noticed during thunderstorms. Ozone is an unstable allotrope of oxygen:

$$\frac{3}{2}O_2 \rightarrow O_3 \quad \Delta H = +285\ kJ$$

At room temperature, the equilibrium constant for this reaction is extremely low. Thus, even though this strongly endothermic reaction is favoured by higher temperatures, the equilibrium concentration of O_3 does not become appreciable at any temperature, and not much O_2 can be converted to O_3 by the simple addition of heat. However, when energy is added in other forms, such as electric energy or high-energy radiation, significant amounts of O_3 are obtained, which slowly revert to O_2. Ozone is best prepared by the action of a silent discharge on air or oxygen. This discharge differs from ordinary discharge in that one or two layers of an insulating material, such as glass or mica, are inserted between the metallic terminals. The formation of large sparks or of an electric arc, which tend to produce oxides of nitrogen (Lowry and Cavell, 1958), is thereby prevented, and the discharge is restricted to a series of minute sparks oscillating to and fro between

the insulating surfaces. A potential of 10 000 - 20 000 volts is applied across the metallic terminals. Concentrations as high as 5 percent are easily obtained. A second method for preparing ozone is the irradiation of oxygen with ultraviolet light. This method is used for the preparation of low concentration of ozone for sterilization and disinfection purposes. A large amount of ozone is produced photochemically in the stratosphere. Trace amounts in the lower levels of the atmosphere can increase more than tenfold under the right combination of sunlight with industrial and automotive pollutants, as occurs under smog conditions. In a third preparative method, the electrolytic one, cold aqueous solutions of sulphuric or perchloric acid are electrolyzed with extremely high anodic current densities. Oxygen and, in the case of sulphuric acid solutions, peroxydisulfuric acid, $H_2S_2O_8$, are by-products.

When ozonised oxygen is cooled by means of liquid air, the ozone condenses to a deep blue liquid. Pure liquid ozone, prepared by the fractional distillation of this liquid, boils at - 112 °C. It is fairly stable when pure, but explodes violently in the presence of the smallest trace of organic matter.

Ozone is more soluble in water than oxygen and imparts to it a disagreeable taste and smell. Water which has been disinfected by means of ozone must therefore be exposed to air in order to render it palatable again.

The use of ozone in wastewater treatment has so far been limited to tertiary processes where it is used in improving effluent quality for reuse. Ozone converts many organic substances to more biodegradable forms. A small ozone dosage followed by biological treatment can achieve better removal of organic substances than biological treatment alone (van Leeuwen, 1988a). Ozone is a more powerful oxidant than either hydrogen peroxide or chlorine and normally it does not form toxic residues nor does it contribute to salinity. However only one reference (Saayman and Van Leeuwen, 1992) to its use in full-scale bulking control has been found. Because of the limited data on which this paper was based the results are inconclusive.

Ozone is effective in the control of non-filamentous and filamentous bulking control in activated sludge systems. In a pilot plant treating fuel synthesis waste water the addition of 1 g O_3. kg^{-1} MLSS.d^{-1} reduced the growth of zoogloeal growth which causes poor sludge settleability and to some extent the formation of pinpoint flocs (Van Leeuwen (1989)). It leads to improved COD removal as well. Van Leeuwen (1988b) and van Leeuwen and Pretorius (1988) investigated the use of ozone for bulking control in nutrient removal activated sludge systems on pilot scale. Van Leeuwen (1988) indicated that an ozone dose of 2 g.kg^{-1} MLSS.d^{-1} could lower the DSVI from 180 to less than 100 $ml.g^{-1}$ within one sludge age without affecting the nutrient removal processes. Even at elevated doses of 30 g.kg^{-1} MLSS.d^{-1} the nutrient removal processes were not disturbed. Van Leeuwen and Pretorius (1988) reported that filamentous growth can be controlled by a continuous dosage of 4 g O_3.kg^{-1} MLSS.d^{-1} in the aerobic zone of an activated sludge system.

At this level of ozonation an activated sludge with superior settling properties is formed. They also found that bulking control with ozonation is not significantly more expensive than bulking control with chlorination or solving the problem associated with bulking with additional settler capacity. Saayman and Van Leeuwen (1992) observed in limited trials that a dosage of 0,3 to 0,7 g $O_3.kg^{-1}$ MLSS.d^{-1} in a full-scale biological nutrient removal plant reduced the DSVI from an average of 180 to 120. Temporary inhibition of phosphate removal was observed but no significant inhibition of nitrification/denitrification occurred.

2.5.2.2.1.3 Chlorine

Chlorination of activated sludge is the most generally used non-specific bulking control procedure. The procedure is well documented in the literature such as in the bulking control manual of Jenkins *et al.* (1986) They suggest that the chlorine be employed at a point of lowest oxygen demand and where there will be rapid dispersion of the chlorine. They tested a number of dosing points, i.e.

- into the return activated sludge flow (RAS);
- directly into the aeration basin; and
- into a sidestream abstracted from and returned to the aeration basin.

The most common and preferred chlorine dosing point is into the RAS flow. Excellent initial mixing of the chlorine solution is however of the utmost importance. The result of poor initial mixing is the consumption of chlorine without control of bulking. Prolonged dosing with poor mixing can lead to the production of turbid effluent and a reduction of treatment efficiency.

Jenkins *et al.* (1986) set several criteria that must be followed for successful bulking control by chlorination. The more important ones are:

- A target value of the SVI or some other sludge settling property must be established. This value should be the value below which the plant can be operated satisfactorily. In the setting of this target value the impact of the poorly settling sludge on the secondary settling tanks as well as the waste activated sludge (WAS) processing units must be evaluated.
- Chlorination should be used only when the target value is significantly and consistently exceeded. Trend plots should be made to see when the target value is being approached. Such plots will aid in deciding whether a given value in excess of the target value is consistent with the trend or is the result of a measurement error or is an outlaying data point. Trend plots also assist in fine tuning the chlorine dose to anticipate changes in the values.
- The chlorine must be dosed in known and controlled doses to all of the activated sludge at a point of excellent mixing. Ideally a separate chlorinator should be dedicated to chlorination for bulking

control because the dosing rates required are usually much lower than those used for effluent disinfection. If the chlorine solution is taken off an effluent chlorinator line an independent rotameter and sampling point is mandatory because the flow rate and concentration in this line must be known accurately. Jenkins *et al.* (1986) identified four parameters that appear to influence the efficiency of chlorination in bulking control:

Overall mass dose rate, T_m g $Cl_2.kg^{-1}$ MLSS.d^{-1}

$$T_m = \frac{\textit{mass of chlorine dosed per day}}{\textit{mass of sludge in system (including sludge in settling tank if appreciable)}}$$

Chlorine concentration at the dosing point, C mg $Cl_2.l^{-1}$

$$C = \frac{\textit{mass chlorine dosed per day}}{\textit{flow rate past dosing point}}$$

Local mass dose at the dosing point, T g $Cl_2.kg^{-1}$ MLSS

$$T = \frac{\textit{mass of chlorine dosed per day}}{\textit{TSS mass flow rate past dosing point}}$$

Frequency of exposure of activated sludge to chlorine dose, F d^{-1}

$$F = \frac{\textit{TSS mass flow rate past dosing point}}{\textit{TSS mass in system}}$$

Neethling *et al.* (1985) examined the importance of these parameters. They came to the conclusion that there is a limiting frequency of exposure of the activated sludge to chlorine below which no control of bulking can be achieved. According to Jenkins *et al.* (1986) a frequency of F>3 times per day will provide a sufficient frequency of exposure of the sludge inventory to the chlorine to obtain sludge bulking control. This limiting value depends on the relative growth rates of the filaments and floc-formers, the relative number of filaments and floc-formers and the relative survival of filaments and floc-formers when exposed to chlorine.

Lakay *et al.* (1988) investigated bulking control with chlorination in a UCT nutrient removal activated sludge system on a laboratory-scale plant. Chlorine was dosed into the stream between the second anoxic and aerobic reactors. The final total mass dose was 8 g $Cl_2.kg^{-1}$ MLSS.d^{-1}. At this dose the chlorine concentration at the dose point was 7,6 mg $Cl_2.l^{-1}$, the frequency of exposure 4,7 d^{-1} and the local mass dose 1,7 g $Cl_2.kg^{-1}$ MLSS. They came to the following conclusions:

- Filamentous bulking caused by Type 0092, Type 0914 and *M. parvicella* was controlled by chlorination.
- With a dosing program of 5 days at an overall mass dose rate of 2 g $Cl_2.kg^{-1}$ MLSS.d^{-1}, 6 days at 4 g $Cl_2.kg^{-1}$ MLSS.d^{-1} and 19 days at 8 g $Cl_2.kg^{-1}$ MLSS.d^{-1} the DSVI decreased from 230 to 48 $ml.g^{-1}$. Dosing was terminated when the system shows overdosing symptoms, e.g. high effluent

turbidity.

- During dosing COD removal was essentially unchanged except when overdosing became apparent. Nitrification was unaffected and denitrification only marginally affected. Biological phosphate removal was initially reduced at the higher dose, but recovered in the following 4 to 5 days. After 16 days at 8 g $Cl_2.kg^{-1}$ MLSS.d^{-1} P-removal declined. After cessation of chlorination P-removal recovered within 5 days.
- Of the three filamentous organisms Type 0914 was the least and *M. parvicella* the most resistant to chlorination. Type 0914 disappeared completely from the system while *M. parvicella* and Type 0092 was considerably reduced. About a week after termination of chlorination regrowth of *M. parvicella* and Type 0092 commenced and Type 0914 reappeared.

2.6 SUMMARY

Sludge bulking caused by the excessive growth of filamentous organisms is a problem in most biological nutrient removal activated sludge plants in South Africa. This leads to the carry over of solids in the final effluent and makes the operation of the plants difficult. It also increases the dewatering cost of the sludge.

There are two general approaches towards sludge bulking control:

- Specific control methods.

  Although a number of site-specific remedies have been evaluated and proven successful, it is concluded from the literature that existing specific control methods are not always effective in controlling the proliferation of the low F/M filaments present in N and P removal activated sludge systems. If the hypothesis that intermediate products of denitrification favour the development of filamentous organisms in biological nutrient removal plants is true then the excessive growth of these organisms is an inherent phenomenon in such plants, because nitrification and denitrification can occur simultaneously in the aerobic zone of the plant. On the evidence available to date one cannot design and operate a BNR plant that will never bulk with low F/M filamentous organisms.

- Non-specific control methods.

  The application of oxidants in biological nutrient removal systems was investigated mainly on bench scale plants. The methods are successful but there are cost and quality implications, such as the potential formation of organic derivatives, to it.

Chlorine has been used for more than fifty years in sludge bulking control in fully aerobic activated sludge systems. Although it controls sludge bulking in biological nutrient removal plants it causes loss of nitrification and phosphorus removal, and turbid effluent in full-scale plants. This may be due to over dosing of chlorine because operators of full-scale plants tend to chlorinate only when bulking has reached crisis proportions. In this case chlorine is applied in shock doses.

Two promising oxidants for sludge bulking control in nutrient removal systems are hydrogen peroxide and ozone. Hydrogen peroxide was used in full-scale fully aerobic activated sludge plants for sludge bulking control but not in biological nutrient removal activated sludge systems. On bench scale hydrogen peroxide controlled sludge bulking in nutrient removal systems without influencing the nutrient removal process. In pilot plant investigations ozone was effective in combatting filamentous and non-filamentous sludge bulking. The nutrient removal process and chemical oxygen demand (COD) removal was not disturbed, if anything, improved. Only limited work on full-scale plant application has been reported.

2.7 CONCLUSIONS AND RECOMENDATIONS

There is a need to investigate non-specific sludge bulking control methods in a full-scale plant in order to assess if the technologies are successful and if so to transfer the technologies to consultants and operators of full-scale plants. Consultants will not easily recommend the application of a capital intensive technology, such as ozone, which has not been proven in full-scale plants. There is also a need to investigate if these technologies can be applied in full-scale plants as preventative bulking control measures and not as shock treatments only.

The biological nutrient removal activated sludge plant at the Daspoort Works of the City Council of Pretoria consists of three parallel, identical units making it ideally suitable for the comparison of sludge bulking control chemicals. While one of the units could serve as a control, the chemicals, hydrogen peroxide, ozone or chlorine could be dosed into others.

The research which forms the basis of this thesis was performed over a period of 39 months from May 1991 to August 1994 at the Daspoort works. The objective was to investigate the feasibility of the techniques of non-specific bulking control measures in a full-scale biological nutrient removal plant, and to determine the effect of hydrogen peroxide, ozone and chlorine on biological nitrogen and phosphorus removal, and on the filamentous species composition.

CHAPTER 3

EXPERIMENTAL DESIGN AND PROCEDURE.

3.1 DESCRIPTION OF THE PLANT.

The Daspoort activated sludge plant consists of three identical modules or units, designated as Unit 9, Unit 10 and Unit 11. These units have been operational since April 1975 and were designed to treat a total volume of 13 $Ml.d^{-1}$ each,and were not originally designed for biological nutrient removal. All the units were upgraded to 3-stage Bardenpho nutrient removal plants. A diagrammatic outlay of one of the upgraded units is shown in Figure 4.

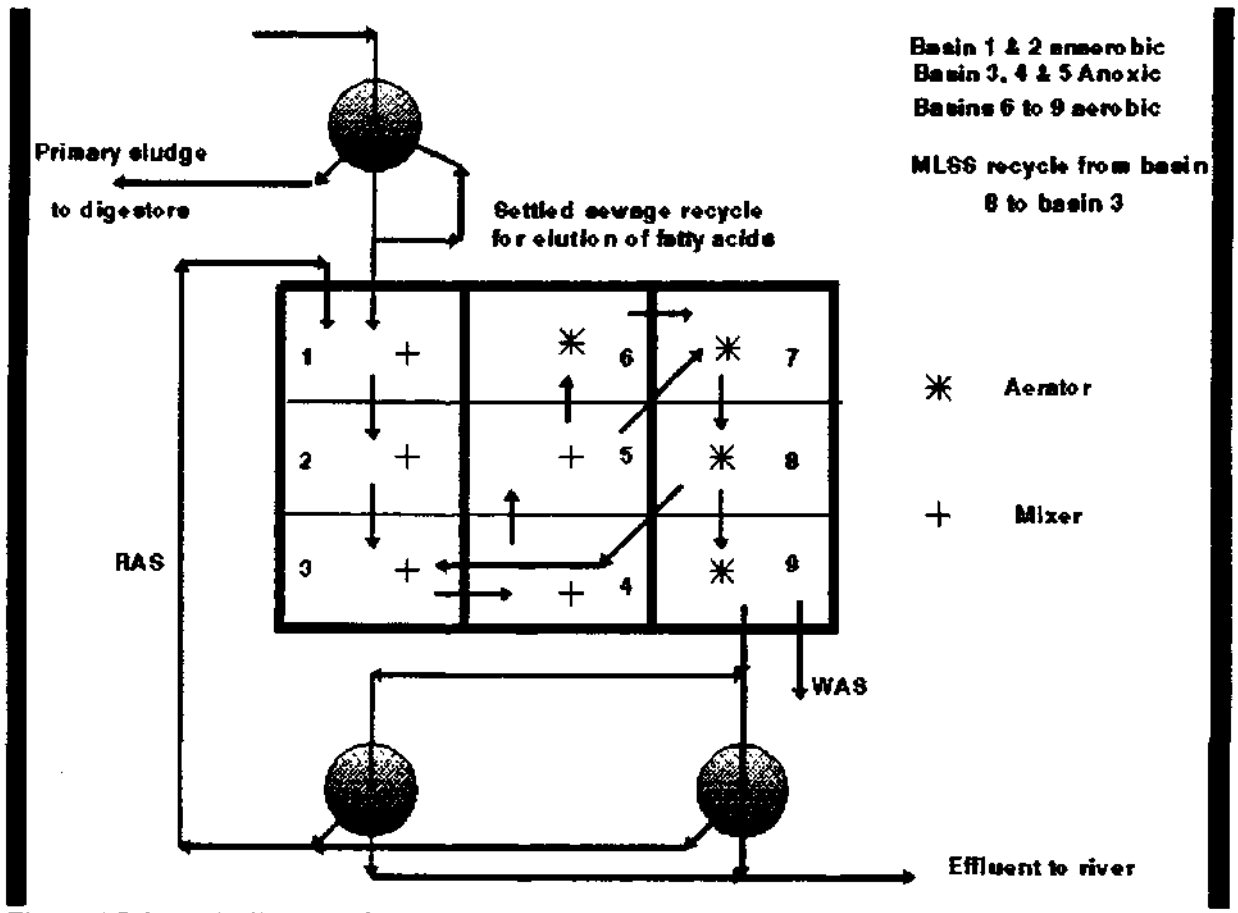

Figure 4 Schematic diagram of one of the biological nutrient removal units of the plant at Daspoort.

After upgrading, the preliminary treatment consists of two mechanically back-racked bar screens and two gravity degritters and primary sedimentation in three Dortmund tanks. The Dortmund settling tanks were

transformed into "biological fatty acid reactors" or activated primary settling tanks in order to increase the soluble readily biodegradable fraction of the influent COD. In the activated primary settling tank the primary sludge is allowed to form a sludge blanket in which the formation of short chain fatty acids by anaerobic fermentation is possible. Settled sewage is recycled through the desludging pipeline to the bottom of the settling tanks into the settled sludge in order to elutriate short-chain fatty-acids formed in the primary sludge by fermentation. The primary settling tanks are desludged every fifth day and the primary sludge digested in anaerobic digesters.

The biological reactors of each unit are divided into nine compartments. Each compartment has a volume of 750 m^3. Compartments 1 to 4 are equipped with submersible mixers, compartment 5 with a mixer and a surface aerator, and compartments 6 to 9 with surface aerators only. Recirculation from the penultimate aerobic to the first anoxic basin is achieved by three internal recycling pumps, each with a capacity of 300litres per second, from compartment 8 to compartment 3. This results in a configuration of :

Zone	Volume in cubic metres	Compartments numbers
Anaerobic	1 500	1 & 2
Anoxic	2 250	3, 4 & 5
Aerobic	3 000	6, 7 ,8 & 9.

There are two secondary settling tanks per unit. The volume of each secondary settling tank is 1 850 m^3 giving a total settling tank volume of 3 700 m^3 per unit.

Return activated sludge from the secondary sedimentation tanks is pumped to the head of the reactor by means of screw pumps where it is mixed with the incoming settled sewage before it enters the first anaerobic basin.

Sludge is wasted continuously directly from compartment 9 of the reactors to maintain the sludge age in the reactor. The wasted sludge is thickened by dissolved air flottation. The thickened sludge is mixed with primary sludge to remove all dissolved oxygen before it is transferred to the anaerobic digesters for stabilisation.

The dissolved oxygen concentration in the aerobic zone is regulated by adjustable weirs.

Ozone was generated in a Degrémont type MB-110-G serie 4 horizontal-tube 50 Hz ozonizer with a capacity of 2 kg $O_3.h^{-1}$.

3.2 OPERATIONAL CONDITIONS.

The primary settling tanks were operated as active primary settling tanks. They were desludged every fifth day to prevent the establishment of methanogenic bacteria that can convert short chain fatty acids to methane. The anaerobic fermentation products, which is mainly short-chain fatty acids, were elutriated by recycling settled sewage at a rate of 1750 *l* per minute into the settled sludge in the bottom of the settling tanks. The recycling was timer controlled and started at 13:00 in the afternoons and ended at 06:00 the next morning. During this period the recycling pumps were on for 2,5 hours and off for 0,5 hour to prevent channeling through the sludge layer.

All efforts have been made to keep the conditions in the activated sludge reactors constant. The conditions were:

	Average	Standard deviation
Influent	13,26 M*l*.d^{-1}	1,01 M*l*.d^{-1}
Return activated sludge	14,38 M*l*.d^{-1}	2,40 M*l*.d^{-1}
Internal recycle	51,84 M*l*.d^{-1}	N.A.
Mixed liquor suspended solids	3 225 mg.*l*$^{-1}$	240 mg.*l*$^{-1}$
Sludge age	11,31 days	0,96 days

The DO level was maintained between 0,5 to 1,5 mg.*l*$^{-1}$.

The units were fed with settled wastewater, 95 % domestic and 5 % industrial. At dry weather flow, the following average composition based on three 24 hour composite samples per week, was measured:

	Average	Standard deviation
Permanganate value mg.l^{-1} as O	37,8	4,3
Chemical oxygen demand mg.l^{-1} as O	469	47
Total Kjeldahl nitrogen mg.l^{-1} as N	42,4	1,3
Total phosphorus mg.l^{-1} as P	10,1	0,7
Alkalinity mg.l^{-1} as Ca CO_3	224	14
pH	7,4	0,1

3.3 CHEMICAL DOSING.

It was planned to compare the bulking control capability of hydrogen peroxide and ozone during the first phase of the project. However technical problems with the ozone generator caused a considerable delay and at the request of the supplier of the hydrogen peroxide it was decided to continue with the evaluation of the peroxide. In phase 1 of the project Unit 10 was treated will hydrogen peroxide. Unit 9 was used as a control.

After the problems with the ozonizer were solved phase 2 of the project comprised chlorination of Unit 10 and ozonation of Unit 11 at a rate of 2 kg oxidant per hour, with Unit 9 as the control.

In phase 3 of the project Unit 10 was chlorinated at a rate of 4,33 kg.h^{-1} with Unit 9 as the control. To ensure that the activated sludge was comparable the return activated sludge of the units was combined during the week preceeding the experimental period. During normal operation the return activated sludge of the units is kept separate.

3.3.1 Hydrogen peroxide dosing.

Hydrogen peroxide has been studied in a bench scale nutrient removal system at the University of Pretoria (internal report). It was found that an overall dose of 8 g H_2O_2.kg^{-1} MLSS.d^{-1} controlled bulking. To make hydrogen peroxide economically viable as a continuous means of bulking control, it was decided to start the investigation at a low dose of 1,5 g H_2O_2.kg^{-1} MLSS.d^{-1}.

Hydrogen peroxide was dosed as a 50 % solution into the draught tube of the aerator to ensure rapid mixing. In the first experiment the H_2O_2 was initially dosed into the final aeration basin (basin 9). According to Caropresso *et al.*, (1974) hydrogen peroxide destroys filamentous organisms by attacking the sheath of the

filaments. In preliminary bench work bulking control was achieved in a system where Type 0092 was the dominant organism. Type 0092 does not have a sheath. This may imply that the action of hydrogen peroxide is similar to that of chlorine. It was therefore decided to dose it at a point where the hydrogen peroxide demand is at a minimum. The hydrogen peroxide was dosed at a rate of 4,25 $kg.h^{-1}$, from day 34 to day 67, and 8,39 $kg.h^{-1}$, from day 68 to day 86. This was equivalent to 1,5 g $H_2O_2.kg^{-1}$MLSS.d and 3,0 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} respectively. Bulking control was not as good as was predicted from the bench scale work.

In the preliminary bench work, a system with a single completely mixed aerobic zone was used, it was therefore decided to move the dosing point to the first aeration basin. Dosing into the first basin would expose a larger fraction of the aerated sludge for a longer period to the hydrogen peroxide. From day 87 to day 102 the hydrogen peroxide was dosed into the first aeration basin (basin 6) at a rate of 8.39 $kg.h^{-1}$. This was equivalent to 3,0 g $H_2O_2.kg^{-1}$ MLSS.d^{-1}

In the second experiment H_2O_2 was dosed from day 16 to day 43 into the first aeration basin at a rate of 26,67 $kg.h^{-1}$, this was equivalent to 9,5 g $H_2O_2.kg^{1}$ MLSS.d^{1} . The parameters for the various hydrogen peroxide doses, that may influence chemical bulking control, are summarised in Table 3.

Table 3 Parameters for Hydrogen Peroxide that may influence bulking control

Dose $kg.h^{-1}$	4,25	8,39	26,67
Overall mass dose, T_m g $H_2O_2.kg^{-1}$ MLSS.d^{-1}	1,5	3,0	9,5
Concentration at dose point , C mg $H_2O_2.l^{-1}$	1,8	3,6	11,6
Local mass dose T, g $H_2O_2.kg^{-1}$ MLSS	0,6	0,8	3,6
Frequency of exposure F d^{-1}	2,64		

3.3.2 Ozone dosing.

The ozone generator used in this study was purchased in 1976 by the Water Research Commission. It is a horizontal-tube ozonizer made of stainless steel. Basically it consist of a cylindrical shell, containing a nest of horizontal stainless steel tubes, open at both ends and welded to circular vertical end plates also made of stainless steel. The cylindrical shell, metal tubes and circular end plates form a central enclosure within which water is circulated to cool the tube nest. The cylindrical shell is longer than the tube nest and is sealed at both ends by tight convex covers with inspection ports. Hence there are two chambers which are inter-connected by the stainless steel tubes.

Concentric dielectric tubes of special Pyrex type glass are placed inside the stainless steel tubes. The dielectrics are centred in relation to the metal tubes by stainless steel centring pieces to ensure a uniform gap between the dielectric tubes and the stainless steel tubes. The metal tube and shell is earthed and behaves like an external electrode. The inside walls of the dielectric tubes are lined with a metallic coating which is connected by stainless steel contacts to a common high-voltage terminal. A high potential difference arises between the metal electrodes and the internal metallic coating of the dielectric tubes while dried air flows through the annular openings between the metal electrodes and the external wall of the dieletric tubes.

The 50 Hz single-phase alternating current on which the ozonizer works is supplied by a step-up voltage transformer. One end of the secondary winding is connected to the ozonizer earth and the other to the insulated input terminal, which is in turn connected by contacts to each of the elctrodes formed by the internal metallic coating of the dielectric tubes.

Air is supplied by a compressor set at a pressure of 6 Bar. A two-stage low pressure drying process was used to remove dust oil and moisture from the air to a dew point of approximately - 60 °C. The air is first fed to a cooler unit where most of the water vapour it contains condenses. The cold almost saturated air is then fed to an adsorbent type drier. The drier has two dessicant cylinders, one of which is in service while the other is being regenerated. Regeneration is automatic and consists of blowing dry air through the saturated material to remove adsorbed water vapour. The dried air is fed to the regenerator at ambient temperature.

Based on the assumption that the mechanism of bulking control was one of disinfection it was decided to dose the ozone in that part of the reactor where the ozone demand for other oxidation reactions was the lowest. Ozone was, therefore, introduced into the last aeration compartment (no. 9) of Unit 11 by means of fine bubble ceramic diffusors mounted on the floor of the reactor in the downward action-zone of the surface aerator. This point was chosen because the counter flow of the liquid and gass bubbles should facilitate a better transfer of ozone. However, attempts to measure the transfer efficiency of the ozone failed because of the turbulence caused be the surface aerators.

From day 20 to day 62 ozone was dosed at a rate of 2 kg $O_3.h^{-1}$. Due to operational problems with the ozone generator the dose was lowered to 1,5 kg $O_3.h^{-1}$ on day 62. On day 86 the dosage had to be further reduced to 0,5 kg $O_3.h^{-1}$. This dosage was maintained till the end of the experimental period. The parameters for ozone are summarised in Table 4.

Table 4 Parameters for Ozone that may influence bulking control

Dose $kg.h^{-1}$	2,0	1,5	0,5
Overall mass dose, T_m g $O_3.kg^{-1}$ MLSS.d^{-1}	1,42	1,07	0,36
Concentration at dose point , C mg $O_3.l^{-1}$	1,74	1,3	0,43
Local mass dose T, g $O_3.kg^{-1}$ MLSS	0,54	0,4	0,13
Frequency of exposure F d^{-1}	2,64		

3.3.3 Chlorine dosing.

Chlorine was introduced as an aqueous solution into the last aeration compartment where the chlorine demand is at a minimum.

In the first trail (phase 2 of the project) dosing of chlorine at a rate of 2 $kg.h^{-1}$ started on operational day 20. This rate was maintained untill the target DSVI was reached on day 63. After the target DSVI was achieved the dosage was reduce to 1 $kg.h^{-1}$ to see whether the low DSVI can be maintained and if the nutrient removal will recover while the activated sludge is chlorinated. When it became clear that the deterioration in nutrient removal was apparently caused by other factors than chlorination the dosing of chlorine was stopped on day 82.

In the second experiment (phase 3 of the project) chlorine was dosed at a rate of 4.33 $kg.h^{-1}$ for a period of 16 days. The parameters for the various chlorine doses, that may influence bulking control, are summarised in Table 5.

Table 5 Parameters for chlorine that may influence bulking control

Dose $kg.h^{-1}$	2,0	1,0	4,33
Overall mass dose, T_m g $Cl_2.kg^{-1}$ MLSS.d^{-1}	1,42	0,71	3,07
Concentration at dose point , C mg $Cl_2.l^{-1}$	1,74	0,87	3,74
Local mass dose T, g $Cl_2.kg^{-1}$ MLSS	0,54	0,27	1,17
Frequency of exposure F d^{-1}	2,64		

3.4 CHEMICAL AND PHYSICAL ANALYSIS.

Daily grab samples were withdrawn from the final aerobic zone for the determination of the sludge volume index of the activated sludge. The thirty minute diluted sludge volume index ($DSVI_{30}$) was determined by using the procedure described by Jenkins, *et al.* (1986)

Ortho-phosphate, nitrate (nitrate+nitrite) and ammonia were analysed with on-line Brann &Luebbe auto-analysers on a hourly basis.

Time plots, smoothed by using 5-day moving averages, of the diluted sludge volume indices of the daily grab samples of the activated sludge, and of the daily average of hourly phosphate, ammonia and nitrate analysis, during the experimental periods, are presented and discussed in Chapter 4.

Microscopic examination on weekly grab samples were conducted following the procedure suggested by Jenkins *et al.* (1986).

CHAPTER 4
RESULTS AND DISCUSSION

4.1 The effect of the oxidants on sludge settleability

One important aspect in the control of sludge bulking is the setting of a target value for the settleability parameter, used in the operation of the plant, below which the plant can be operated satisfactorily without the problems associated with poorly settling sludge. In the setting of this target value the operation of the secondary settling tank as well as the impact of sludge settleability on the sludge handling units must be taken into consideration. In this investigation the diluted sludge volume index (DSVI) was used as a measure of sludge settleability with a value of 100 $ml.g^{-1}$ for the DSVI being set as target.

4.1.1 The effect of hydrogen peroxide on sludge settleability

The diluted sludge volume indices of the activated sludge during the period of the first experiment are represented in Figure 5.

Although the design of both the hydrogen peroxide and control plants were "identical" and specific steps taken to maintain the operating conditions in both plants the same, differences in the DSVI's were observed. In order to be able to evaluate the changes in the DSVI, a regression line (straight line of best fit) is drawn through the daily values. The regression lines for the DSVI's are presented in Figure 6.

During the period of ca 3 sludge ages just prior to dosing H_2O_2 the DSVI in the experimental plant was decreasing at a slightly higher rate than that in the control. The slopes of the regression lines being $-0,22\pm0,15$ and $-0,24\pm0,15$ respectively indicate that the difference in the rate of change is insignificant.

While hydrogen peroxide was dosed at a rate of 1,5 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} into the final aerobic compartment of the plant (basin 9) two trends are discernible in the DSVI of the activated sludge (Figure 5 and 6). Initially the DSVI in the control plant decreased from ca 200 $ml.g^{-1}$ to ca 178 $ml.g^{-1}$ and in the hydrogen peroxide treated plant increased from ca 178 to 185 $ml.g^{-1}$. After ca 1,5 sludge ages the DSVI of the sludge in both plants increased. The rate at which the DSVI increased was similar in both plants but higher than the initial increase observed in the H_2O_2 treated plant.

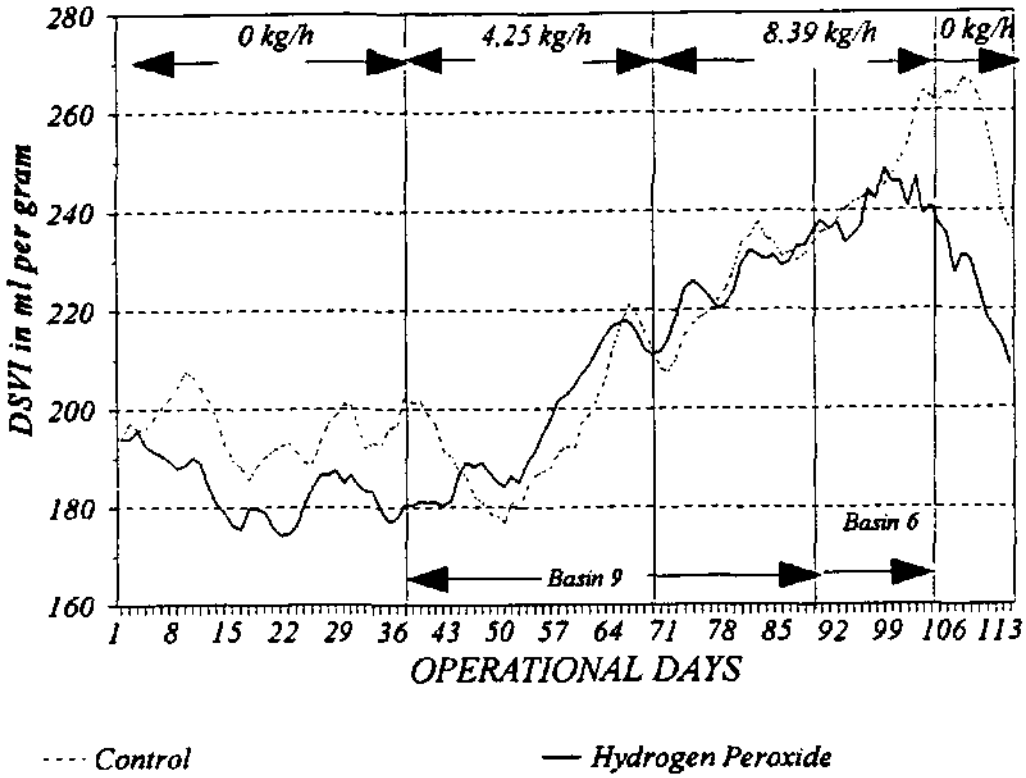

Figure 5 The effect of 1,5 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} (4,25 kg.h^{-1}) and 3 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} (8,39 kg.h^{-1}) on the settleability of activated sludge.

Increasing the dose to 3 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} did not improve the settleability of the activated sludge. The settleability of the sludge continued to deteriorate, however at a marginally lower rate than before. Moving the dosing point from the last to the first aeration compartment had no effect. The DSVI increased at almost the same rate for both dosing points, the slope of the regression line being 1,08±0,33 during dosing in the last aeration compartment and 1,12±0,84 while dosing in the first aeration compartment.

Four days before the addition of the hydrogen peroxide was stopped, the settleability of the activated sludge in the treated plant started to improve. The improvement continued after the dosing was stopped. The settleability of the sludge in the control plant deteriorated for a further 11 days before it started to improve.

At this stage the entire stock of 14 000 kg of hydrogen peroxide was depleted. Due to delivery delays the experiment had to be interrupted while new supplies were awaited.

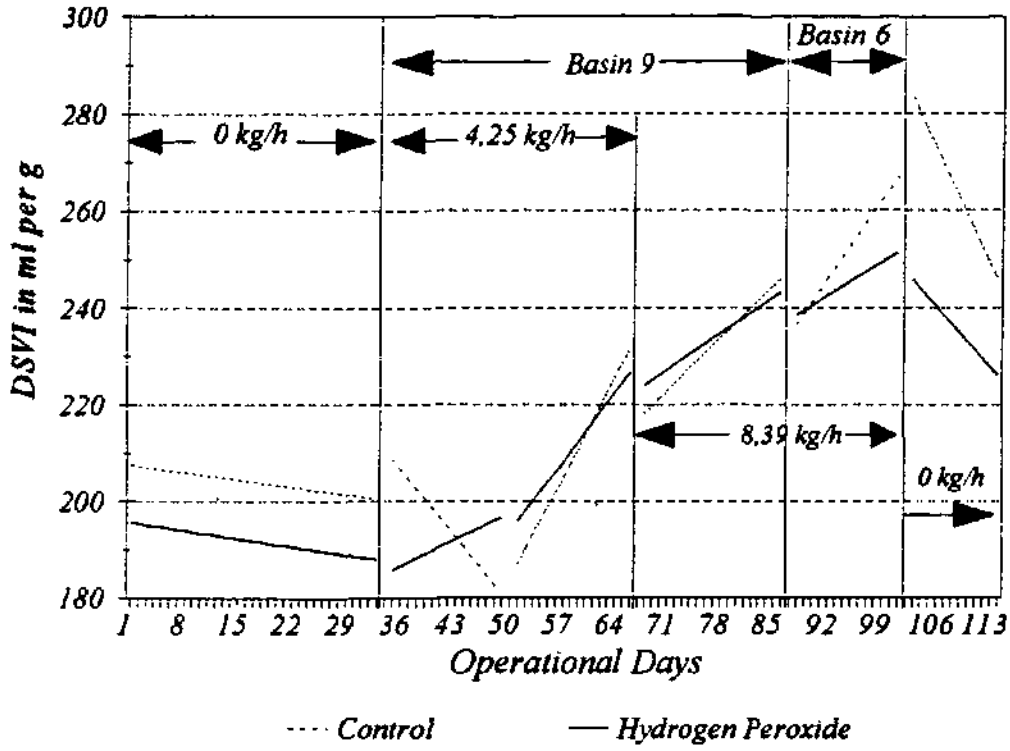

Figure 6 The trends in the DSVI during the addition of hydrogen peroxide at rates of 1,5 and 3 g $H_2O_2.kg^{-1}$ MLSS.d^{-1}.

In the second part of the investigation, which started 2,5 months later, hydrogen peroxide was dosed into the first aeration basin at a rate of 9,5 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} (26,67 kg H_2O_2 .h) into the first aeration compartment. Figure 7 shows the changes in DSVI during this period.

During the period preceding the addition of hydrogen peroxide the DSVI of the activated sludge in the experimental reactor was unstable showing large increases and decreases. When it appeared as if the DSVI was going to exceed the safe limits, it was decided to start treating the sludge with hydrogen peroxide. Initially the settleability of the sludge improved, but after 11 days, one sludge age, it started to deteriorate once more, but at a slightly lower rate rate than before. This deterioration continued for a further 4 days after the addition of the hydrogen peroxide was stopped.

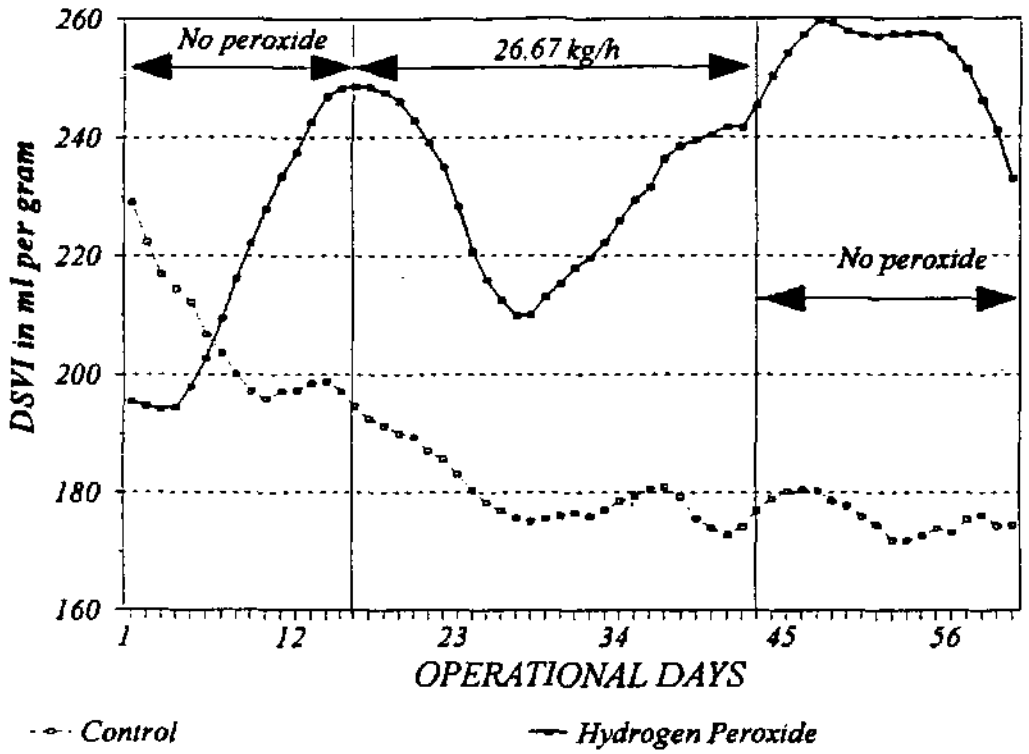

Figure 7 The effect of 9,5 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} (26,67kg.h^{-1}) on the settleability of activated sludge.

In a preliminary investigation into the use of H_2O_2 in a bench scale nutrient removal activated sludge system, Van Leeuwen (1991) found that a continuous dose of 10 mg.l^{-1} (8 g $H_2O_2.kg^{-1}$ MLSS.d^{-1}) was able to lower the sludge volume index in one sludge age. His results indicated that a dosage of 5 mg.l^{-1} (4 g $H_2O_2.kg^{-1}.d^{-1}$) had little effect on the settleability of the sludge in one sludge age of twenty days. It was therefore to be expected that the low dose rates used in the full scale plant may not have the desirable effect. However, due to the high cost of treating a full-scale plant, it was decided to investigate the effect of a low dosage which would make hydrogen peroxide economical viable. The results obtained by Van Leeuwen (1991) could not be reproduced in the full-scale plant. At low doses of 1,5 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} and 3 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} no bulking control was achieved, while at a higher dose, 9,5 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} a temporary improvement of sludge settleability was observed during the first twelve days (ca one sludge age) of treatment. Thereafter sludge settleability deteriorated as before. One of the reasons for this failure to achieve the same results might have been the fact that the bench scale plant consisted of a completely mixed reactor, while the full-scale plant was compartmentalised. Even though the concentration to which the organisms were exposed in the full-scale plant was higher, it did not improve the settleability of the activated sludge.

Due to the high cost of chemicals for full-scale investigations these tests were not duplicated. These observations are based on one set of results only. Further investigations on bench or pilot scale should be

conducted to verify these findings.

4.1.2 The effect of ozone on sludge settleability

The diluted sludge volume indices of the control and experimental plant are shown in Figure 8.

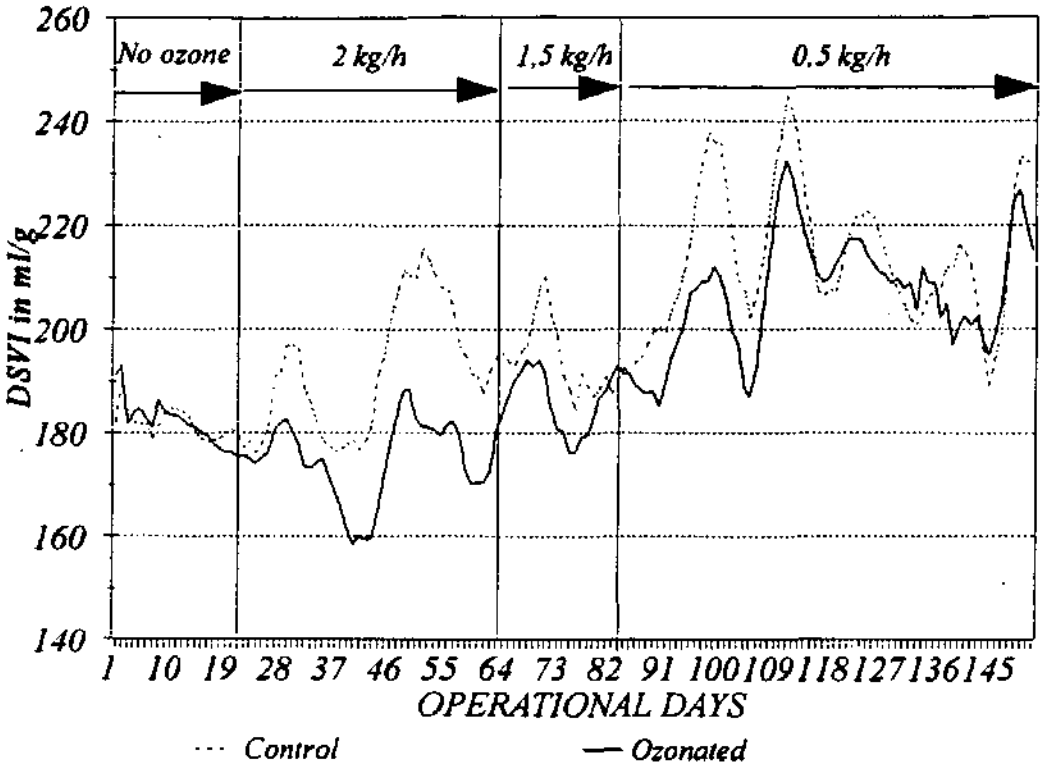

Figure 8 The effect of ozone on the diluted sludge volume index of activated sludge.

The diluted sludge volume indices of both plants were of the same order during the period of ca 2 sludge ages before the addition of ozone. During the initial stages of ozonation at a rate of 1,42 g $O_3.kg^{-1}$ MLSS.d^{-1} (2 kg O_3 .h^{-1}) the DSVI in both plants increased reaching, a maximum on day 29. The increase in the ozonated plant from 175 to 182 was, however, less than that in the control plant, from 180 to 196. The DSVI of the sludge in both plants decreased from day 29. The minimum DSVI in the control plant was 178 on day 37 and in the ozonated plant 158 on day 40. From day 43 to day 49 the DSVI of the activated sludge with ozonation increased from 158 to 188, the maximum DSVI measured during the peroid of the higher ozone dose. The DSVI of the control sludge increased from 178 on day 42 to 216 on day 51. During the dosing of 2 kg $O_3.h^{-1}$ the changes in the DSVI of the activated sludge in the ozonated and control plant followed a similar general pattern. The average DSVI of the ozonated sludge was 175 ml.l^{-1} and that of the control sludge 192 ml.l^{-1}. The trend in the DSVI of the ozonated sludge shows an average daily increase of 0,07 and the control sludge an average daily increase of 0,52. There are many factors that affect sludge settleability and to control bulking successfully the driving force towards non-bulking caused by the agent employed

should be strong enough to neutralise all these factors. During this period some degree of bulking control was achieved but the dosing rate was not high enough to overcome the tendency towards bulking completely. Because of technical problems with the ozone generator the dose rate was lowered. During the period when ozone was dosed at a rate of 1,07 g O_3. kg^{-1} MLSS.d^{-1} (1,5 kg.h^{-1}) the DSVI in the ozonated plant was initially lower than that in the control plant, but towards the end it was higher. The average DSVI for the entire period was lower, for the ozonated sludge it was 187 ml.l^{-1} and for the control sludge 193. The trend in the DSVI of the control plant shows a decrease of 0,85 ml.l^{-1}.d^{-1} and that of the ozonated sludge 0,18 ml.l^{-1}.d^{-1} only. An indication that ozone at a rate of 1,07 g O_3.kg^{-1} MLSS.d^{-1} is in all probability too low to control bulking effectively.

At 0,5 kg.h^{-1} (0,36 g O_3.kg^{-1} MLSS.d^{-1}) the ozone affected sludge settleability very little. On two occasions the tendency towards bulking was lower in the ozonated system than in the control.

The stirred sludge volume index, shown in Figure 9, which is a more representative indicator of sludge settling behaviour in full-scale plants, show similar trends.

Before the addition of ozone both plants had similar SSVI's. At the start of the experimental period the unozonated system developed a serious tendency towards bulking. The ozonated system did not react in the same way to this unknown external stimulus, but the settleability of the sludge continued to improve. The SSVI of the sludge in the experimental plant stabilised at ca 120 ml.g^{-1} and that in the control plant at 140 ml.g^{-1}. Even at the lower dose of only 0,36 g O_3.kg^{-1} MLSS.d^{-1}, the SSVI of the sludge in the ozonated system was consistently lower than that of the unozonated system.

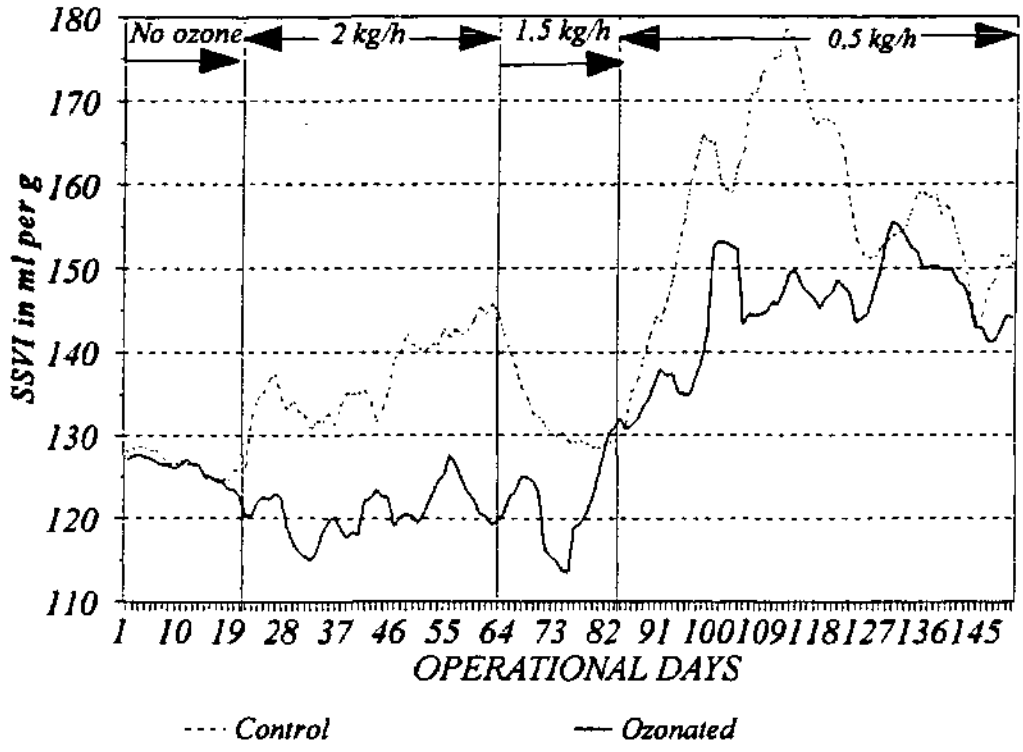

Figure 9 Effect of ozone on the stirred sludge volume index of activated sludge.

Van Leeuwen (1988) applied ozone continuous at overall mass doses of 1, 2 and 4 g $O_3.kg^{-1}$ MLSS.d^{-1} At these concentrations he achieved an average improvement of 50 ml.g^{-1} in the DSVI. The highest overall mass dose that could be achieved with the limited ozone generation capacity was 1,45 g.kg^{-1} MLSS.d^{-1}. This low dose gave an average, but consistent, improvement of 17 ml.g^{-1} in the SSVI over ca 3,5 sludge ages. The lower doses of 1,07 and 0,36 g $O_3.kg^{-1}$ MLSS.d^{-1} gave SSVI's which were on the average 12 ml.g^{-1} lower than that of the control plant. To evaluate the bulking control potential of ozone properly a generator with a capacity of at least 8 kg $O_3.h^{-1}$ would be required.

4.1.3 The effect of chlorine on sludge settleability

The influence of chlorine on the settleability of the activated sludge is shown in Figures 10 and 11.

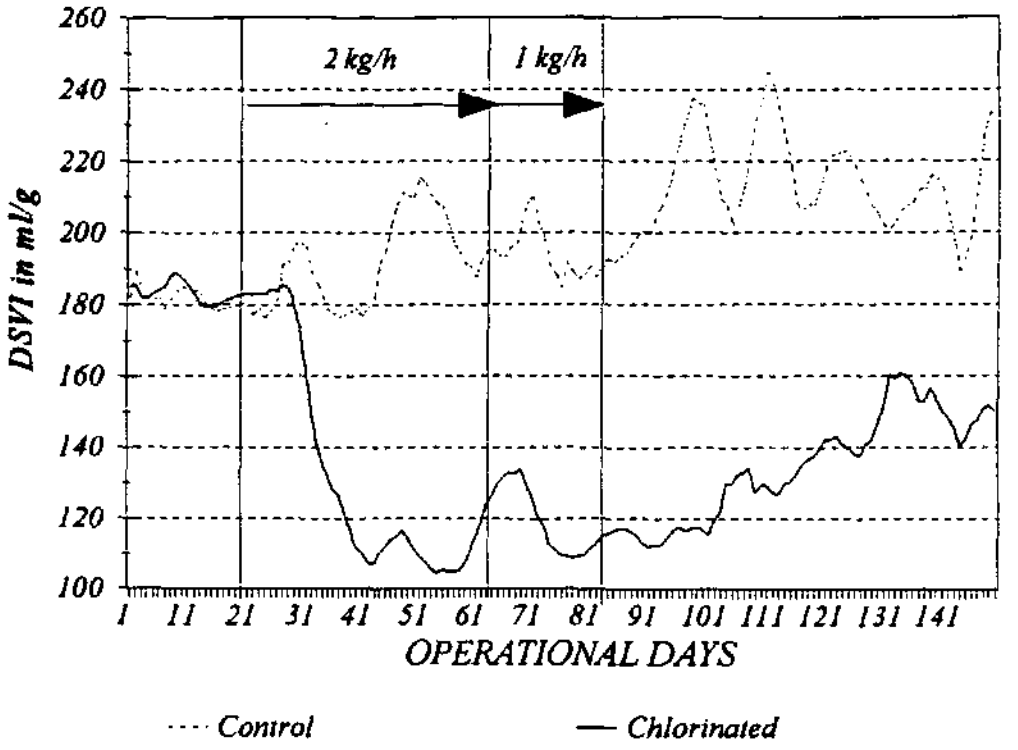

Figure 10 The effect of 1,42 g $Cl_2.kg^{-1}$ MLSS.d^{-1} (2 $kg.h^{-1}$) and 0,71 g $Cl_2.kg^{-1}$ MLSS.d^{-1} (1$kg.h^{-1}$) on the diluted sludge volume index of activated sludge.

During the first 10 days of chlorination at a rate of 2 $kg.h^{-1}$ (1,42 g $Cl_2.kg^{-1}$ MLSS.d^{-1}) there was no improvement in the sludge settleability as measured by the DSVI. After 10 days or ca 1 sludge age the settleability of the chlorinated activated sludge started to improve. The DSVI steadily dropped from 185 to 110 $ml.g^{-1}$ over a period of 21 days. During the period from operational day 44 to day 48 the DSVI increased sligthly to 113 before it started to decrease further to 105. When the value of the DSVI started to increase once more, it became apparent that the target value of 100 $ml.g^{-1}$ would not be reached. It was decided to lower the dosing rate to 1 kg $Cl_2.h^{-1}$ to see if the phosphate removal would recover. When it became clear that factors other than the chlorine was influencing the nutrient removal more than the chlorine, dosing was stopped. From operational day 83 to day 151 the DSVI of the chlorinated activated sludge increased to an average value of 150 $ml.g^{-1}$

During the operational period the DSVI of the activated sludge in the control reactor remained high, at times showing a tendency towards bulking, with the DSVI reaching values of up to 240 $ml.g^{-1}$.

The diluted sludge volume index of the activated sludge measured during the second chlorination trial at a rate of 3,07 g $Cl_2.kg^{-1}$ MLSS.d^{-1} is shown in Figure 11.

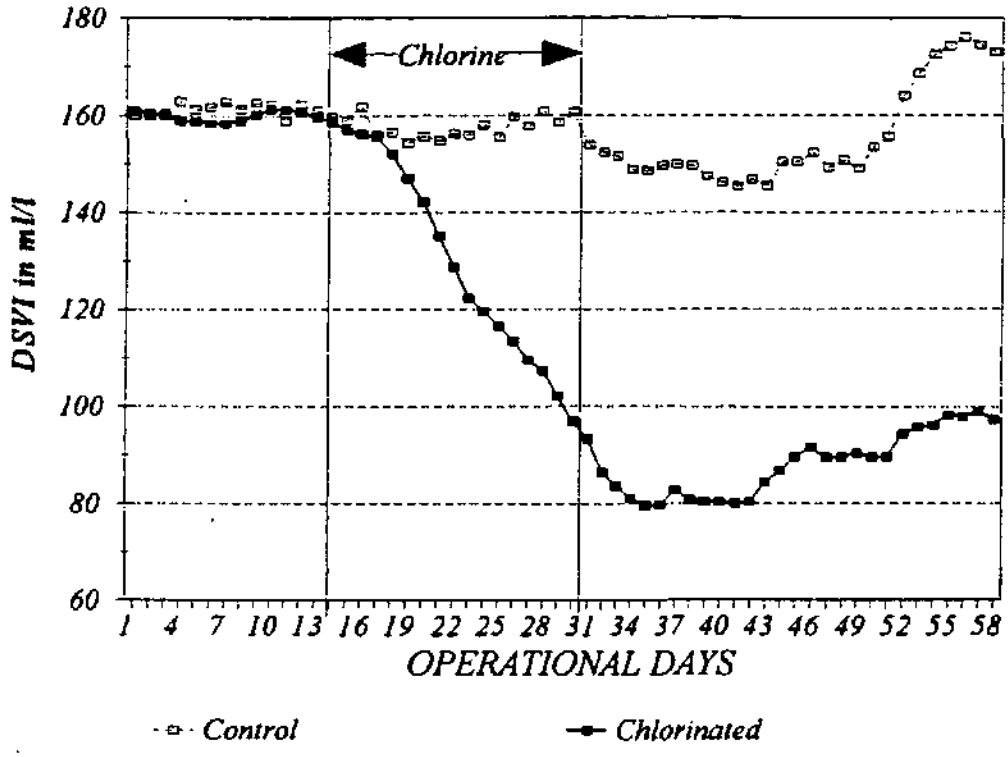

Figure 11 The effect of 3,07 g $Cl_2.kg^{-1}$ MLSS.d^{-1} (4,33 kg $Cl_2.h^{-1}$) on the diluted sludge volume index of activated sludge.

Dosing chlorine at 3,07 g $Cl_2.kg^{-1}$ MLSS.d^{-1} started on operational day 15. On day 17 the DSVI began to decrease reaching the target value of 100 m*l*. g $^{-1}$ on day 29. Dosing was stopped on day 31. The DSVI was 90 m*l*. g $^{-1}$. After cessation of chlorination the DSVI decreased further reaching a minimum value of 80 m*l*. g $^{-1}$ on operational day 34. From day 34 to day 42 the DSVI of the chlorinated sludge remained constant. From day 42 to day 58 the DSVI increased to 100 m*l*. g $^{-1}$.

The DSVI of the activated sludge in the control reactors showed a decreasing trend untill operational day 43. The value of the DSVI decreased from an average of 160 to a value of 145 m*l*.g^{-1} over this period. From day 45 to the end of the experimental period, the DSVI increased from 145 to an average of ca 175 m*l*.g^{-1}.

The effect of the chlorine dose on the settleability of the activated sludge, expressed as normalised DSVI, is shown in Figure 12. In order to compare the results the DSVI in each experiment is expressed as a percentage of the highest DSVI measured during the experimental period.

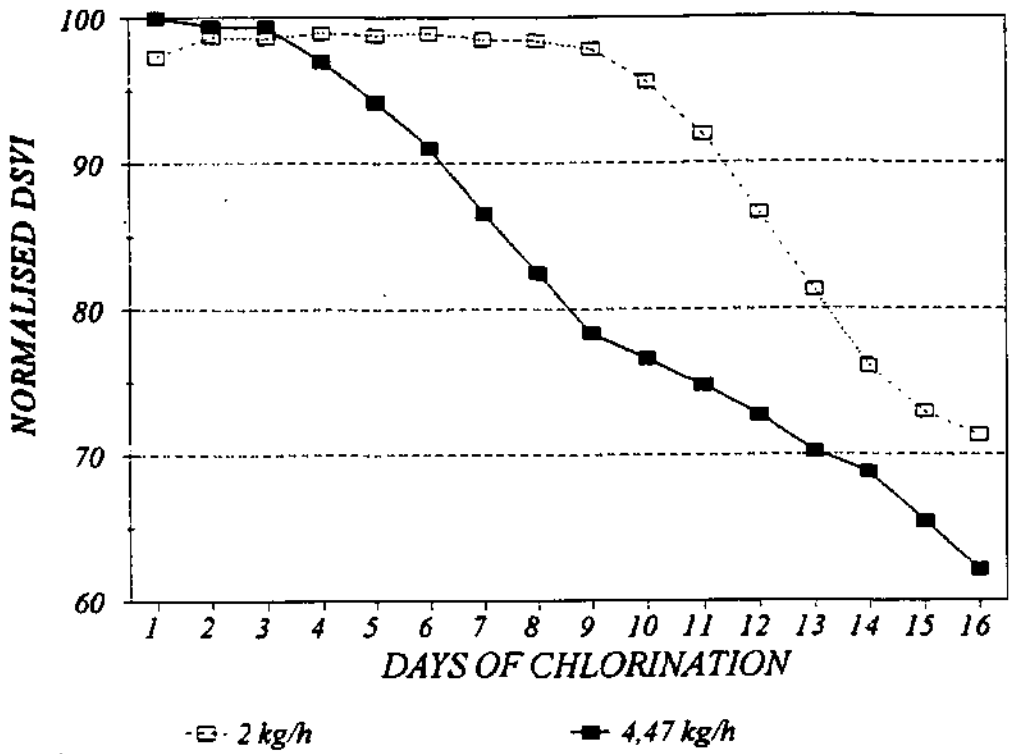

Figure 12 The effect of the chlorine dose on the DSVI of activated sludge.

From Figure 12 it is clear that at Cl_2 dose of 1,42 g $Cl_2.kg^{-1}$ MLSS.d^{-1} bulking control starts after 9 days. At a dose of 3,07 g $Cl_2.kg^{-1}$ MLSS.d^{-1} bulking control was achieved in 3 days. According to Lakay *et al.* (1988) a dose of 2 g $Cl_2.kg^{-1}$ MLSS.d^{-1} in a laboratory-scale plant did not control bulking within two days, but a dose of 4 g $Cl_2.kg^{-1}$ MLSS.d^{-1} brought the DSVI under control within 6 days.The chlorine was dosed at a point between the anoxic and aerobic zones of the reactor. At this point the chlorine demand will be higher than at the end of the aerobic zone where only a small fraction of the oxidizeable organics present would be left over. This could explain the slower reponse observed. During a period of 19 days at 8 g $Cl_2.kg^{-1}$ MLSS.d^{-1} in the laboratory-scale plant, the DSVI decreased from 204 to 51 $m\ell.g^{-1}$, but the sludge was bleached to a light brown colour and foam accumulated on the reactor surface. These signs of overdosing were not observed in the full-scale plant at a dose of 3,07 g $Cl_2.kg^{-1}$ MLSS.d^{-1}.

4.1.4 SUMMARY

Each of the three oxidants investigated, influenced the settleability of the activated sludge to some degree. At a dose of 9,5 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} a temporary decrease in the DSVI of the activated sludge in the treated plant was measured. At the lower dose of 1,5 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} a temporary lowering in the rate of increase in the DSVI only was observed. Increasing the dose to 3,0 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} lowered the rate

at which the DSVI was increasing also slightly only. Ozone at a dose of 1,42 g $O_3.kg^{-1}$ MLSS.d^{-1} gave some degree of bulking control. This was more noticeable in the SSVI than in the DSVI. The dose of 1,42 g $O_3.kg^{-1}$ MLSS.d^{-1} was probably too low to give conclusive results. However, the results obtained justifiy further investigation on full-scale at higher doses. Chlorine at a dose of 1,42 g $CL_2.kg^{-1}$ MLSS.d^{-1} controlled bulking within 10 days, but could not achieve the target DSVI of 100 $ml.g^{-1}$. At a dose of 3,07 g $Cl_2.kg^{-1}$ MLSS.d^{-1} bulking control was obtained in 4 days and the target DSVI was reached in 14 days.

4.2 The effect of oxidants on the filaments population

A Zeiss phase contrast microscope at a magnification of 400X was used for a study of the filamentous organisms. Identification was done on visual morphological properties using the procedure of Jenkins *et al.* (1986). A subjective scoring of filament abundance was made using the scale suggested by Jenkins *et al.* (1986).

4.2.1 The effect of hydrogen peroxide on the filaments

Before the H_2O_2 treatment the dominant filamentous organism in both reactors was Type 0092 followed by *M. parvicella* and Types 0041 and 021N. The flocs were of medium size, irregular with no free cells in the bulk of the solution. Most of the filaments were present inside the flocs. At the end of the experiment Type 0092 was still the dominant filament in both plants. However in the experimental plant there were fewer *M. parvicella* and Types 0041 and 021N present. Of the four dominant filaments present only type 021N possesses a sheath. The action of H_2O_2 on the filamentous organisms is reported to be one of attacking the sheath, thus destroying the filamentous form (Caropresso *et al.*, 1974). If this is the case, H_2O_2 will have little effect on Types 0092 and 0042, and *M. parvicella* but it will destroy Type 021N, which was, however, present in relatively low numbers only.

The flocs in the control plant were well developed with most of the filaments inside the flocs, but with some protruding from the flocs. Small diffused flocs were formed in the experimental plant. The experimental plant contained visually fewer filaments, however this was apparently not enough to improve sludge settleability. No physical damage to the filaments was observed. In both plants there were no free cells in the bulk of the solution.

Although the H_2O_2 treatment did not improve sludge settleability, it did influence the filamentous population in the treated plant. *M. parvicella* is a low DO filament. Steps that increase the available oxygen in a reactor will, inhibit the growth of *M. parvicella*. When H_2O_2 decomposes 0,47 mg O_2 per mg H_2O_2 is released, this

will help to meet the oxygen demand in the first part of the reactor. The impact of the H_2O_2 on the *M.parvicella* may make it worthwhile as a possible chemical agent in combatting foaming in full-scale plants.

The temporary improvement in the settleability of the activated sludge during the dosing of 9,5 g H_2O_2.kg^{-1} MLSS.d^{-1} may be due to the additional oxygen introduced by the decomposition of H_2O_2 in the first aeration compartment. A temporary improvement in sludge settling and lowering in the number of *M. parvicella* filaments present were also observed during the dosing of pure oxygen into the first aeration compartment to alleviate the oxygen deficiency (unpublished results).

4.2.2 The effect of ozone on the filaments

At the start of the experimental period the dominant filament present was Type 0092. Some *M. parvicella* and a few Type 0041, Type 1701 and Type 0961 were present. After ozonation the same filaments were still present but at slightly reduced numbers. All the filaments were inside the flocs.

In the ozonated plant both large and small flocs were observed while in the control plant there were only large flocs. This indicates that shorter filaments were formed in the ozonated activated sludge system due to the oxidation/disinfection action of ozone compared to the unozonated control system.

No visual damage to the filaments was observed.

4.2.3 The effect of chlorine on the filaments

Dosing chlorine at an overall mass dose of 1,42 g Cl_2.kg^{-1} MLSS.d^{-1} resulted in a visual reduction of the number of filaments as well as a reduction in the floc sizes. The formation of smaller flocs is an indication that the length of the filaments was reduced. No breakup or physical damage to the filaments was visible, however. According to Jenkins *et al.* (1986) physical damage to filaments can precede observeable improvements in activated sludge settling properties and can be used as early signals that chlorination is beginning to exert control over filamentous growth. At the low dose applied, however, this was not the case. The settleability of the sludge improved without physical damage to the filaments. The composition and order of dominance did not change during chlorination. No changes were observed in the control plant.

Before chlorination at 3,07 g Cl_2.kg^{-1} MLSS.d^{-1}, only large diffused flocs were present. Although most of the filaments were inside the flocs, some bridging was observed. Type 0092 was abundant and *Micro parvicella* and Type 0042 were very common. Only a few Type 1701 and Type 021N were present. During

the initial stage of chlorination the only change that was observed was the formation of smaller flocs. After 3 days of chlorination a decrease in the number of filaments was noticeable. Type 0092 was still abundant but less than at the start of the chlorination; only a few of the other filaments were present. Physical damage to the filaments was discernible. Smaller flocs started to appear. This trend continued for the first 8 days of chlorination. On the eighth day larger firmer flocs started to form.

Two days after chlorination was stopped Type 0092 and *M. parvicella* started to increase. Floc sizes increased, but the flocs were more compact. It is clear that bulking control by chlorination is temporary only.

During the same period, no changes were observed in the control plant.

4.2.4 SUMMARY

Hydrogen peroxide treatment caused the formation of small diffused flocs and a reduction in the number of filaments and in particular *M. parvicella*. During peroxide treatment no physical damage to the filaments was visible. In the ozonated plant large and small flocs were formed, an indirect indication that shorter filaments were formed during ozonation. No visual damage to the filaments was observed. A dose of 1.42 g $Cl_2.kg^{-1}$ MLSS.d^{-1} resulted in a reduction of the number of filaments and in the floc sizes. No physical damage to the filaments was visible. At a dose of 3,07 g $Cl_2.kg^{-1}$ MLSS.d^{-1} smaller flocs were formed initially and after chlorination for 8 days larger firmer flocs started to form. Within 3 days of chlorination, a decrease in the number of filaments was noticeable. Physical damage to the flocs was detected.

4.3 The effect of oxidants on nutrient removal

The main objective of this investigation was to determine the effect of oxidants on sludge bulking. However, since the interest was the control of sludge bulking in nutrient removal systems the effect of the oxidants on nutrient removal, cannot be ignored.

Phosphate, nitrate-N and ammonium-N were monitored by means of on-line analyzers. To smooth out sharp daily variations, the 5-day moving average of the daily average of hourly samples was plotted as a function of time.

4.3.1 The effect of oxidants on biological phosphate removal

The biological phosphate removal process is not very stable and is easily upset by changes in the influent quality, equipment breakdown and the introduction of chemicals into the reactor. Disturbances such as a rapid decrease in temperature and the inflow of rainwater, causing a hydraulic shock and/or dilution of the wastewater, may also cause serious damage to plant performanance. In bench scale experiments, some degree of inhibition of biological removal was observed during bulking control trials (Lakay *et al.*, 1988)

4.3.1.1 The effect of hydrogen peroxide on biological phosphorus removal

The phosphate in the effluent of the plants during the dosing of H_2O_2 at low dosages of 3 and 6,34 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} (4,25 kg.h^{-1} and 8,39 kg.h respectively) into the first and final aeration compartments is shown in Figure 13.

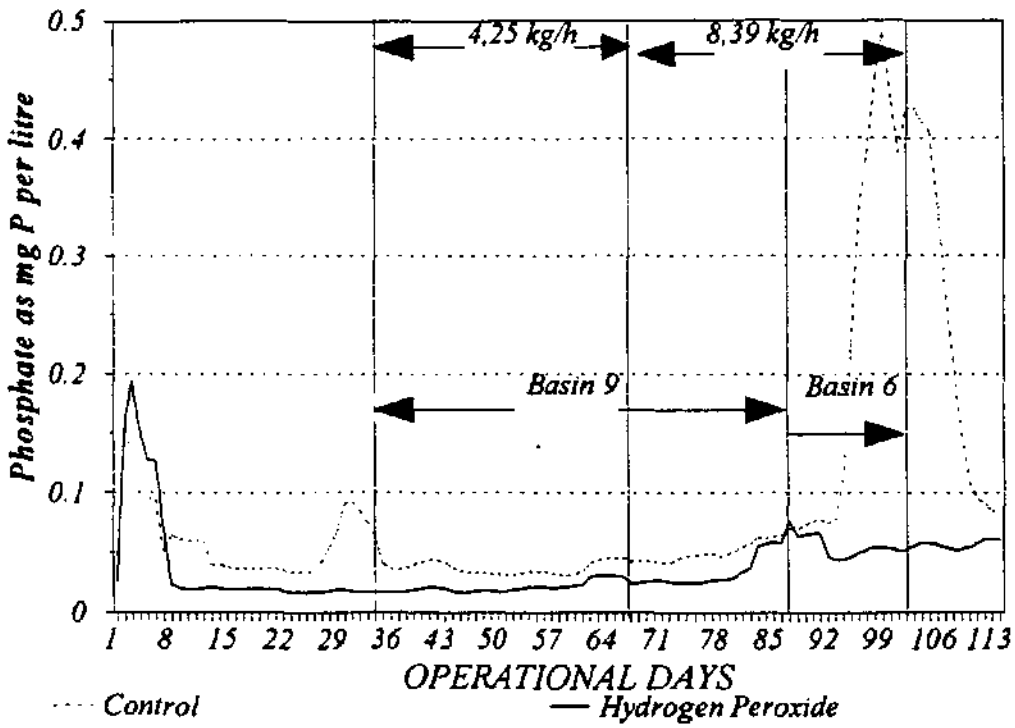

Figure 13 The effect of H_2O_2 at dosing rates of 3 and 6,34 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} into the first and last aeration compartement on phosphate removal.

Dosing H_2O_2 at these concentrations into the first and final aeration compartments did not affect the phosphate removal. On day 94 the phosphate removal in the control plant deteriorated slightly. This was due to the fact that the mixer in basin 2, the last anaerobic basin, broke down causing a buildup of activated sludge in the anaerobic zone of the reactor. Phosphate removal recovered within 3 days after this mixer was

replaced. In this full-scale plant where more than 98 % of the incoming ortho-phosphate is removed, even the slightest decrease in phosphate removal will be noticeable. This illustrates how easily optimum biological phosphate removal in an activated sludge system can be lost.

When H_2O_2 was dosed at a rate of 9,5 g H_2O .kg^{-1} MLSS.d^{-1} (26,67 kg.h^{-1}) into the first aeration compartment, the effect on ortho-phosphate in the effluent of the plants is shown in Figure 14.

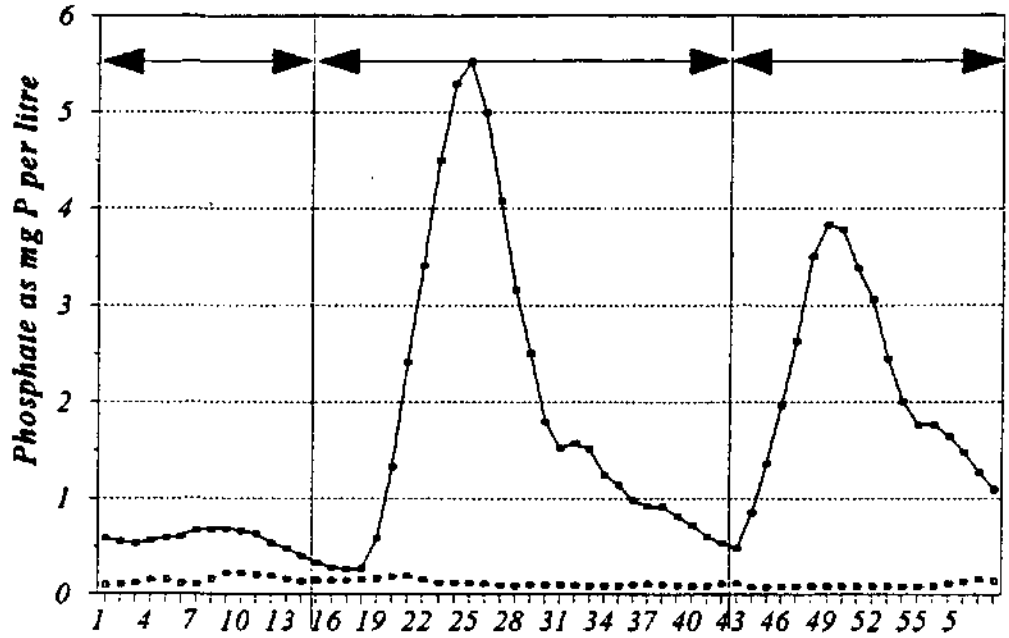

Figure 14 The effect of dosing 9,5 g H_2O_2.kg^{-1} MLSS.d^{-} into the first aeration compartment on phosphate removal.

Before the addition of hydrogen peroxide, the ortho-phosphate concentrations in the effluent of both plants were below 1 mg P .l^{-1} (Figure 14). Four days (operational day 19) after starting the peroxide treatment , at 9,5 g H_2O_2.kg MLSS.d^{-1}, the ortho-phosphate in the experimental plant began to increase. It reached a maximum of 5,80 mg P. l^{-1} after 10 days. Then it decreased to below 1 mg P .l^{-1} after 21 days. This slow rate of change may be indicative of the destruction of the phosphate removing organisms and a regrowth of an acclimatised population. On removal of the H_2O_2 (from the environment) the phosphate removal decreased again but started to recover after 7 days. Van Leeuwen (1992) reported, however, that even in massive doses of 100 g H_2O_2.kg^{-1} MLSS.d^{-1}, into the aeration "chamber", the nutrient removal organisms were not affected. Details of his bench scale experiment is not available. When hydrogen peroxide decomposes, each gram of hydrogen peroxide yields 0,47 gram oxygen. The withdrawal of this additional oxygen from a part of the reactor where the oxygen concentration is normally low, will upset phosphate removal temporarily until the organisms adjust to the new conditions.

4.3.1.2 The effect of ozone on biological phosphate removal

The ortho-phosphate concentrations measured in the effluent of the plants during ozonation are shown in Figure 15.

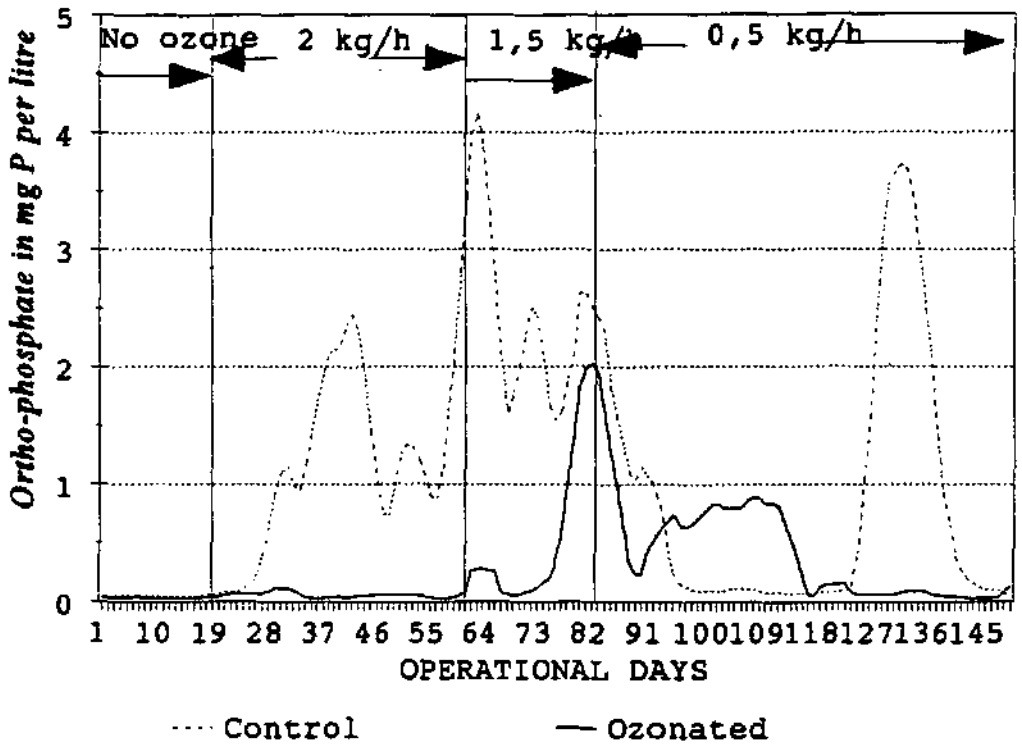

Figure 15 The effect of ozone on biological phosphate removal.

While the phosphate concentration in the effluent of the unozonated control exceeded the 1 mg $P.l^{-1}$ limit, that of the ozonated system was below the limit most of the time, at an overall mass dose of 1,42 g $O_3.kg^{-1}$ MLSS.d^{-1}. At a dosage of 1,07 g $O_3.kg^{-1}$ MLSS.d^{-1} the limit in the ozonated activated sludge system was exceeded from day 77 to day 85 only. During the same period the unozonated control exceeded the limit most of the time. Even at the low dose of 0,36 g $O_3.kg^{-1}$ MLSS.d^{-1} the phosphate in the effluent was below 1 mg $.l^{-1}$, while that in the control system exceeded this limit. It appears as if ozone stabilised phosphate removal. Van Leeuwen (1988) reported that ozone at dosages of 2, 4 and 6 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} phosphate removal was marginally enhanced by ozonation in a pilot plant, but not sufficiently to comply with a limiting concentration of 1 mg $P.l^{-1}$.

4.3.1.3 The effect of chlorine on biological phosphate removal

The ortho-phosphate concentrations measured in the effluent of the plants are shown in Figures 16 and 17.

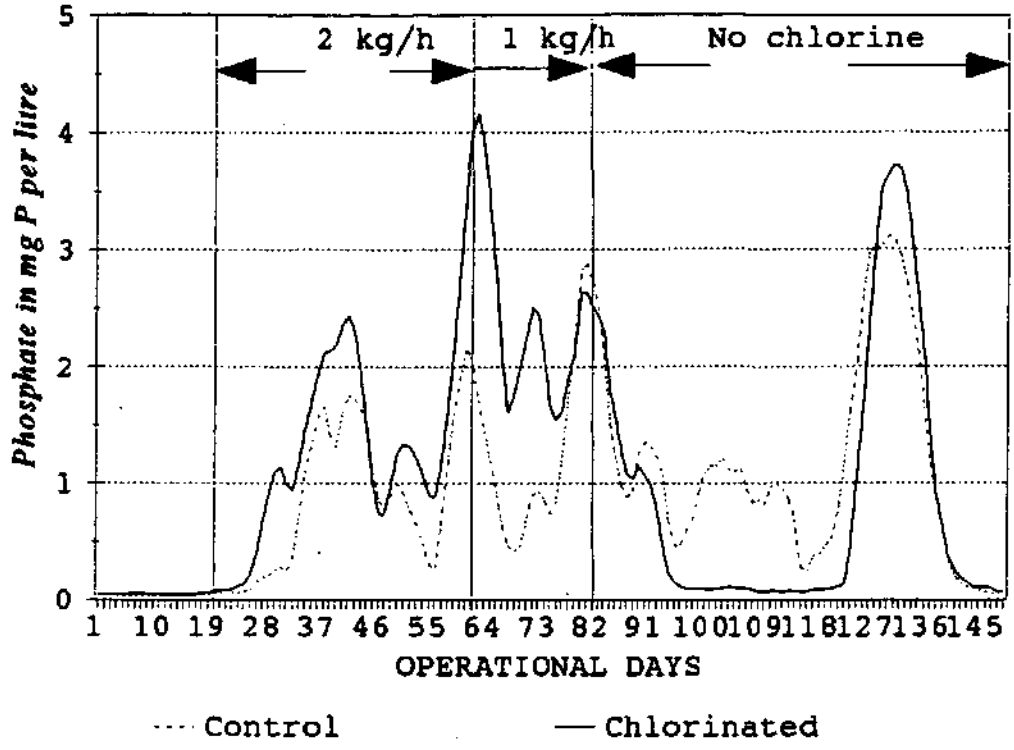

Figure 16 The effect of 0,71 and 1,42 g $Cl_2.kg^{-1}$ MLSS.d^{-1} (1 kg and 2 kg $Cl_2.h^{-1}$ respectively) on biological phosphate removal in activated sludge.

Before the addition of chlorine both plants removed more than 99 % of the incoming ortho-phospate. The average ortho-phosphate in the effluent of the control was 0,04 mg P.l^{-1} and in the experimental plant 0,05 mg P.l^{-1}. After four days of chlorination at a rate of 1,42 g $Cl_2.kg^{-1}$ MLSS.d^{-1} (2 $kg.h^{-1}$), operational day 24, the phosphate removal in both plants started to deteriorate. The increase in the ortho-phosphate in the effluent of the chlorinated activated sludge was marginally higher than that of the control system. During the period in which the chlorine dose was maintained at 1,42 g $Cl_2.kg^{-1}$ MLSS.d^{-1}, the average ortho-phosphate in the effluent of the chlorinated system was 1,44 mg P.l^{-1} and in the control system 0,92 mg P.l^{-1}. Lowering the chlorine dose to 0,71 g $Cl_2.kg^{-1}$ MLSS.d^{-1} did not result in an improvement in the phosphate removal. The phosphate removal deteriorated even further. During this period, the average ortho-phosphate in the chlorinated system increased to 2,18 mg.l^{-1} and in the control plant to 1,38 mg.l^{-1}. It was, therefore, decided to terminate the chlorine dosing. On termination of the chlorination the phosphate removal in both plants recovered. The activated sludge in the chlorinated plant showed the same degree of phosphate removal as before. On operational day 125, the phosphate removal in both plants showed a decline. This cause for this decline was indentified as a low COD : P ratio.

The effect of chlorine at a dose of 3,07 g $Cl_2.kg^{-1}$ MLSS.d^{-1} (4,33 $kg.h^{-1}$) is shown in Figure 17.

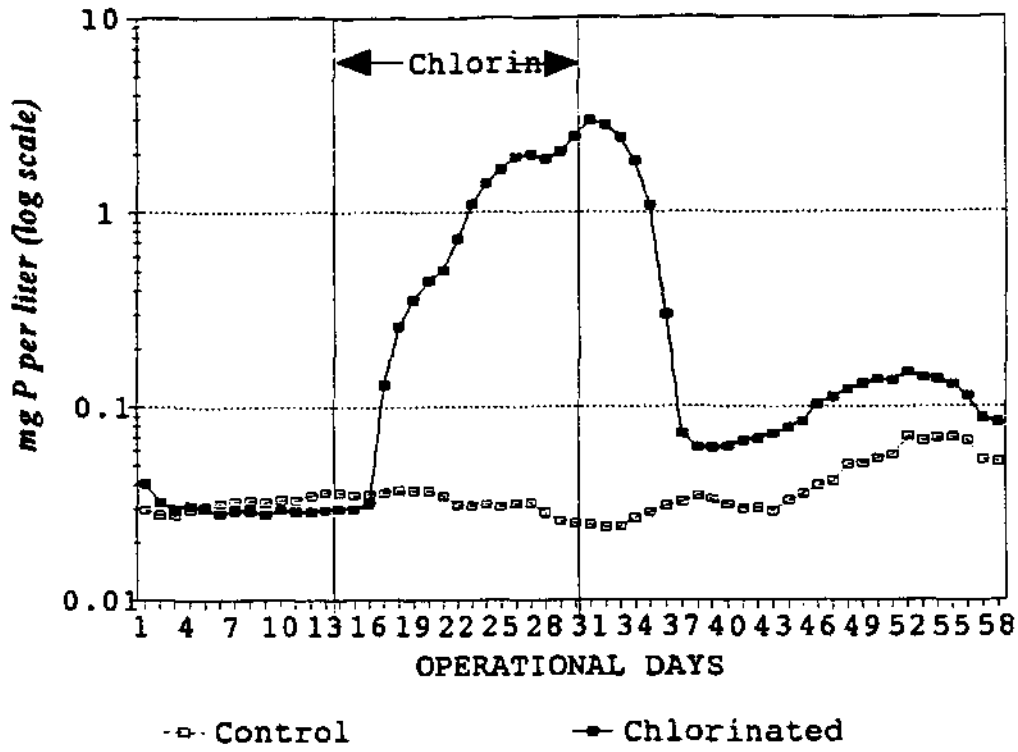

Figure 17 The effect of 3,07 g $Cl_2.kg^{-1}$ MLSS.d^{-1} (4,33 kg $Cl_2.h^{-1}$) on biological phosphate removal in activated sludge.

Chlorination started on operational day 15. On day 17 a slight drop in phosphate removal in the experimental plant was observed. On day 23, the daily average ortho-phosphate in the effluent of the chlorinated system exceeded 1 mg.l^{-1}. While the chlorination was maintained the phosphate values increased, reaching a maximum value on the last day of chlorination. On termination of the chlorination on day 31 phosphate removal showed a rapid recovery. On day 35 the phosphate was below 1 mg P.l^{-1} and on day 37 below 0,1 mg.l^{-1}.

From the diurnal variation in the ortho-phosphate in the effluent, Figure 18, it is clear that some degree of biological P-removal was still achieved during chlorination. The phosphate in the effluent showed a maximum value of 4,98 mg P.l^{-1} at midnight and a minimum value of 0,51 mg/l . This pattern in the phosphate removal corresponded with the diurnal variation in influent load.

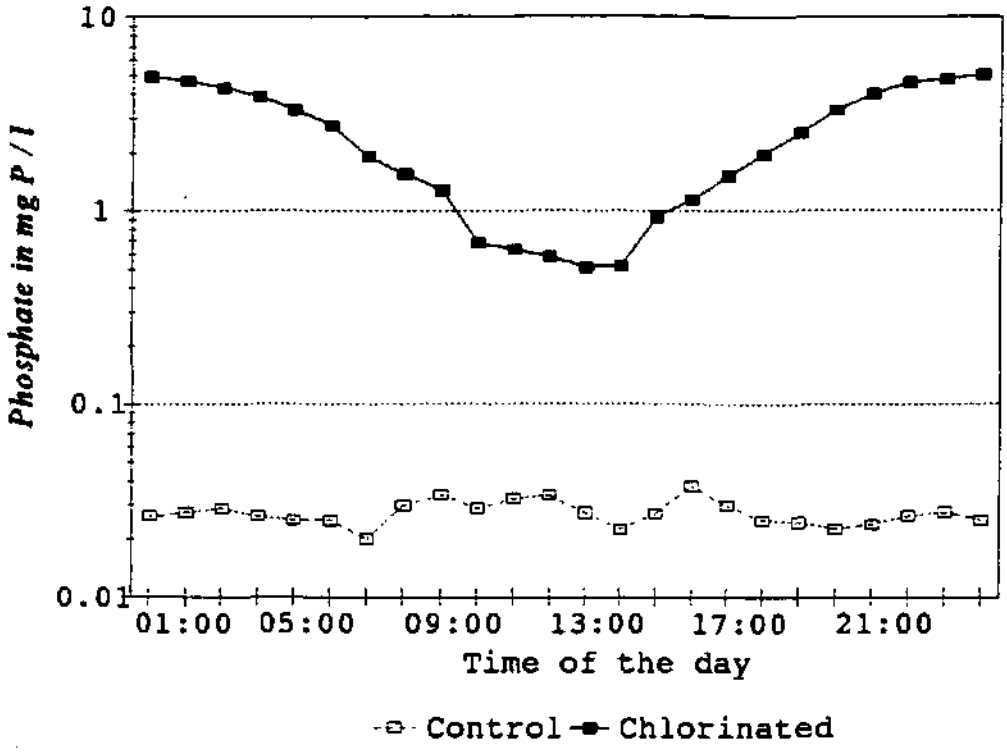

Figure 18 The effect of 3,07 g $Cl_2.kg^{-1}$ MLSS.d^{-1} (4,33 kg $Cl_2.h^{-1}$) on the diurnal variation in ortho-phosphate values.

Lakay *et al.* (1988) report that a dose of 2 g $Cl_2.kg^{-1}$ MLSS.d^{-1} did not affect P-removal in a laboratory-scale nutrient removal activated sludge system. This dose was, however, maintained for two days only. At 4 g $Cl_2.kg^{-1}$ MLSS.d^{-1} P removal declined initially and then recovered. After 6 days, the chlorine dose was increased to 8 g $Cl_2.kg^{-1}$ MLSS.d^{-1}. At this dose, a sharp drop in P-removal was observed followed by a recovery, even after an overdose of chlorine. After 17 days a sharp drop in P-removal is reported. Chlorination was terminated after 19 days. In the full-scale plant the dose of 1,42 g $Cl_2.kg^{-1}$ MLSS.d^{-1}did not affect P-removal significantly. At this dose the trend in the ortho-phosphate levels in the chlorinated plant show a daily increase of 0,031±0,007 mg P.l^{-1},against an increase of 0,024±0,006 mg P l^{-1} in the control plant. At a dose of 3,07 g $Cl_2.kg^{-1}$ MLSS.d^{-1} the average P removal dropped from 98 % to 66 %. No recovery of P-removal was observed during chlorination. P-removal recovered rapidly after chlorine addition was stopped.

4.3.2 Summary

All the oxidants investigated affected biological phosphate removal in some way or the other. At low doses of 3 and 6,34 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} hydrogen peroxide did not affected P-removal, but at a dose of 9,5 g $H_2O_2.kg^{-1}$ MLSS.d^{-1}P-removal deteriorated during the first 10 days. However, while hydrogen peroxide

was being dosed P-removal recovered. On cessation of the hydrogen peroxide treatment phosphate removal deteriorated again. This may mean that in the presence of H_2O_2 a population of P-removing organisms was established that cannot operate in the absence of hydrogen peroxide. The temporary effect of H_2O_2 on sludge settleability could have been due to the establishment of organisms that can decompose hydrogen peroxide. However, this reaction of the P-removing organisms on the removal of the peroxide rule out this posibility. If the H_2O_2 survived long enough in the bulk of the liquid to affect the P-removing organisms, which are protected inside the flocs, it should affect the filaments protruding from the flocs to a larger extent. Ozone at a dose of 1,42 g $O_3.kg^{-1}$ MLSS.d^{-1}, appears to stabilise and promote phosphate removal. Chlorine affected phosphate removal negatively depending on the dosing rate. At 1,42 g $Cl_2.kg^{-1}$ MLSS.d^{-1} phosphate removal in the chlorinated plant was only marginally lower than in the control plant. If P-removal in the plant was at an optimum, it would have been possible to achieve the 1 mg P.l^{-1} limit. At a dose of 3,07 g $Cl_2.kg^{-1}$ MLSS.d^{-1}, P-removal deteriorated to such an extent that the 1 mg P.l^{-1} limit was exceeded. From the diurnal variation in the phosphate concentration and the rapid recovery after chlorination was stopped, it is clear that P-removal was inhibited only.

4.4 The effect of oxidants on nitrification and denitrification

Nitrification is the conversion of ammonia nitrogen (NH_4^+-N) to nitrate (NO_3^--N). The major nitrifying bacteria have been identified as belonging to the genera *Nitrosomonas* (responsible for the oxidation of ammonium to nitrite) and *Nitrobacter* (responsible for the oxidation of nitrite to nitrate). In an activated sludge system, the nitrifying bacteria may comprise only about 5 % of the total biomass and a large proportion of the nitrifying organisms would be found inside the floc (Gerber, 1990), where they should be protected against the action of oxidants. However, they have earned a reputation for being more susceptible to toxic chemicals than other organisms in activated sludge. Denitrification is the transformation of nitrates to gaseous nitrogen. The ability to bring about denitrification is a characteristic of a wide variety of common facultative bacteria including the genera *Pseudomonas, Achromobacter* and *Bacillus*. Because of the large variety of organisms involved the chances of maintaining denitrification in the presence of oxidants should the better than for nitrification.

4.4.1 The effect of hydrogen peroxide on nitrification and denitrification.

The 5-day moving average of the daily average concentration of ammonium-N and nitrate-N, measured in the effluent of the plants are plotted as a function of time in Figures 19 and 20.

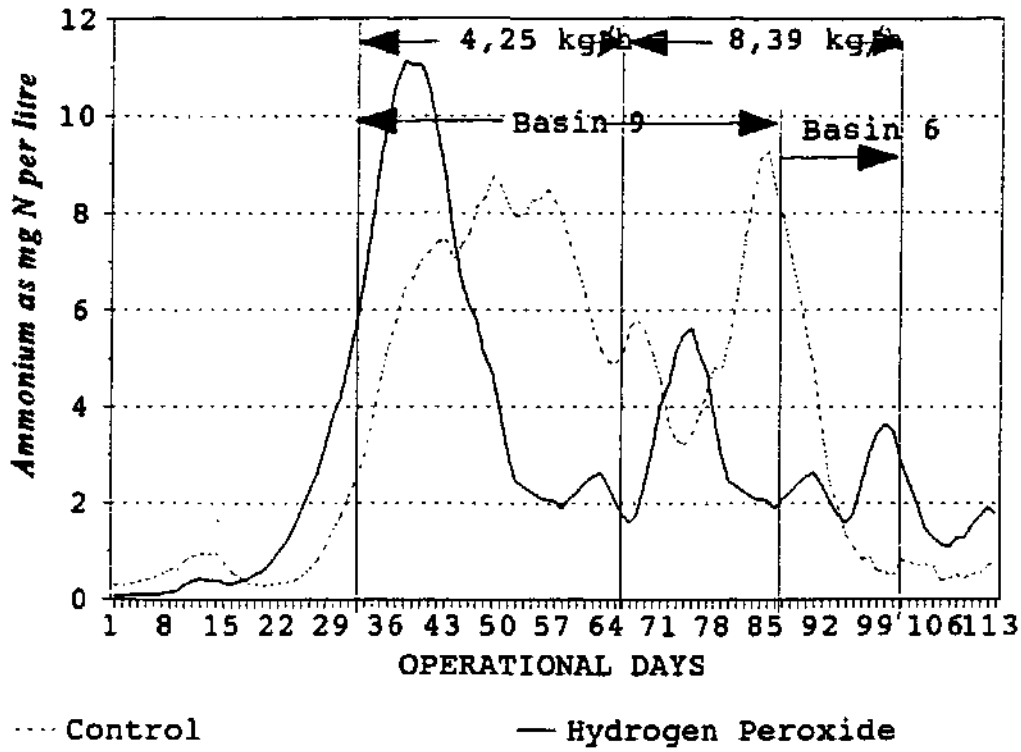

Figure 19 The effect of 1,5 and 3,0 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} on nitrification in activated sludge.

During the period before adding hydrogen peroxide nitrification in both the control and the experimental plant was upset. Nitrification in the plant treated with hydrogen peroxide recovered marginally faster than that in the control plant. After ca 10 days the nitrification in the plant treated with 1,5 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} was better than nitrification in the control plant. Increasing the dose to 3,0 g $H_2O_2.kg^{-1}$ MLSS.d^{-1}, however did not improve nitrification. A factor that influences nitrification to a large extent, is the dissolved oxygen concentration (DO) inside the floc, where the DO is consumed. The DO inside the floc will depend on the size of the floc. In the peroxide treated plant, small diffused flocs were formed while the flocs in the control plant were well developed. The difference in nitrification observed could be the result of different floc sizes. It is, therefore, clear that the peroxide affected nitrification indirectly only.

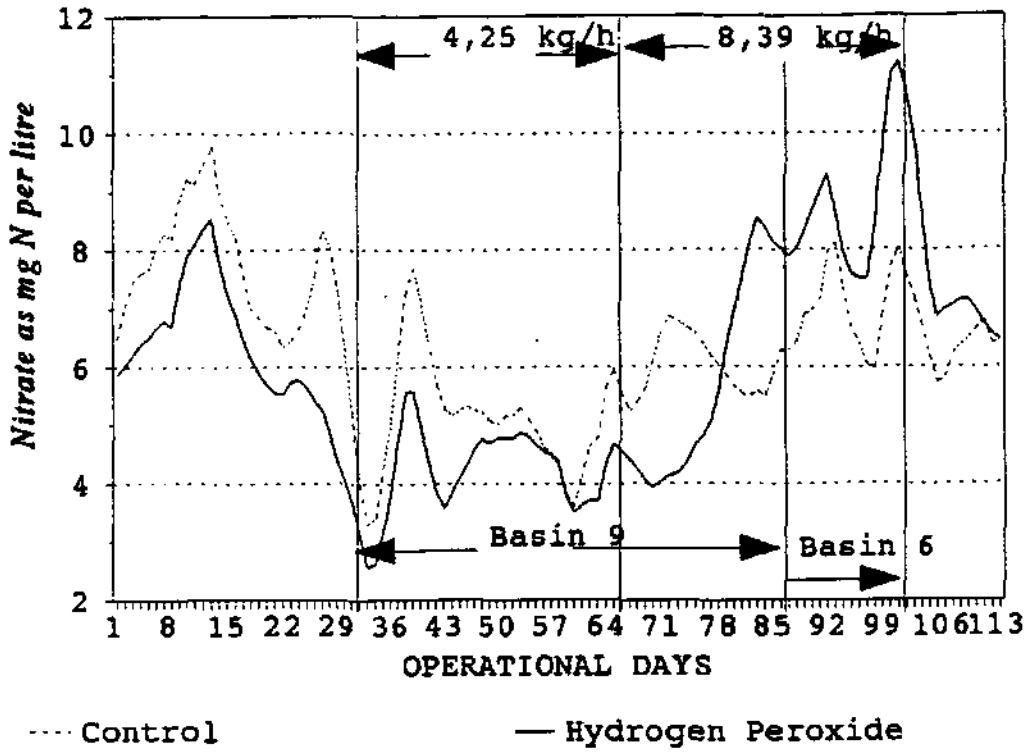

Figure 20 The effect of 1,5 and 3,0 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} on denitrification in activated sludge.

The average nitrate concentration and the standard deviation is shown in Table 6.

Table 6 The average nitrate-N during peroxide treatment of activated sludge in a biological nutrient removal system with H_2O_2.

	Control plant		Peroxide treated plant	
	Average	Std. deviation	Average	Std. deviation
Before treatment	6,7	1,3	5,7	1,9
Basin 9: 1,5 g $H_2O_2.kg^{-1}$ MLSS.d^{-1}	6,5	1,2	4,4	1,0
Basin 9: 3,0 g $H_2O_2.kg^{-1}$ MLSS.d^{-1}	7,0	1,0	6,2	1,8
Basin 6: 3,0 g $H_2O_2.kg^{-1}$ MLSS.d^{-1}	6,0	0,5	9,1	2,0
After treatment	5,6	1,0	6,8	0,5

Although the daily nitrate values varied between the two plants, the variance in the daily values, measured by the standard deviation, is so large that the difference was not significant.

Figures 21 and 22 show the daily average (hourly samples) ammonium-N and nitrate-N in the effluent of the plants during the dosing of H_2O_2 at 9,5 g $H_2O_2.kg^{-1}$ MLSS.d

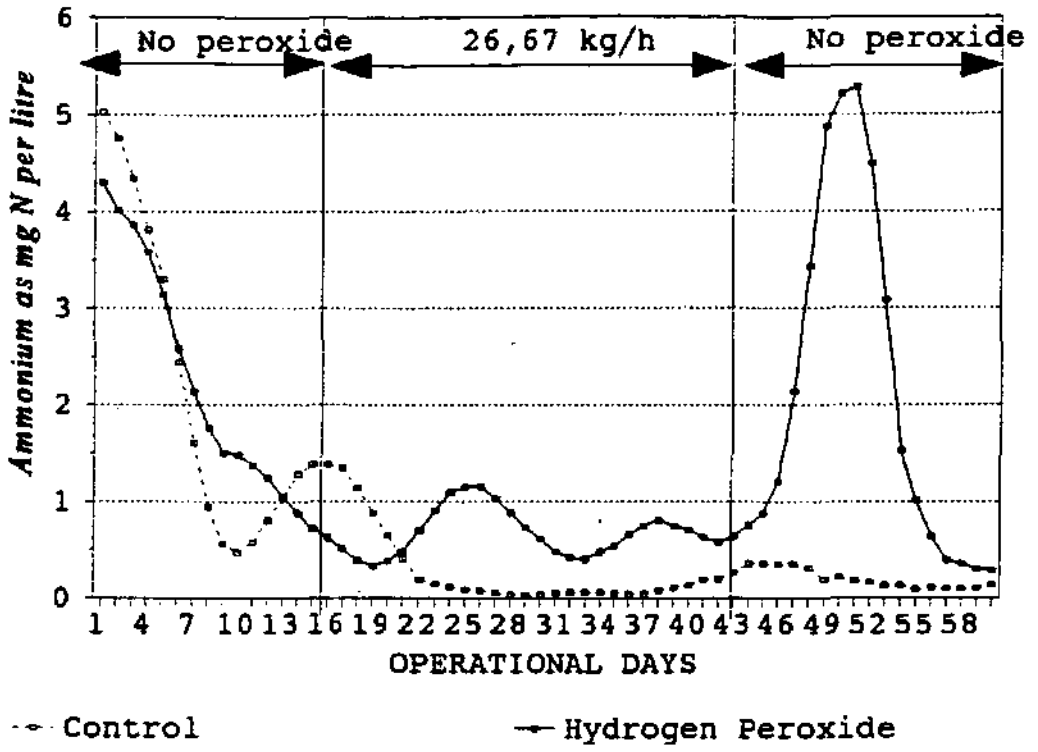

Figure 21 The effect of 9,5 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} on nitrification in activated sludge.

Hydrogen peroxide at the high dose of 9,5 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} led to a slightly lower nitrification rate indicated by consistent higher saline ammonium-nitrogen (Figure 21) in the effluent of the treated plant. On cessation of the peroxide treatment, nitrification in the experimental plant deteriorated temporarily. This lowering in nitrification may be due to the sudden decrease in available oxygen, caused by the withdrawal of hydrogen peroxide.

Denitrification in the plant treated with hydrogen peroxide, was for most of the period lower than denitrification in the control plant. This was the case even before treatment was started. During treatment, the denitrification in the experimental plant was on two occasions better than that in the control. However, the variation in the daily average was so large that the observed differences were not significant.

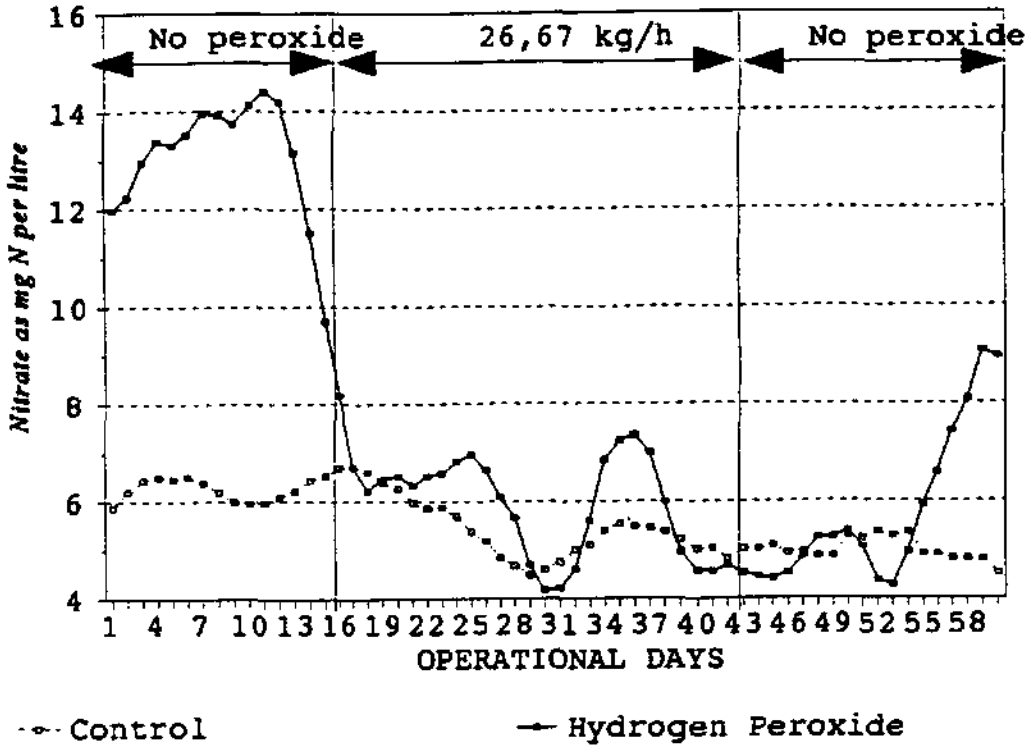

Figure 22 The effect of 9,5 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} on denitrification in activated sludge.

4.4.2 The effect of ozone on nitrification and denitrification

The ammonium-N and nitrate-N concentrations measured during the ozonation of the activated sludge are shown in Figures 23 and 24.

The average nitrate-N and ammonium-N values, and standard deviation in these values, measured during ozonation are presented in Table 7.

The daily average saline ammonia values for the ozonated plant was marginally higher than that for the control plant (Table 7). However, the standard deviation in the values is so large that these differences were insignificant. From day 43 to day 51, the ammonia values in the ozonated plant increased substantially. This increase was in all probability caused by factors other than ozone, because a similar but smaller peak is noticeable in the control plant, however, their effect might have been accentuated by ozone.

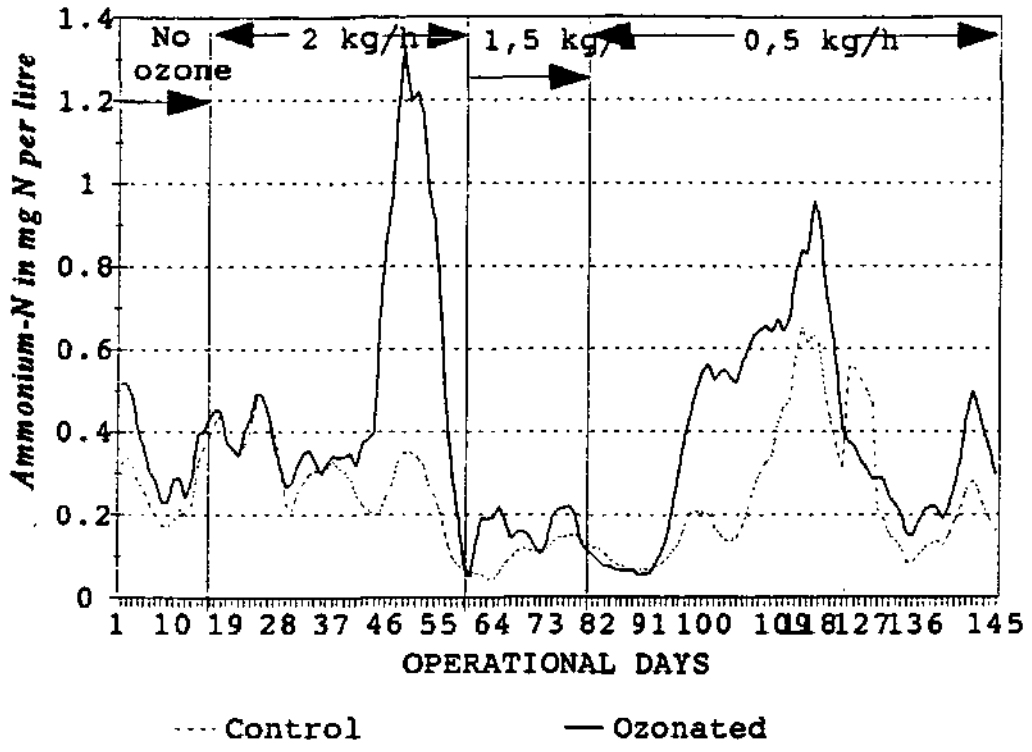

Figure 23 The effect of ozone on nitrification in activated sludge.

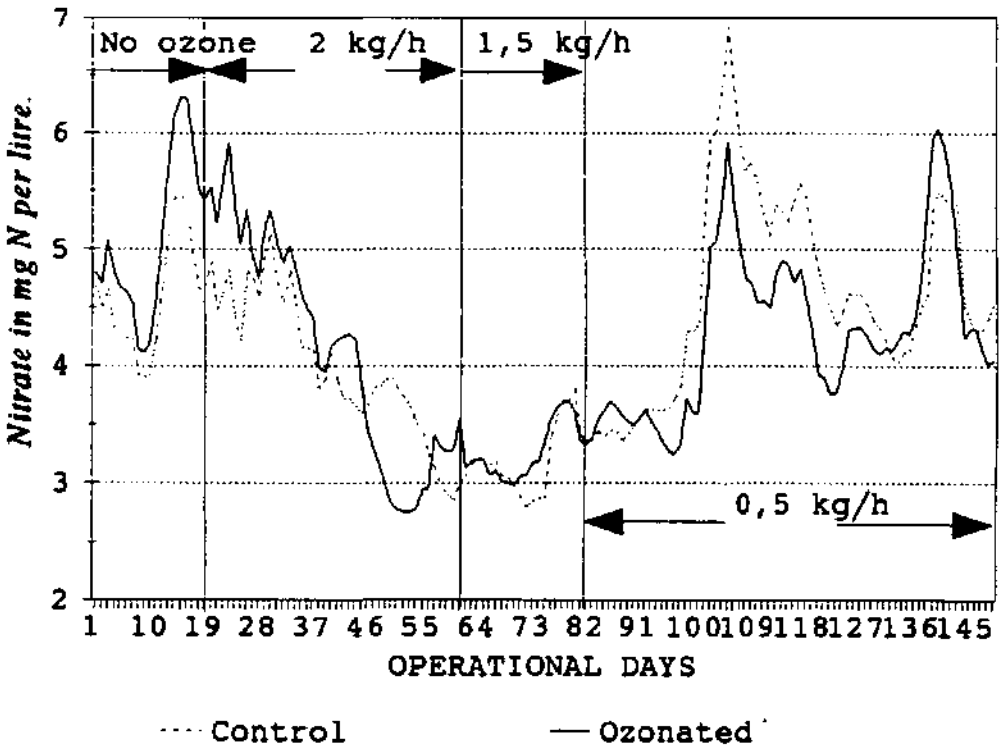

Figure 24 The effect of ozone on denitrification on activated sludge.

Table 7 Average nitrate and saline ammonia concentrations as well as standard deviation as mg $N.l^{-1}$ in the effluent of the biological nutrient removal activated sludge plants during ozonation..

Ozone dose	Nitrate Control		Nitrate Ozonated		Ammonia Control		Ammonia Ozonated	
	Ave.	Std. dev.	Ave.	Std. dev.	Ave.	Std. dev.	Ave.	Std. dev
2,0 $kg.h^{-1}$	3,92	0,84	4,02	1,17	0,25	0,14	0,50	0,40
1,5 $kg.h^{-1}$	3,26	0,49	3,34	0,39	0,11	0,04	0,13	0,10
0,5 $kg.h^{-1}$	4,32	1,08	4,32	1,01	0,25	0,23	0,41	0,26
Overall	4,2	1,08	4,05	1,05	0,23	0,19	0,41	0,32

Without ozone, nitrification in the experimental plant was slightly lower than that in the control plant. With ozone the denitrification in the ozonated plant improved and was at times better than in the control plant. As can be seen from Table 7 the difference between the two plants was, however, insignificant.

Nitrification and denitrification in the plants under investigation were at an optimum. The ammonium-nitrogen in the ozonated plant exceeds the legal limit of 1 mg $N.l^{-1}$ from day 45 to day 51 only. Control of aeration in full-scale plants is not always effective and could have resulted in temporary over- and under-aeration; this may explain the slightly higher ammonuim-nitrogen values observed. Van Leeuwen (1988) reported that ozone at a dose of 3 mg $O_3.l^{-1}$ had little effect on effluent quality. At 6 mg $O_3.l^{-1}$ nitrification was improved in the activated sludge. At 1,74 mg $O_3.l^{-1}$, the maximum attainable dose with the available equipment, one can not expect large improvements in nitrogen removal.

4.4.3 The effect of chlorine on nitrification and denitrification

The effect of chlorine at doses of 1,42 g $Cl_2.kg^{-1}$ MLSS.d^{-1} and 3,07 g $Cl_2.kg^{-1}$ MLSS.d^{-1} on nitrification are shown in Figures 25 and 26 respectively.

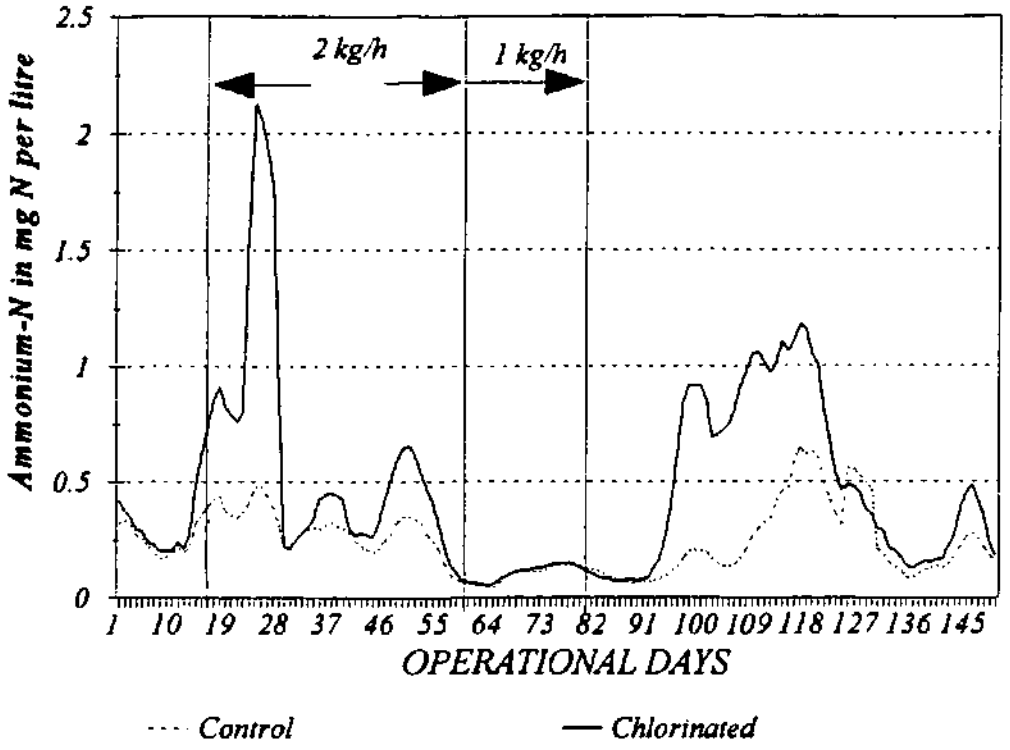

Figure 25 The effect of 0,71 and 1,42 g $Cl_2.kg^{-1}$ MLSS.d^{-1} (1 and 2 kg $Cl_2.h^{-1}$ respectively) on nitrification in activated sludge.

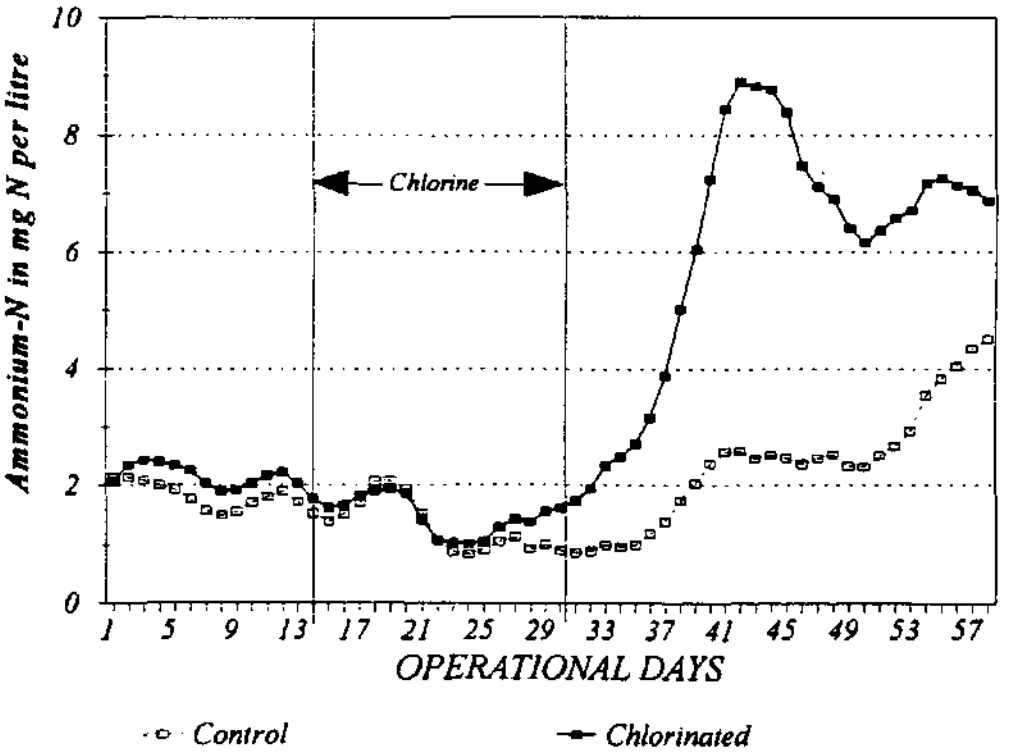

Figure 26 The effect of 3,07 g $Cl_2.kg^{-1}$ MLSS.d^{-1} (4,33 kg.h^{-1}) on nitrification in activated sludge.

Before dosing chlorine, the ammonium-N in the chlorinated system showed an increase. Nitrification is affected by the DO available to the nitrifiers. The concentration of DO inside the floc, where the oxygen consumption takes place, is not necessarily the same than that in the bulk of the liquid. This value depends on the size of the floc, mixing intensity and diffusion rate into the floc.These conditions are difficult to duplicate in plants. Differences in nitrification rates in plants can therefore occur. During the initial stage of the chlorination, this increase continued. It reached a maximum value of ca 2,2 mg $N.l^{-1}$ before it dropped sharply to the same levels as that in the control plant, indicating that nitrification was not affected by the dosage of 1,42 g $Cl_2.kg^{-1}$ MLSS.d^{-1} (2 kg $Cl_2.h^{-1}$).

During the dosing of 3,07 g $Cl_2.kg^{-1}$ MLSS.d^{-1}, the amonium-N concentration in the chlorinated and control plants remained almost the same. After chlorination was stopped, the ammonium-N concentration in the effluent from the chlorinated plant showed an increase. There is no assignable cause for this increase.

The nitrate concentrations measured during the chlorination of the activated sludge at rates of 1,42 g $Cl_2.kg^{-1}$ MLSS.d^{-1} and 3,07 g $Cl_2.kg^{-1}$ MLSS.d^{-1}, are plotted in Figures 27 and 28 respectively.

During the dosing of 0,71 and 1,42 g $Cl_2.kg^{-1}$ MLSS.d^{-1} (1 and 2 kg $Cl_2.h^{-1}$), the daily nitrate values in the effluent of the control and chlorinated activated plants showed small variation (Fig 27). However, the individual values showed such sharp variations that the differences observed are not significant.

For the first 45 days during the second chlorination trial, the nitrate concentration in the chlorinated system was higher than that in the control plant (Fig. 28). Before chlorination, the nitrate-N in the experimental plant was on avarage 1 mg $N.l^{-1}$ higher than that in the control plant. During the first five days of chlorination, the difference increased to ca 4 mg $N.l^{-1}$. Over the following 4 days it decreased to an average of ca 2,5 mg $N.l^{-1}$. This difference was maintained for the rest of the chlorination. During the four days after chlorination was stopped, the difference between the control and chlorinated plant increased to ca 7,5 mg $N.l^{-1}$ before it started to decrease. After chlorination was stopped the measured nitrate-N in the effluent of the chlorinated plant showed a steady decrease until it reached the same levels as that in the control plant. During the entire operational, period the nitrate-N values in both plants followed broadly the same general pattern and it, therefore, appears as if there is no correlation between the chlorination and denitrification.

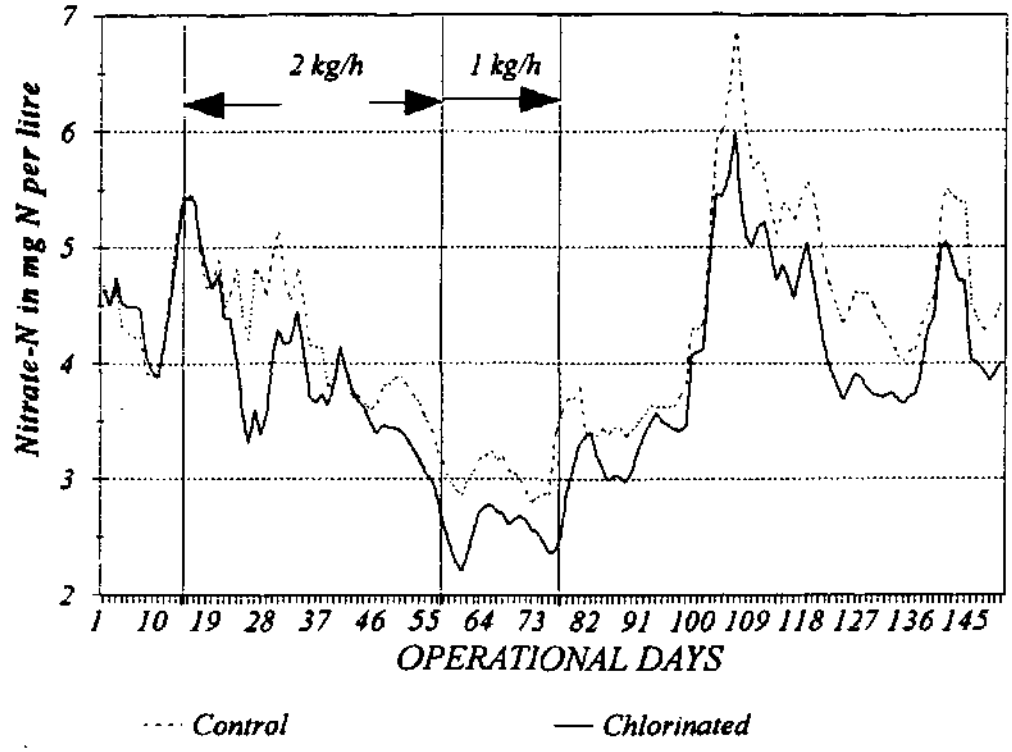

Figure 27 The effect of 0,71 and 1,42 g $Cl_2.kg^{-1}$ MLSS.d^{-1} on denitrification in activated sludge.

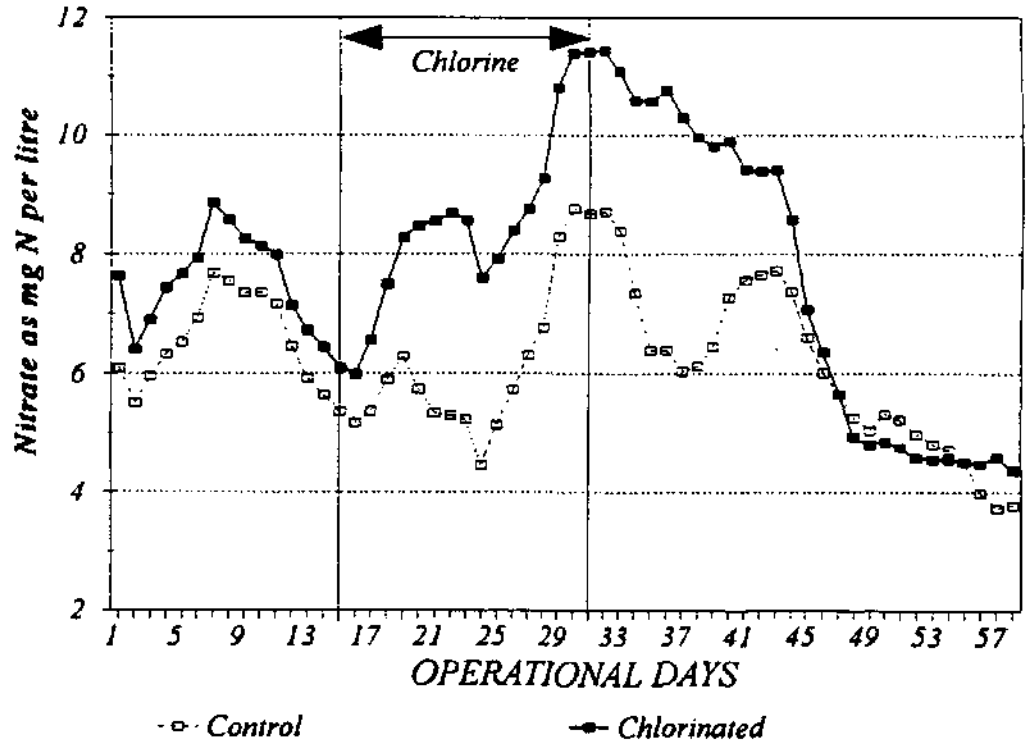

Figure 28 The effect of 3,07 g $Cl_2.kg^{-1}$ MLSS.d^{-1} on denitrification in activated sludge.

4.4.4 SUMMARY

The results obtained in this investigation showed that hydrogen peroxide and ozone influenced nitrification and denitrification marginally only, if at all. Van Leeuwen (1992) reported that hydrogen peroxide at a dose of 100 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} did not affect the "sensitive nutrient removal organisms". Results obtained on pilot scale by Van Leeuwen (1988), showed a marginal increase in nitrification at doses of 4 and 100 g $O_3.kg^{-1}$ MLSS.d^{-1}. At doses of 1 and 2 g $O_3.kg^{-1}$ MLSS.d^{-1} there was no significant difference in the ammonium-N and nitrate-N values between the experimental and control plant.

No correlation was observed between nitrification/dentrification and the chlorine dose. Lakay, *et al.*, (1988) reported an increase in Total Kjeldahl Nitrogen (TKN) due to increases in effluent turbidity. They concluded that nitrification *per se* was not affected at a chlorine dose of 8 g $Cl_2.kg^{-1}$ MLSS.d^{-1}. At this dose, there was also no correlation between the variation in denitrification and the chlorine dose. An accidental overdose of 16 g $Cl_2.kg^{-1}$ MLSS.d^{-1} for 15 hours led to a steady decrease in denitrification. Denitrification recovered after 5 days in the system operated at a sludge age of 21 days.

CHAPTER 5
THE COST IMPLICATIONS OF NON-SPECIFIC BULKING CONTROL MEASURES

The cost implications of using hydrogen peroxide, ozone and chlorine for bulking control, should be considered against the cost implications of building additional secondary settling capacity in existing plants to maintain the designed capacity of the plant or the continuation of the practice of over-designing secondary settling tanks in new plants. Another factor to be considered, is the practice to increase sludge handling and dewatering equipment capacity to cope with the increased volumes of waste activated sludge caused by bulking sludge.

5.1 THE EFFECT OF BULKING SLUDGE ON SECONDARY SETTLING CAPACITIES

The design procedure for secondary settling tanks in activated sludge plants, is largely based on sets of *ad hoc* design rules. Design rules are set down on experience that has been shown to provide adequate results over a range of sludge settling behaviour. Often these rules do not incorporate any sludge settling characteristics directly. One such example, shown in Table 8, is taken from the "Guide to the design of sewage purification works", (Institute for Water Pollution Control, 1973).

Table 8 Secondary settling tank design criteria for activated sludge plants.

Maximum overflow rate at peak wet weather flow	1,0 $m.h^{-1}$
Minimum retention time at peak dry weather flow	1,5 h
Maximum weir loading rate at peak dry weather flow.	8,3 $m^3.h^{-1}.m^{-1}$

Source: Guide to the design of sewage purification works (1973)

In setting these design rules, it was accepted that the sludge settling characteristics will not fall below a certain minimum quality. Ekama and Marais (1986) have shown the that maximum permissible overflow rate for a sludge with a MLSS of 3,5 $g.l^{-1}$ and a DSVI of 150 $ml.g^{-1}$ is 1 $m.h^{-1}$. The DSVI of activated sludge in a nutrient removal plant is influenced by a number of factors, some of which are uncontrollable and can easily

exceed this value at times. This may result in failure of the settling tank and the nutrient removal process.

In the proposal by the Abwasser Technik Verband (AFV) (1973, 1976) (cited by Ekama and Marais 1986), the DSVI has been integrated into secondary settling tank design procedures. In this procedure, the maximum permissible overflow rate is empirically linked to sludge volume. The sludge volume is the product of the DSVI and the biological reactor sludge concentration or MLSS. The permissible overflow rate in given by:

$$Q = 2\,400\,(MLSS \times DSVI)^{-1,34} \text{ subject to } Q < 1,6 \text{ m.h}^{-1}.$$

The effect of DSVI at various sludge concentrations on the permissible overflow rate is shown in Figure 29.

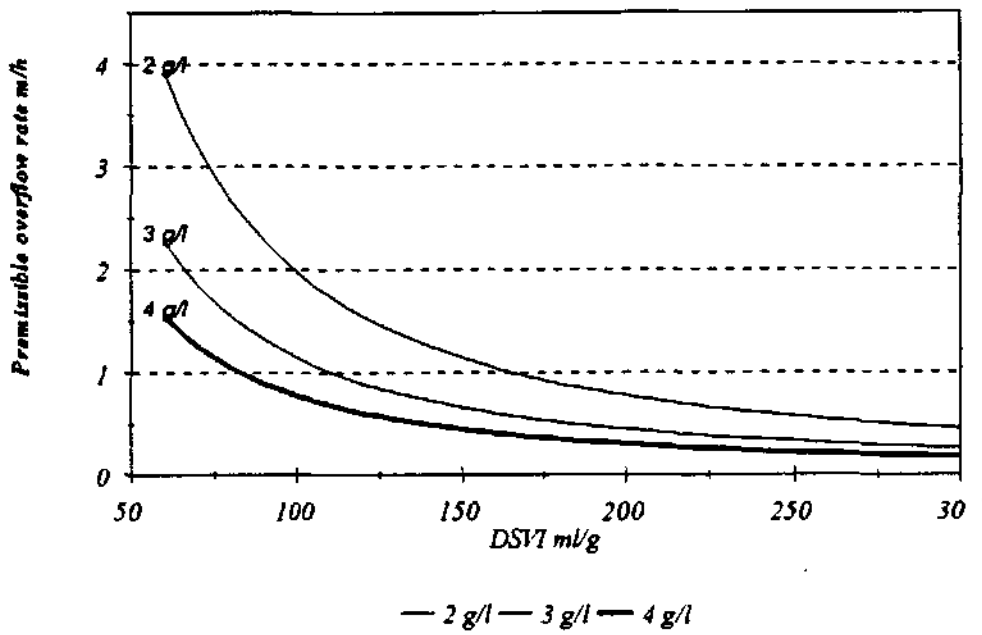

Figure 29 The effect of DSVI on the permissible overflow rate.

The permissible overflow rate function is not based on settling tank failure *per se*, but on limiting the effluent solids concentration to below 30 mg.l^{-1}. However, there is only a small difference in permissible overflow rate for an effluent solids concentration of 30 mg.l^{-1} and settling tank failure.

From Figure 29, it can be seen that if a system is designed to operate at a MLSS of 3 g.l^{-1} and an overflow rate of 1 m.h^{-1}, the tank will not fail, provided that the DSVI is below 110 ml.g^{-1}. If the DSVI increases by 50 % to 165 ml.g^{-1}, the overflow rate must be decreased to 0,59 m.h to maintain biological sludge concentration and the capacity of the biological reactor. The alternative is to decrease the MLSS concentration to 2 g.l^{-1}. To maintain the COD loading rate in the reactor the throughput must be reduced. It is, therefore clear that sludge settleability governs the throughput and COD loading rate in an activated sludge plant.

In a survey of most of the major nutrient removal plants in South Africa, Van Leeuwen (1992) found that virtually all had secondary settling tanks designed with overflow rates well below 1 $m.h^{-1}$. Most of the major plants are equipped with settling tanks about 50 % in excess of the capacity required for non-bulking sludge.

5.2 THE EFFECT OF BULKING SLUDGE ON SETTLING COST

Savings in settling cost are possible if smaller settling capacities could be installed. In the calculation of these savings the following cost figures were used. According to a leading consultant in the field of watercare the current cost of secondary settling tanks for activated sludge are as follows (1994 cost):

A 24 metre settler	R 1 026 000.
A 27 metre settler	R 1 260 000.
A 30 metre settler	R 1 442 000.

The savings have been amortised over a period of 15 years at a discount rate of 4,5 % per annum (the difference between the interest and inflation rates) and expressed as a unit cost saving based on the capacities of the plants. The result of these calculations is shown in Table 9.

Table 9 Cost implications of excessive settling capacity.

Plant capacity ($Ml.d^{-1}$)	Peak wet weather overflow rate $m.h^{-1}$	Settlers	Possible alternative for 1 $m.h^{-1}$	Capital saving R 1 000 000	Saving (cents/m^3)
132*	0,67*	18 x 30 m*	12 x 30 m*	8,65	1.67
20*	0,79*	3 x 27 m*	2 x 30 m*	0,90	1.15
30*	0,79*	4 x 30 m*	3 x 30 m*	1,44	1.22
22.5*	0,66*	3 x 27 m*	2 x 27 m*	1,26	1.43
6*	0,30*	2 x 24 m*	1 x 24 m*	1,03	4.38
18*	0,38*	3 x 30 m*	2 x 24 m*	2,27	3.22

* Source : Van Leeuwen (1992)

The weighted mean savings in amortisation cost, based on throughput, in Table 9 amount to 1,74 $c.m^{-3}$. Due to the fact that there will be fewer settling tanks to operate there will also be a reduction in operating and maintenance cost, this may amount to a further 0,26 $c.m^{-3}$. A total saving of 2 $c.m^{-3}$ is possible if sludge bulking can be controlled.

5.3 THE EFFECT OF BULKING SLUDGE ON SLUDGE THICKENING AND DEWATERING COSTS.

Osborn *et al.* (1986) reported the costs for thickening and dewatering high DSVI (150 - 200 $m\ell.g^{-1}$) and low DSVI (70 - 80 $m\ell.g^{-1}$) sludges. Waste sludge from the activated sludge was thickened by means of dissolved air flotation and dewatered on belt presses.

According to Osborn *et al.* (1986):

- the float concentrations from the dissolved air flotation unit at equivalent loading rates depends on the sludge volume index (SVI) of the sludge. The approximate relationship was

Float concentration (%) = -0,02 X SVI + 6

- the loading rate on the belt press is also dependent on the sludge SVI. At low SVI loading rates of up to 7 $m^3.m^{-1}.h^{-1}$ was possible. At high SVI the loading rate was limited to 3,5 $m^3.m^{-1}.h^{-1}$.

This will affect the capital and operating cost of the sludge handling equipment. According to Osborn *et al.* (1986) the cost of dewatering a tonne of bulking sludge amounted to R 87,30 against R 42,80 for non-bulking sludge. The costs, updated at an escalation rate of 15 % per annum, now (1994) amount to R 307 and R 150 per tonne of bulking and non-bulking sludge respectively.

According to Ekema *et al.* (1984) sludge production relates to the COD of the influent according to a complex function of yield, decay rates and sludge age. The average daily sludge production in the Daspoort plants during this investigation was 1 924 $kg.d^{-1}$ at an average in inflow of 13,26 $M\ell.d^{-1}$. The unit cost to dewater this sludge during bulking would amount to 4,45 $c.m^{-3}$ against 2,18 $c.m^{-1}$ during non-bulking.

According to Van Leeuwen (1992) the cost of land disposal will not be so profoundly affected by bulking sludges.

If bulking in an activated sludge plant can be controlled on a continuous basis, savings of 2 cents in settling cost and 2,27 cents in dewatering cost per cubic metre influent can be achieved. The improvement in nutrient removal cannot be quantified.

5.4 THE COST OF NON-SPECIFIC BULKING CONTROL

The cost of controlling bulking sludge with oxidants will dependent on the concentration of the oxidant necessary to bring bulking under control, the duration of the dosing to effect control and the frequency of treatment required.

5.4.1 Cost of bulking control with hydrogen peroxide

In this investigation in a full-scale nutrient removal plant, no lasting bulking control could be achieved at a dosage of 9,5 g $H_2O_2.kg^{-1}$ MLSS.d^{-1}. Hydrogen peroxide has been used for bulking control in conventional fully aerobic activated sludge processes (Carapresso *et al.*, 1974) and in bench scale trials conducted at the University of Pretoria, it was found that bulking control at a dosage of 8 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} in a nutrient removal activated sludge was possible (Van Leeuwen and Van Rossum, 1990 cited by Van Leeuwen, 1990; Van Leeuwen, 1991). Assuming that the failure to control bulking in the full-scale plant was due to the compartmentalisation of the reactor and/or incorrect selection of the dosing point, the cost implications of using H_2O_2 is calculated on the dose of 9,5 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} at which temporary bulking control was observed.

The current price of hydrogen peroxide is R 3,10 per kilogram of a 50 % solution or R 6,20 per kg pure H_2O_2. Because of the large quantities of hydrogen peroxide that must be handled, bulk storage and delivery are recommended. A bulk storage tank of 20 m^3 is required to ensure continuity of supply. Hydrogen peroxide is highly corrosive and all equipment in contact with it should be made of high density polyethylene or 316 stainless steel. A 20 m^3 stainless steel tank costs approximately R 150 000. A variable-speed drive metering pump and installation will add a further R 40 000 to the capital cost of the installation. This capital outlay amortised over a period of 15 years at a rate of 14,5 % per annum adds an amount of R 31 710 to the yearly expenditure. In the treatment of the experimental plant at Daspoort this will increase the cost of hydrogen peroxide by R 0,27 per kg H_2O_2 (100 %) to R 6,47 per kilogram H_2O_2.

The daily oxidant requirements (in kilograms) for bulking control based on a overall mass dose (g.kg^{-1} MLSS.d^{-1}) are given by

$$T_m \times MLSS \times Q \times \tau + 24$$

where MLSS is in g.l^{-1}

Q is in Ml.d^{-1}

τ is in hours including retention time in the reactor and the settling tank.

In the experimental plant at an average flow of 13,26 $Ml.d^{-1}$ the retention time was 18,91 hours. At a dose of 9,5 $g.kg^{-1}$ MLSS.d^{-1} and MLSS concentration of 3,225 $g.l^{-1}$, 320 kg of pure hydrogen peroxide was used per day. The cost of treating a cubic metre of influent was therefore 15,61 cents.

5.4.2 Cost of bulking control with ozone

Ozone has to be generated on site requiring expensive equipment. According to Van Leeuwen (1990) the main contributor towards the cost of ozonation is amortization of capital. The capital outlay of the plant will depend on the size of the required equipment, it is, therefore, necessary to first determine the size of the ozonizer. The decision on the required ozonizer capacity should not be based on the minimum ozone requirements for bulking control, because from this investigation it is clear that dosing ozone at too low rates will not given the desired degree of bulking control. The loss of ozone due to the transfer rate of the ozone should also be taken into account. Ozone is a powerful oxidant and disinfectant that does not contribute to the salinity of the water treated. Surplus capacity can be employed to disinfect the final effluent of the plant in the place of chlorine.

An ozone generator producing 5 kg $O_3.h^{-1}$ using pure oxygen, is sold for about R 800 000 (1994). Auxiliary equipment including pipework, diffusors and a contact column could increase the capital requirements by a further R 100 000. A building to house the generator can be erected at a cost of R 50 000, bringing the total capital requirement to R 950 000. The annual amortisation cost at a rate of 14,5 % per annum amounts to R 158 551. At a depreciation rate of 4 % per annum, the equipment will have a salvage value of R 360 000 in fifteen years. The uniform annual equivalent present value of this salvage value at an annual rate of 14,5 % will be R 7 882. At a depreciation rate of 2,5 % the building will have a salvage value of R 31 250 in fifteen years. The uniform annual equivalent present value of this salvage value is R 684. The total fixed cost of ozone generation will be R 149 985 per annum.

The variable cost of ozonation consist of oxygen, 10 kg $O_2.kg^{-1}$ O_3 at R 0,40.kg^{-1} O_2, hiring of a oxygen storage vessel and evaporator at R 3 000 per month, electricity cost, 8,5 $kWh.kg^{-1}$ O_3 at R 0,12.kWh^{-1} and maintenance costs of 2,5 % on the equipment and 1,5 % on the building.

The total cost per annum to generate 43 200 kg O_3 will be as follows:

Fixed cost		R 149 985
Oxygen	432 000 kg @ R 0,40/kg	R 172 800
Storage vessel hire	12 weeks @ R 3000/month	R 36 000
Electricity		R 44 064
Maintenance cost		R 23 250
TOTAL		R 426 099

This amounts to R 9,86.kg O_3.

At a dosage of 2 g $O_3.kg^{-1}$ MLSS.d^{-1} and a transfer rate of 95 % in a plant operated under the same conditions as the experimental plant at Daspoort, 24,86 M$l.d^{-1}$ can be treated with this ozone generator. This would amount to 4,70 c.m^{-3}

Instead of purchasing an ozone generator ozone can be obtained in terms of a long term supply agreement. A 20 $kg.h^{-1}$ generator is available at a price of R 6.90 per kilogram O_3. The generator is owned, maintained and operated by the supplier. Power for the generator at 8,5 $kWh.kg^{-1}$ O_2 and cooling water as well as the necessary building to house the generator is excluded. A generator of this size can serve a 120 M$l.d^{-1}$ activated sludge plant .

5.4.3 Cost of bulking control with chlorine

The capital outlay to chlorinate activated sludge consists of a building to house the chlorination facilities and a chlorinator to control the chlorination. The building cost is estimated at R 40 000. Amortised at a rate of 14,5 % per annum results in an annual fixed cost of R 6 675. At a depreciation rate of 2,5 % the building will have a salvage value of R 25 000 in fifteen years. The uniform annual equivalent present value of this salvage value is R 545. The life expectancy of a chlorinator at a cost of R 12 000 is five years. This adds a further R 3 537 to the fixed cost. The total fixed costs of chlorination is therefore R 9 667 per annum.

The variable costs of chlorination consist of chlorine at a current price of R 3,26 per kilogram, cylinder hire at R 38,50 per week and maintenance of the building at 1,5 % of the capital cost.

The cost of using chlorine to control bulking sludge will depend on the sensitivity of the catchment into which the effluent is to be discharged. In catchments where the phosphate standard is not rigorously enforced, periodic treatment at higher doses can be applied than in catchments where the standard must be maintained all the time. In sensitive areas were the 1 mg P.l^{-1} standard must be maintained the maximum chlorine dose

should be below 1,5 g Cl_2.kg MLSS.d^{-1}. At this dose, it will take longer to bring bulking under control but it might be possible, or essential, to treat the reactor continuously. The cost per annum to treat activated sludge in the experimental plant at Daspoort, would have be been as follows:

Fixed cost		R 9 667
Chlorine	180447 kg @ R 3,26/kg	R 60 138
Cylinder hire	52 weeks @ R 38,50/week	R 2 002
Maintenance cost		R 1 300
TOTAL		R 73 107

This amounts to 1,51 cent per kilolitre influent.

In catchments were an occasional non-compliance to the 1 mg P.l^{-1} can be tolerated, higher chlorine doses can be employed to control bulking. The main advantage of employing higher doses is that bulking can be brought under control in a shorter period These higher doses can, however, not be maintained over long periods. At a dose of 3 g Cl_2.kg MLSS.d^{-1} bulking was controlled within 16 days. From the rate at which bulking return after the chlorination was stopped, it is estimated that five treatments per year should be sufficient to maintain the settleability of the activated sludge. The cost per annum incurred at a dose of 3 g Cl_2.kg MLSS.d^{-1} *in the experimental plant* at Daspoort, would have be been as follows:

Fixed cost		R 9 667
Chlorine	8080 kg @ R 3,26/kg	R 26 342
Cylinder hire	12 weeks @ R 38,00/week	R 462
Maintenance cost		R 1 300
TOTAL		R 37 771

This amounts to 0.78 cent per kilolitre influent.

5.5 SUMMARY OF FINANCIAL IMPLICATIONS OF NON-SPECIFIC BULKING CONTROL

The financial implications of controlling bulking sludge in a nutrient removal activated sludge plant are summarised in Table 10.

Table 10. Financial implications of non-specific sludge bulking control.

	Total savings cents per kilolitre influent	Total additional costs cents per kilolitre influent
Settling costs	2,00	
Sludge thickening and dewatering	2,27	
Hydrogen peroxide		15,61
Ozone		4,70
Chlorine		
Continuous		1,51
Intermittent		0,78

Only with chlorine is a nett saving possible.

CHAPTER 6
CONCLUSIONS AND RECOMMENDATIONS

The objectives of the research undertaken were to investigate preventative non-specific bulking control measures in a full-scale biological nutrient removal plant to ascertain the effect of hydrogen peroxide, ozone and chlorine on sludge settleability and the filamentous species composition, biological nitrogen and phosphorus removal, and to demonstrate the feasibility of the techniques on full-scale.

6.1 CONCLUSIONS

6.1.1 Sludge settleability

- H_2O_2 At low doses of 1,5 g H_2O_2.kg^{-1} MLSS.d^{-1} and 3 g H_2O_2.kg^{-1} MLSS.d^{-1} no bulking control was achieved, at a higher dose, 9,5 g H_2O_2.kg^{-1} MLSS.d^{-1}, sludge settleability improved temporarily. Hydrogen peroxide could not contain sludge bulking on the long term.
- O_3 : At the maximum attainable dose of 1,45 g O_3.kg^{-1} MLSS.d^{-1}, ozone gave an average, but consistent, improvement of 17 m$l.g^{-1}$ in the SSVI over ca 3,5 sludge ages. The lower doses of 1,07 and 0,36 g O_3.kg^{-1} MLSS.d^{-1} gave SSVI's which were on the average 12 m$l.g^{-1}$ lower than that of the control plant.
- Cl_2 : Chlorine at a dose of 1,42 g Cl_2.kg^{-1} MLSS.d^{-1} controlled bulking within 10 days, but could not achieve the target DSVI of 100 m$l.g^{-1}$. At a dose of 3,07 g Cl_2.kg^{-1} MLSS.d^{-1} bulking control was obtained in 4 days and the target DSVI was reached in 14 days.

6.1.2 Filamentous population

- H_2O_2 : Hydrogen peroxide treatment caused the formation of small diffused flocs and a reduction in the number of filaments and in particular *Microthrix parvicella*. The peroxide treatment did not physically damage the filaments,
- O_3 : In the ozonated plant large and small flocs were formed, an indirect indication that shorter filaments were formed during ozonation. No visual damage to the filaments was observed.
- Cl_2 : A dose of 1,42 g Cl_2.kg^{-1} MLSS.d^{-1} resulted in a reduction of the number of filaments and in the floc sizes, however no physical damage to the filaments was visible. At a dose of

3,07 g $Cl_2.kg^{-1}$ MLSS.d^{-1} smaller flocs were formed initially and after chlorination for 8 days larger firmer flocs started to form. Within 3 days of chlorination a decrease in the number of filaments was noticeable. Physical damage to the flocs was detected.

6.1.3 Nutrient removal

- H_2O_2 : At low doses of 3 and 6,34 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} hydrogen peroxide did not affect P-removal, but at a dose of 9,5 g $H_2O_2.kg^{-1}$ MLSS.d^{-1} P-removal deteriorated during the first 10 days. However, the P-removing organisms adapted to hydrogen peroxide because P-removal recovered while H_2O_2 was being dosed. On cessation of the hydrogen peroxide treatment phosphate removal deteriorated temporarily again.
  Hydrogen peroxide influenced nitrification and denitrification marginally only.
- O_3 : Ozone at a dose of 1.42 g $O_3.kg^{-1}$ MLSS.d^{-1} stabilises and promotes phosphate removal. Ozone influenced nitrification and denitrification marginally only.
- Cl_2 : Chlorine affected phosphate removal negatively depending on the dosing rate. At a dose of 1,42 g $Cl_2.kg^{-1}$ MLSS.d^{-1} phosphate removal is only marginally lower. If P-removal in a plant is at an optimum, it would be possible to achieve the 1 mg P.l^{-1} limit. At a dose of 3,07 g $Cl_2.kg^{-1}$ MLSS.d^{-1} and higher, P-removal can deteriorate to such an extent that the 1 mg P.l^{-1} limit will be exceeded. However, at this rate of chlorination P-removal is only suppressed and will recover rapidly after chlorination is stopped.
  No correlation was observed between nitrification/denitrification and the chlorine dose.

6.1.4 Costs

Chlorination of activated sludge is the most economical non-specific method of controlling bulking sludge in a nutrient removal activated sludge plant. With the necessary precaution, such as a daily trend plot of SVI values to follow the effect of chlorination and dosing at the minimum effect dose bulking can be controlled by chlorination with only a marginal effect, on phosphate removal.

6.2 RECOMMENDATIONS

- From the information available at this stage (Gabb *et al.* 1991, Casey *et al.* 1993) it is clear that it is not possible to design and operate a nutrient removal activated sludge system that will not bulk with low F/M organisms at some stage or another. The search for a more nutrient removal friendly

system of non-specific bulking control than chlorine must continue. To narrow the gap between bench scale and full-scale and to keep the cost down this research should be conducted on at least pilot scale.

- The beneficial use of ozone in wastewater treatment should be investigated further. Aspects that should be looked into are the formation of THM precursors, the impact on pathogenic organisms and the improved purification and nutrient removal potential.

An important aspect which became clear during the course of this project, is that upscaling from bench scale to full-scale can easily give results which can lead to the discarding of serviceable technologies. Bench scale experiments should only be used the establish a principle. To verify under what conditions the technology is applicable, and to gather data for upscaling, it should be tested on an industrial pilot scale in which the actual conditions in the full-scale plant can be simulated. Furthermore in non-specific bulking control in nutrient removal activated sludge systems using oxidants, dosages of the oxidants can only be optimised in a full-scale plant. Operators of full-scale plants should be encouraged to report their experiences in applying technologies.

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