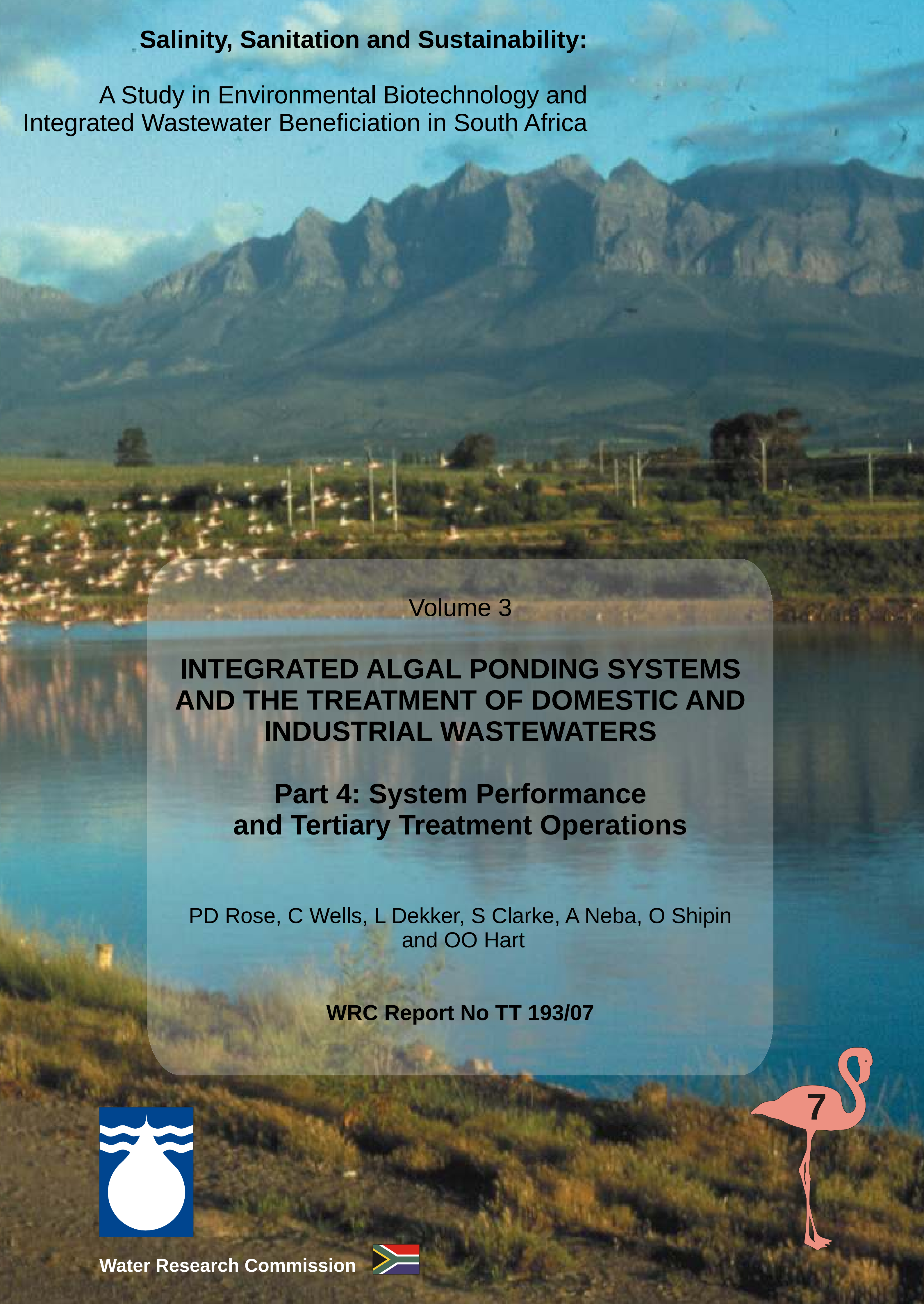

Salinity, Sanitation and Sustainability:

A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa

Volume 3

INTEGRATED ALGAL PONDING SYSTEMS AND THE TREATMENT OF DOMESTIC AND INDUSTRIAL WASTEWATERS

Part 4: System Performance and Tertiary Treatment Operations

PD Rose, C Wells, L Dekker, S Clarke, A Neba, O Shipin and OO Hart

WRC Report No TT 193/07

7

Water Research Commission

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INTEGRATED ALGAL PONDING SYSTEMS AND THE TREATMENT OF DOMESTIC AND INDUSTRIAL WASTEWATERS

Part 4: System Performance and Tertiary Treatment Operations

Report to the Water Research Commission

by

PD Rose, C Wells, L Dekker, S Clarke, A Neba, O Shipin and OO Hart
Environmental Biotechnology Research Unit
Rhodes University
Grahamstown

September 2007
WRC Report No: TT 193/07

Obtainable from:

Water Research Commission
Private Bag X03
Gezina
0031

Report to the Water Research Commission on Projects K5/799, 'Development and monitoring of Integrated Algal Ponding Systems technology for low-cost treatment of sewage and industrial effluents'; K5/1073,'Extension of applications and optimisation of operational performance of Integrated Algal Ponding Systems technology in low-cost treatment of industrial and domestic wastewaters'; K5/1362, 'Development and technology transfer of IAPS applications in upgrading water quality for small wastewater and drinking water treatment systems'.

Project Leader: Prof P.D. Rose

ISBN No: 978-1-86845-890-5
Set No: 1 86845 853 9

Printed in the Republic of South Africa

FOREWORD

The work presented in this series covers a decade of concerted research into critical sustainability issues in the water-scarce Southern African situation. The provision of safe and adequate drinking water and sanitation services to all our people remains a challenge. Pervasive salination from a range of mining, industrial and agricultural activities threatens the quality of our water resources. Simultaneously, the complex ecological needs of the aquatic environment are being understood with ever-increasing clarity.

Significant progress has been made in meeting some of these challenges. In the years since the democratic elections of 1994, millions of previously unserviced South Africans have been supplied with safe drinking water and sanitation services. The problem of increasing salinity of our water resources, with its direct economic impacts and future threat to sustainability, is being addressed at policy and implementation levels, for example by reduction-at-source measures. The ecological needs of the aquatic environment have been recognised by the provision in our water law of a prioritised ecological reserve, to be managed by the catchment management agencies being formed.

Such promising developments notwithstanding, ultimately sustainable resolution of these issues depends crucially also on acquiring appropriate and affordable technologies that provide physical solutions to our water-related challenges. It is in this context that the research described in this series deserves special commendation for the highly innovative biotechnological linkage developed between the treatment of saline wastewaters on one hand and domestic sewage and sludges on the other.

In the novel approach followed, salinity and sanitation issues are each viewed essentially as a resource base (rather than simply as “waste problems”) in a suite of integrated process schemes which can be variously manipulated to deliver products of treated water, recovered nutrients and metals, and algal biomass. The paradigm is consequently changed from one of “managing problems” to one of “engineering opportunities”, with the potential of offering a major contribution towards the management of water and sanitation in the RSA - some applications have already been taken to full scale implementation, for example in the accelerated digestion of sewage sludge. Significantly, the achievements of this research add weight to biotechnology as “the” technology of the 21st century.

So, as we approach the World Summit on Sustainable Development, we can reflect on the provisions of Agenda 21 adopted after the Earth Summit some 10 years ago, and note that in this time we have ourselves in various ways “done something” about our own situation. And we can therefore point with a justifiable sense of pride and achievement to the body of work presented here as being “Made in South Africa”, at a time when social, environmental, political and economic calls are being made to all of Africa to stand up in the continental and global communities of nations.

My deep thanks and appreciation go to the Water Research Commission for the foresight in funding this work, and, in particular, to Prof Peter Rose and his research team at Rhodes University, for the vision, purposefulness, innovation and application with which this work has been conceived and executed.

Ronnie Kasrils

Minister of Water Affairs and Forestry
Pretoria
31 July 2002

EDITOR'S NOTE

In 1990 the Water Research Commission, under the (then) Executive Director Dr Piet Odendaal, appointed the Environmental Biotechnology Group at Rhodes University, led by Prof Peter Rose, to carry out a one-year feasibility study to evaluate the potential of a biotechnological approach to the linked treatment and management of saline and sanitation wastewaters with recovery of useful components such as nutrient bio-products.

In the intervening years, this seminal project has resulted in a rich research programme, managed initially by Dr Oliver Hart, subsequently by Zola Ngcakani, and latterly (since 1997) by myself. The progression of the research programme is reflected in this series of reports. Report 1 critically reviews the main arguments considered in the sustainability discourse and their relation to salinity and sanitation, and presents an overview of the work covered in the individual Reports 2 – 12, each of which deals with specific aspects of the research programme. The reports are also to be issued on CD.

The research period concerned spans approximately the decade between the Rio Earth Summit in 1992 and the World Summit for Sustainable Development in Johannesburg. During this time, international concern has been expressed about the limited extent to which the sustainability objectives formulated at Rio, as captured for example in Agenda 21, have been followed through to implementation.

By contrast, it is a noteworthy achievement of this research programme that the "sustainable biotechnology" originally conceptualised by the researchers has in fact, by dint of rigorous research development, experimentation and testing, been translated into a suite of practicable processes for delivering treated water as well as value-adding organic and inorganic co-products. In some applications, full-scale plants are already being installed, fulfilling the cycle of research development implementation.

It is probably fair to say that the full potential of the original work initiated twelve years ago, with its various applications as they have been developed since then, could at inception only have been dimly foreseen – which, with hindsight, underscores the clarity, breadth and depth of the originators' vision.

It has been a pleasure and a privilege to be involved with this work, as Research Manager and now as Editor of this series. I am confident that you, the reader, will find the contents both informative and as stimulating as I have.

Greg Steenveld
Water Research Commission
Pretoria
31 July 2002

PREFACE

This report is one of a series of twelve Water Research Commission studies undertaken by the Environmental Biotechnology Group at Rhodes University, on biotechnology and integration in the management of saline and sanitation wastewater systems. Environmental problems in these areas are reckoned to be responsible for six of the seven priority pollution issues undermining the sustainable development project in Southern Africa. While both salinity and sanitation have separately been the subject of quite extensive investigation, relatively little has been reported on the potential linkage of these systems in meeting sustainable development objectives.

At the time these studies commenced in 1990, focus on the operationalisation of the sustainability idea had identified 'integrated waste resource management' as a key requirement for progress towards 'closed systems' production. Here human activities, and the associated technological environment, would be detached as far as possible from the bio-physical environment related to natural systems. Waste recovery, recycle and reuse had emerged as major strategies for achieving the radical shift to new technologies which would enable societies to live off nature's income, rather than consuming its capital. Waste beneficiation (a term still more common in the traditional resources sector, and referring to operations that add value by transforming raw material into finished products) was seen as a means of placing treatment operations on an economic footing, with value added in the form of products and services accrued in the waste management operation.

To meet the time-scale of the sustainability agenda, the breakthroughs in technology required would have to be initiated now to guarantee their availability in the next 2 to 4 decades. This led to widespread use of technology-push approaches in sustainable technologies research.

The principal aim of this programme was thus to investigate potential in environmental biotechnology for the development of technological enablement in the linkage of saline and sanitation wastewater management. This involved initial studies in the biology of organic saline wastewater impoundments and an evaluation of the recovery of nutrient values in these wastes in the form of high-value bio-products produced by halophilic micro-organisms. Integrated Algal Ponding Systems were investigated as a 'core technology' in delivering these objectives.

A critical path research methodology was used to identify technological constraints in the organic saline wastewater treatment operation and served to prioritise the research inputs required to underpin bioprocess development. Studies in the microbial ecology and environmental biotechnology of these systems provided the basis for bio-process innovation, and the subsequent development of treatment processes to full-scale engineered applications.

This series includes an introductory volume which provides an overview of the twelve-year programme to date. The reports are listed inside the title page and each study in the series is identified by a 'racing flamingo' number, which also appears on the outside cover. This relates to the appearance of a large flock of flamingos, which took up residence on tannery wastewater ponds following the installation of the *Spirulina*-based Integrated Algal Ponding System developed in the initial studies in this series. The development of the 'Salinity, Sanitation and Sustainability' programme is outlined below in Figure P1, and shows studies in the integrated algal ponding of saline, and domestic and industrial wastewaters, leading to the Rhodes BioSURE Process®, which provides linkage in the treatment of sulphate saline wastewaters and sewage sludge disposal.

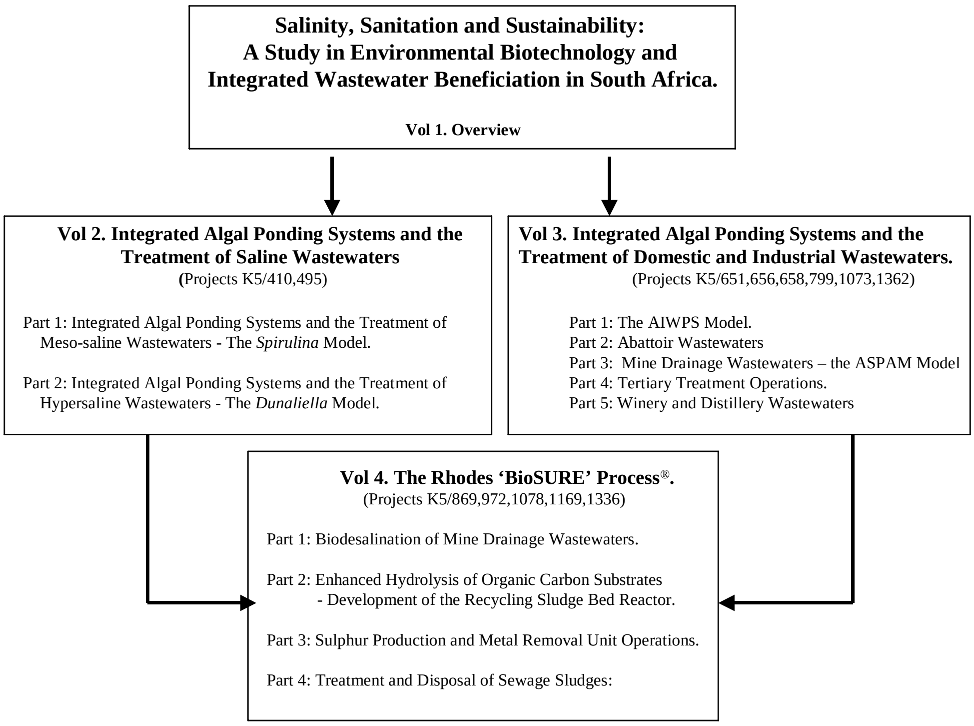

Figure P1. Research projects undertaken as components of the Water Research Commission study 'Salinity, Sanitation and Sustainability'.

A large number of people have assisted generously in many ways in the development of these studies, and are thanked under Acknowledgments. The support of former Water Research Commission Executive Director, Dr Piet Odendaal, is noted in particular. His vision of research needs in water resource sustainability, in the period leading to the Rio Earth Summit in 1992, not only contributed to this study, but also initiated early contributions to sustainable development research in water and sanitation service provision to developing communities. His inputs, together with Research Managers Dr Oliver Hart, Mr Zola Ngcakani, and Mr Greg Steenveld, have made substantial contributions to the development of the ideas investigated in these studies. The contribution and enthusiasm of my post-graduate research students is beyond measure.

Peter Rose
Environmental Biotechnology Research Unit
Rhodes University
Grahamstown

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EXECUTIVE SUMMARY

1. INTRODUCTION

Integrated Algal Ponding Systems (IAPS) utilise anaerobic and aerobic biological processes in wastewater treatment and Shelef (1987) has noted that these systems epitomise the principles of both water and nutrient recycling. They close the cycle of waste to primary biomass more rapidly than any other outdoor process, converting organic wastes into an algal biomass rich in protein, while stripping out nutrients. Furthermore, all this is accomplished without mechanical aeration but capitalising only on solar energy and, following algal harvesting, producing a high quality effluent not surpassed in quality by other biological or physico-chemical wastewater treatment processes. Ponds not only provide low-cost reactors, at least an order of magnitude cheaper than concrete structures (Oswald, 1995), but algal photosynthesis yields large quantities of oxygen to support bacterial breakdown of the organic components.

In 1990 the Water Research Commission (WRC) initiated a study of the application of the IAPS systems to a range of wastewater treatment problems in South Africa. This was undertaken by the Environmental Biotechnology Research Unit (EBRU) at Rhodes University and the result of the programme has been detailed in the current series "Salinity, Sanitation and Sustainability" and includes the investigation of "Integrated Algal Ponding Systems and the Treatment of Domestic and Industrial Wastewaters." In addition to domestic sewage, this study has also investigated applications of IAPS in the treatment of abattoir, tannery, winery and distillery and mine drainage wastewaters. Figure 1 shows the principal unit operations composing the IAPS process.

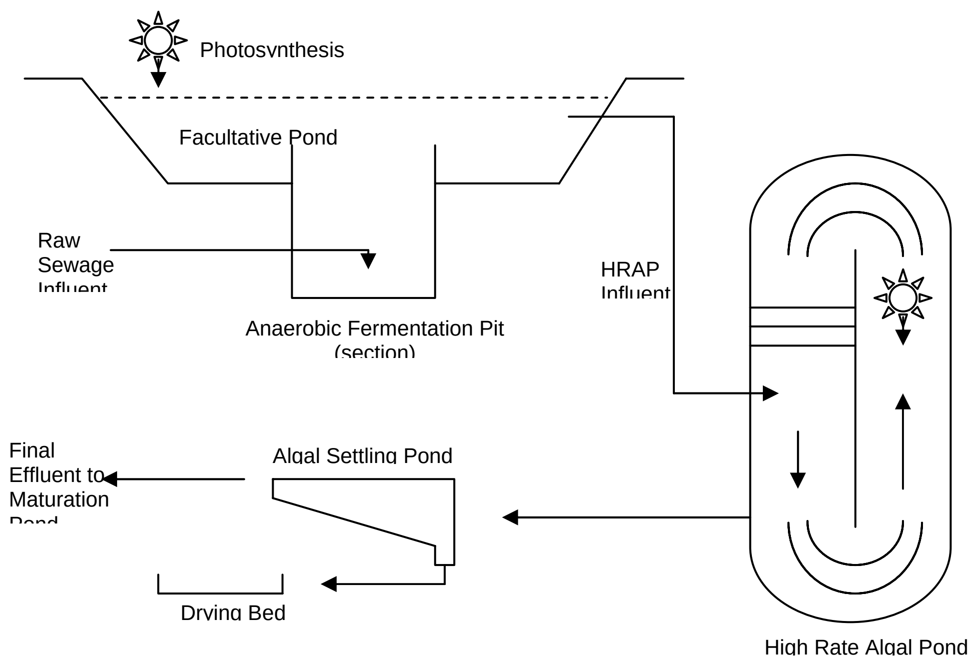

Figure 1: Schematic diagram of the principal unit operations of the IAPS Plant constructed at the Rhodes University Environmental Biotechnology Experimental Field Station, Grahamstown Disposal Works. After Rose *et al.* (2002c).

The design, construction, commissioning and implementation of the first IAPS demonstration and research plant in South Africa has been documented in detail in WRC Report TT 190/02, 'Integrated Algal Ponding Systems and the Treatment of Domestic and Industrial Wastewaters, Part 1: The AIWPS Model' (Rose *et al.,* 2002). This report should be read closely together with the current study which details the 9-year follow-up evaluation of the design as a low-cost domestic wastewater treatment technology with potential application throughout small municipality and rural treatment works. Research is also reported on the development of Tertiary Treatment applications of the system in improved disinfection and N and P removal unit operations. (See Notation for explanation of the use of the term IAPS as a general category and Advanced Integrated Wastewater Ponding Systems, AIWPS, as a trademarked special case of these systems).

1.2 WRC PROJECTS K5/799, K5/1073 AND K5/1362

This report details the results of three WRC Projects undertaken by EBRU as a follow-up to Project K5/651 in which the IAPS plant was constructed at the Environmental Biotechnology Experimental Field Station in Grahamstown. The principal areas of focus were the monitoring of IAPS performance under South African conditions, optimisation of process performance and the extension of the technology to other novel wastewater treatment applications. These studies were undertaken in the following projects:

1.2.1 Monitoring of IAPS Performance under South African Conditions

WRC Project K5/799, 'Development and Monitoring of Integrated Algal High Rate Oxidation Pond Technology for Low-cost Treatment of Sewage and Industrial Effluents'.

1.2.2 Extension of Applications and Optimisation of the Process

WRC Project K5/1073, 'Extension of Applications and Optimisation of Operational Performance of Algal Integrated Ponding Systems Technology in Appropriate Low-cost Treatment of Industrial and Domestic Wastewaters'.

WRC Project K5/1362, 'Development and Technology Transfer of IAPS Applications in Upgrading Water Quality for Small Wastewater and Drinking Water Treatment Systems'.

1.3 OVERALL PERFORMANCE EVALUATION OF THE IAPS RESEARCH AND DEMONSTRATION PLANT – THE 9-YEAR STUDY

The IAPS plant was monitored over a period of 9 years and results of this study are detailed in chapter 4. While the system performed well and delivered a final wastewater superior to most ponding systems operated in South Africa, it was nevertheless evident that, as operated in Grahamstown, the system would be unlikely to meet the DWAF General Standard for nutrient removal with any consistency. With this in mind, the development of the High Rate Algal Pond (HRAP) as a tertiary treatment unit operation was undertaken in what became known as the Independent High Rate Algal Pond (I-HRAP).

In the standard IAPS design (Figure 2A), secondary and tertiary treatment operations are averaged in a single HRAP. In the initial design of the plant at EBRU two HRAPs were operated in parallel. In the I-HRAP, the secondary and tertiary treatment operations are separated into two HRAPs functioning in series and thus enabling the independent optimisation of their functions (Figure 2C). It was shown in this way that the I-HRAP could be added onto the IAPS, and thus deliver a final water that was within the General Standard specification for nitrogen and phosphorus removal and for disinfection. Alternately, the I-HRAP could be applied as a free-standing tertiary treatment unit operation that could be used as an add-on to any other sub-optimally performing water treatment works. This application was demonstrated in the polishing of Grahamstown Disposal Works (GDW) final effluent (Figure 2B), and these results are detailed in chapter 5.

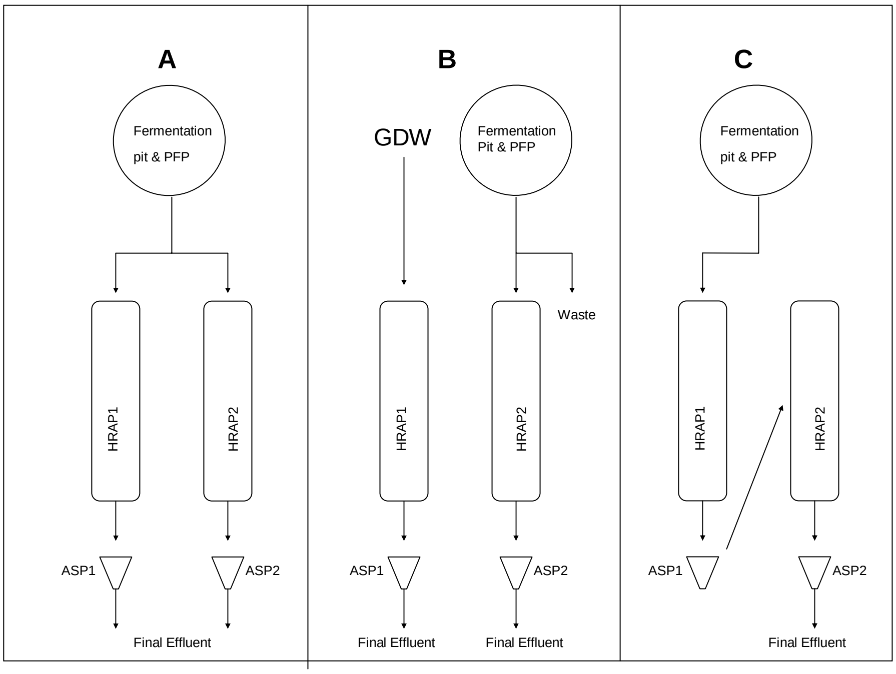

Figure 2: Development of the Independent High Rate Algal Pond (I-HRAP) system and showing the various configurations of the HRAP evaluated in this study. PFP = Primary Facultative Pond, ASP = Algal Settling Pond, GDW = Grahamstown Disposal Works.

The following performance for the system averaged over the 9-year period was recorded as follows:

1.3.1 Organics Removal

Figure 3 illustrates the total (unfiltered) CODt removal performance across the various units of the IAPS averaged over the period July 1996 to October 2004.

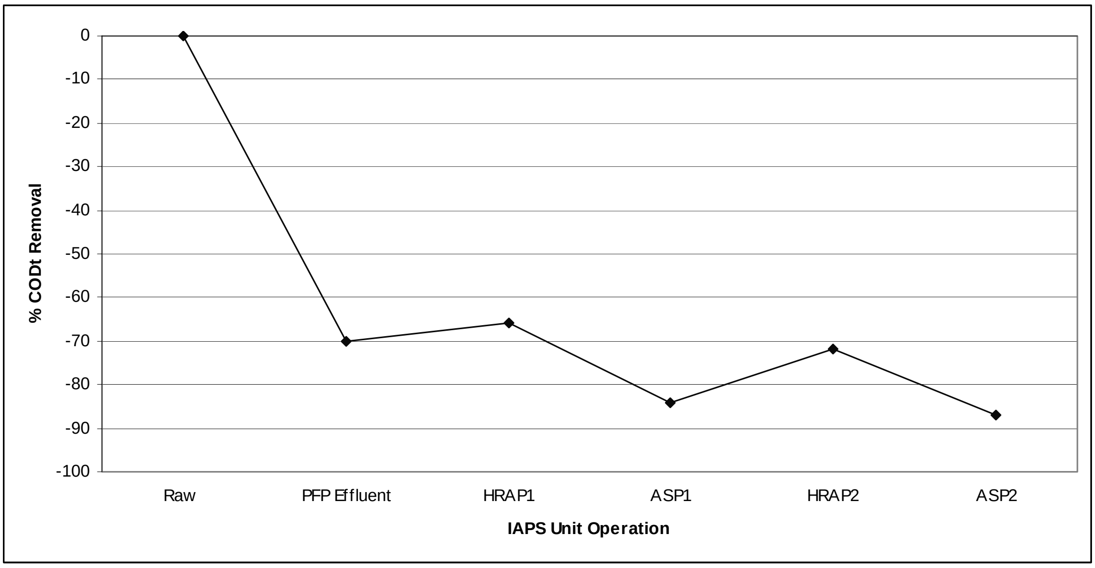

Figure 3: Total chemical oxygen demand removal through the integrated algal ponding system. The results depicted for the raw water, PFP effluent, HRAP1 and ASP1 reflect averages for HRAP treatment for the entire period from July 1996 to October 2004. HRAP2 and ASP2 were only brought online as the I-HRAP operation in July 2003. PFP = Primary Facultative Pond, HRAP = High Rate Algal Pond, ASP = Algal Settling Pond.

As shown in Figure 3, the mean CODt removal rate through the IAPS over the 9 year operation period was 87%. This is comparable with conventional wastewater treatment processes such as activated sludge and trickle filters (Horan, 1996; Maier *et al*., 2000; Henze *et al*., 2002) as well as waste stabilisation ponds (Bryant, 1986; Mara & Pearson, 1986; Soler *et al*., 1995; Racault *et al*., 1995). In a study of a stabilisation pond in Dar es Salaam, Kayomba *et al*. (2002) only found a 71% removal efficiency. Oswald (1991a) reports a slightly better performance of 93% at the AIWPS plant in St Helena, California. The CODt increases in HRAP1 and HRAP2 are related to the increase in algal biomass. Due to the algal component, the General Standard of <75 mg.ℓ^{-1} COD_t was seldom met and averaged between 85 and 120 mg.ℓ^{-1} COD. Where the excess COD is due to algal components, DWAF exemption is required.

1.3.2 Nitrogen Removal

Figures 4 and 5 illustrate the cycling of ammonia and nitrate, respectively, through the IAPS. Due to ammonification and possibly nitrogen fixation, there is an increase in ammonia in the first HRAP. This is then effectively removed in HRAP2 by the probable mechanism of volatilisation and possibly some assimilation into the algal biomass. The mean ammonia level in the final effluent was thus less than 1.5 mg.ℓ^{-1}. This low level was consistently achieved in the system, remaining under the DWAF standard (3 mg.ℓ^{-1}) 92% of the time and under 0.5 mg.ℓ^{-1} 68% of the time. This is a considerably better performance than activated sludge or trickle filters where ammonia levels of between 10 and 40 mg.ℓ^{-1} are common (Horan, 1996). It also appears to be better than ordinary WSP, where effluent ammonia values of between 5 and up to 50 mg.ℓ^{-1} have been reported (Racault *et al*., 1995; Mendes *et al*., 1995; Ceballos *et al*., 1995). High Rate Ponds studied by El Hamouri *et al*. (1995) in Morocco and Green *et al*. (1996) in California also had mean ammonia levels of no lower than 7.8 and 5.3 mg.ℓ$^{-1}$ respectively. With seasonal CaO addition and algae separation units in a HRAP, Nurdogan and Oswald (1995) were able to obtain effluent ammonia levels of between 2 and 3 mg.ℓ$^{-1}$.

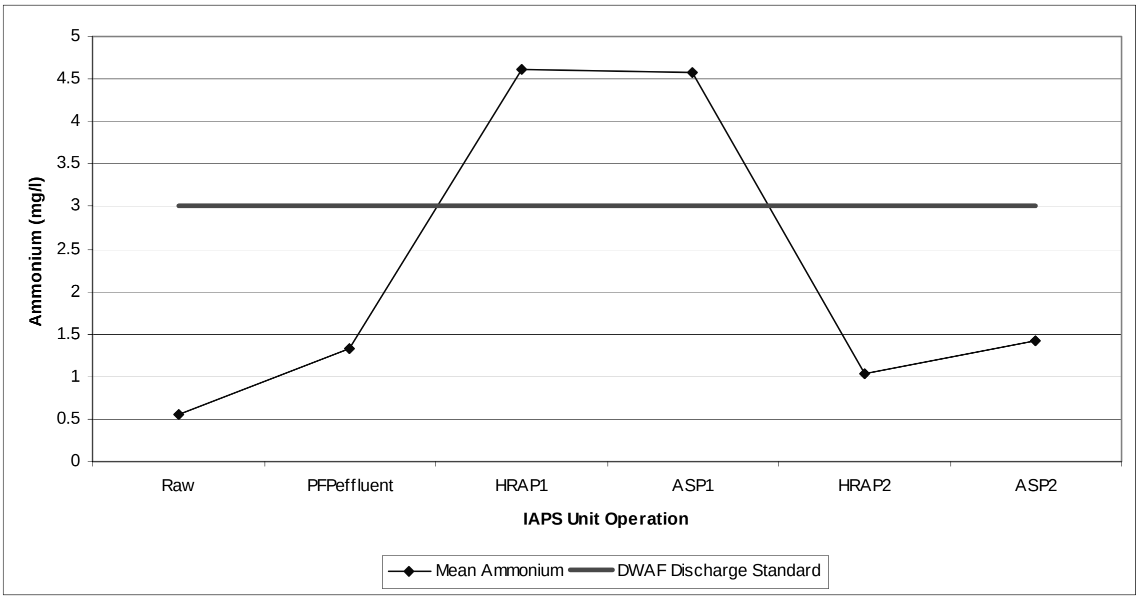

Figure 4: Average results for ammonia cycling through the IAPS, monitored over the period 1997-2004. HRAP2 and ASP2 were only brought online as the I-HRAP operation in July 2003. . PFP = Primary Facultative Pond, HRAP = High Rate Algal Pond, ASP = Algal Settling Pond.

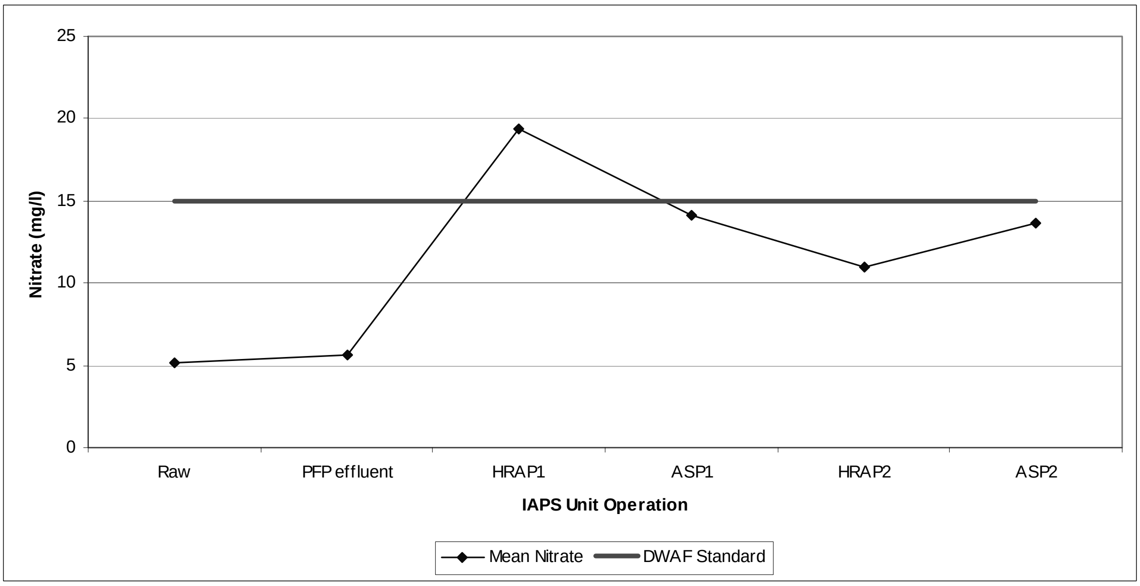

Figure 5: Average values for nitrate cycling through the IAPS, monitored over the period 1997-2004. HRAP2 and ASP2 were only brought online as the I-HRAP operation in July 2003. . PFP = Primary Facultative Pond, HRAP = High Rate Algal Pond, ASP = Algal Settling Pond.

The mean nitrate levels in the effluent over the 9-year life of the IAPS (Figure 5) was below the 15 mg.ℓ$^{-1}$ DWAF discharge standard. A mean total nitrogen (TKN) removal in the system of 55% was observed. Reported TKN removal rates in conventional WSP vary from 35 to 88% (Reed, 1985; Racault *et al*., 1995; Mendes *et al*., 1995; Sukias *et al*., 2003).

1.3.3 Phosphate Removal

The efficacy of phosphate removal in the IAPS is shown in Figure 6. The mean removal rate over the study period was 76%, with >85% removal occurring during 90% of operation. The mean concentration in the treated effluent was 5.4 mg.ℓ$^{-1}$, considerably lower than the South African discharge standard. Studies of WSP in Portugal and France revealed phosphate removal efficiencies of between 50 and 67% (Racault *et al*., 1995; Mendes *et al*., 1995). Constructed wetlands in Brazil and New Zealand, by comparison, reduced phosphate levels by between 5 and 46% (Tanner and Sukias, 2003; Sezerino *et al*., 2003). HRAP in Morocco had mean removal rates from 52 to 61% (El Hamouri *et al.*, 1994; El Hamouri *et al*., 1995). Nurdogan and Oswald (1995) were, however, able to achieve up to 99% phosphate removal with the addition of CaO.

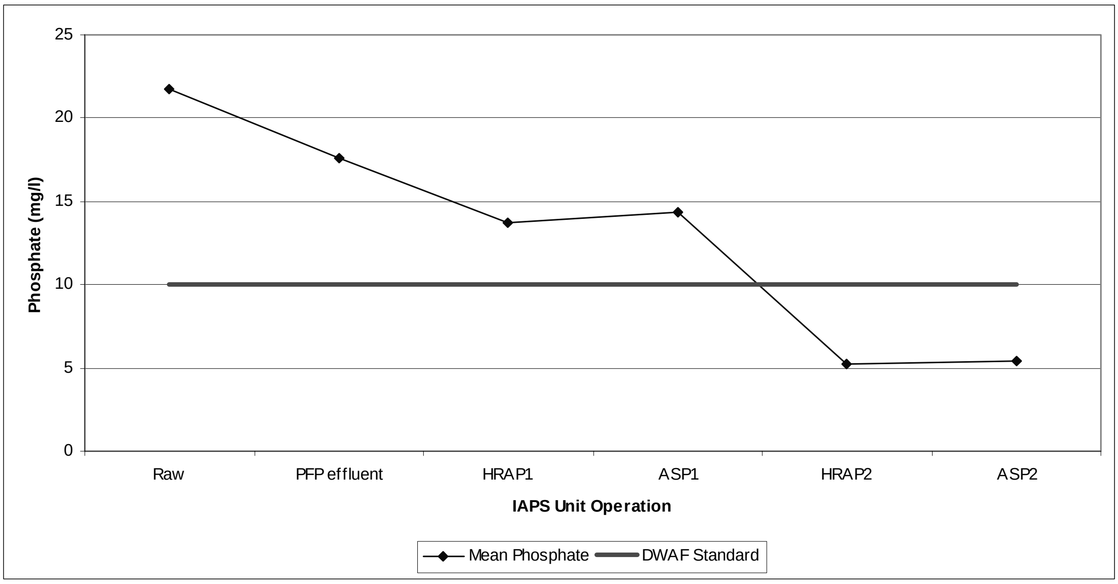

Figure 6: Average results for phosphate removal in the IAPS, monitored over the period 1997-2004. HRAP2 and ASP2 were only brought online as the I-HRAP operation in July 2003. . PFP = Primary Facultative Pond, HRAP = High Rate Algal Pond, ASP = Algal Settling Pond.

1.3.4 Disinfection

Figure 7 illustrates the mean faecal indicator *E. coli* counts in the various ponds in the IAPS sequence before addition of the I-HRAP unit. A greater than 4 log reduction in the *E. coli* CFU count in the system was recorded over the period, with the final effluent having a count of <1000 cfu.100 mℓ$^{-1}$. These figures, however, represent the mean results over the total eighteen month monitoring period, including winter periods and experimental conditions which allowed insufficient hydraulic retention times. A number of studies were undertaken to optimise HRT in the I-HRAP and given results of 6 days retention in winter and 3 days retention in summer, a further 2 log reduction was achieved, with a final mean count of <10 cfu.100 mℓ$^{-1}$ (Figure 8). This equates to a 99.999% reduction. Zero *E. coli* CFU were recorded in 78% of the samples tested producing a result of <1 CFU.100 mℓ$^{-1}$.

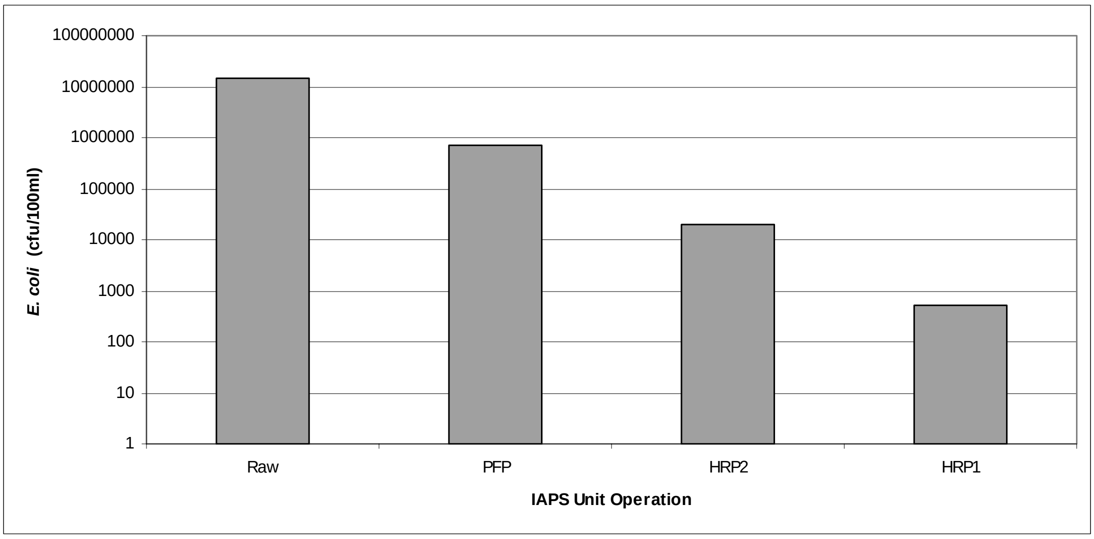

Figure 7: *E. coli* counts through the IAPS and I-HRAP sequence. This figure illustrates all data from the 2003-2004 monitoring period, i.e. including results from operation with sub-optimal hydraulic retention times. . PFP = Primary Facultative Pond, HRAP = High Rate Algal Pond.

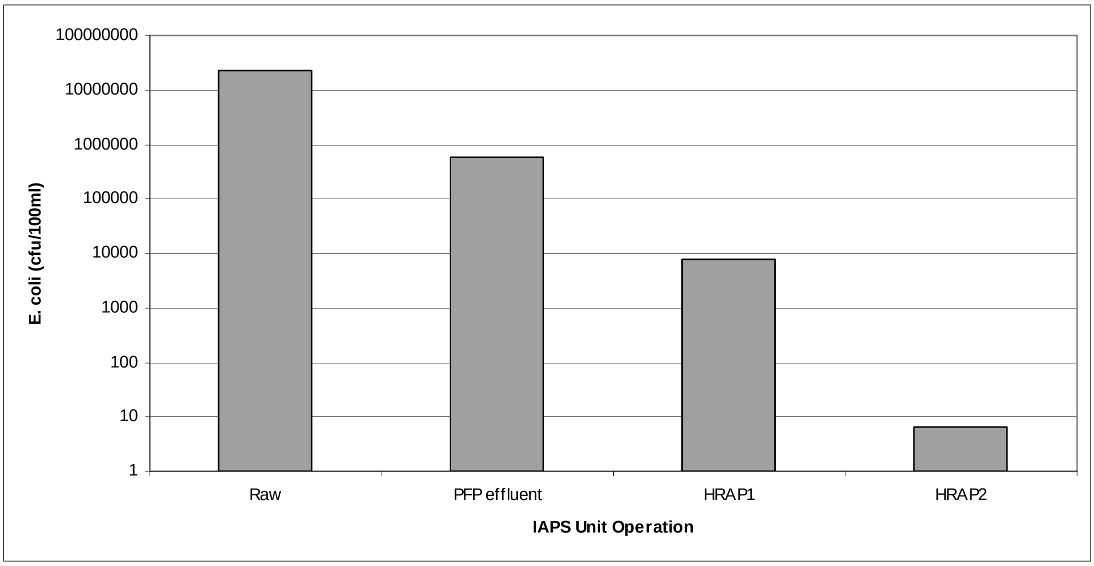

Figure 8: *E. coli* counts through the IAPS and I-HRAP sequence, illustrating results only from operation under optimal hydraulic retention times, both during 2003 and 2004. . PFP = Primary Facultative Pond, HRAP = High Rate Algal Pond.

In most instances, standard WSP are unable to reduce faecal coliforms to below 1000 cfu.100 $m\ell^{-1}$ (El Hamouri *et al*., 1994; Jagels & Lues, 1996; Almasi & Pescod, 1996; Rangeby *et al*., 1996; Garcia & Bécares, 1997; Bahlaoui *et al*., 1997). Wetland systems have also shown potential for biological pathogen reduction but effluents generally contain faecal coliforms in excess of 1000 cfu.100 $m\ell^{-1}$ (Arias *et al*., 2003; Ansola *et al*., 2003). Davies-Colley *et al*. (2003) achieved results similar to the Grahamstown I-HRAP system, using a HRAP followed by a maturation pond in New Zealand. Sebastian and Nair (1984) also reported total *E. coli* removal with a 2 day contact time at pH 11 in an experimental HRAP system operated in India.

1.4 I-HRAP-linked Denitrification Using Algal Biomass as the Carbon Source

Although effective disinfection and phosphate and ammonia removal had been demonstrated for the I-HRAP tertiary treatment operation, nitrate removal performance was found to be variable in both the IAPS and the I-HRAP experimental investigations. The potential for optimizing denitrification in the I-HRAP operation using the algal biomass produced in the system as the carbon source was thus investigated. It was shown that nitrogen removal may be improved in this way and the results of these studies are reported in chapter 9.

1.5 Technology Transfer

A number of technology transfer functions of IAPS technology were undertaken in the treatment of tannery, abattoir, hypersaline and acid mine drainage wastewater applications (See Appendix 1). The potential application of the I-HRAP in upgrading the performance of existing poorly performing small sewage works was also considered and its use as a low-cost add-on unit operation to enable compliance in these treatment works was investigated.

The algae that is settled and separated in the algae settling ponds is a beneficial by-product of the HRAP treatment system and has a number of potential uses other than as a potential carbon source for denitrification. As it is rich in nutrients and plant hormones, the most obvious use would be as a fertilizer (Benemann *et al*., 1980)

Horan and Horan (2004) have undertaken follow-up WRC Project K5/1619, ‘ IAPS Algal Biomass and Treated Effluent Utilisation as a Key Strategy in Sustainable and Low-cost Sanitation’ in order to investigate this potential. For IAPS algal-supplemented trial plantings they have found turnip yields of 1.4 times greater, by mass, compared with crops grown using commercial fertiliser (2:3:2, N:P:K) and 8.7 times those in unfertilised plots. Plots treated with algae and fertiliser yielded turnips with a mean weight 12.6 times that of the control. Similarly, they cultivated Swiss chard at 15.4 $t.ha^{-1}$ in soil enriched with HRAP algae, whilst commercial fertiliser only yielded 10.5 $t.ha^{-1}$ and unfertilised land, 3.2 $t.ha^{-1}$. A combination of algae and fertiliser once again had the greatest yield at 18.5 $t.ha^{-1}$ (Horan & Horan, 2004).

Another potential use of the algae is as a dietary protein feed supplement for animal nutrition including pigs, poultry and cattle (McGarry & Tongkasame, 1970). In Thailand the production of *Tilapia mosambique* was proved feasible with the use of algae-containing pond effluent (McGarry & Tongkasame, 1970). Nutritional analyses of the EBRU HRAP algal biomass revealed an approximate composition (protein 41.5%, lipid 4.8%, carbohydrate 35.1%) similar to that of other feed supplements such as soya oil cake meal and sorghum gluten meal (Potts, 1998). Potts (1998) was able to include this algae in formulated diets at protein levels of up to 20% to productively grow ornamental fish (family: Poeciliidae) in an experimental system. A further potential use of wastewater grown algae is in energy generation via their fermentation to methane (Oswald, 1998c).

The potential of linking water treatment and social activity, including job creation, through the recovery and re-use of treated waters has been the subject of WRC Project K5/1456/Part 4 "The Biotechnology of Saline and Sewage Wastewater Co-treatment" (Rose *et al.*, 2005). This study investigated the application of treated acid mine drainage wastewaters in urban agricultural programmes.

The following model is proposed for the application of the IAPS and particularly I-HRAP technology in linking water treatment and job creation initiatives which are dependent on the ability of the system to produce a water quality that at least meets DWAF irrigation water discharge standards. The development of this model is dealt with in greater detail in WRC Project K5/1619 "IAPS Algal Biomass and Treated Effluent Utilisation as a Key Strategy in Sustainable and Low Cost Sanitation."

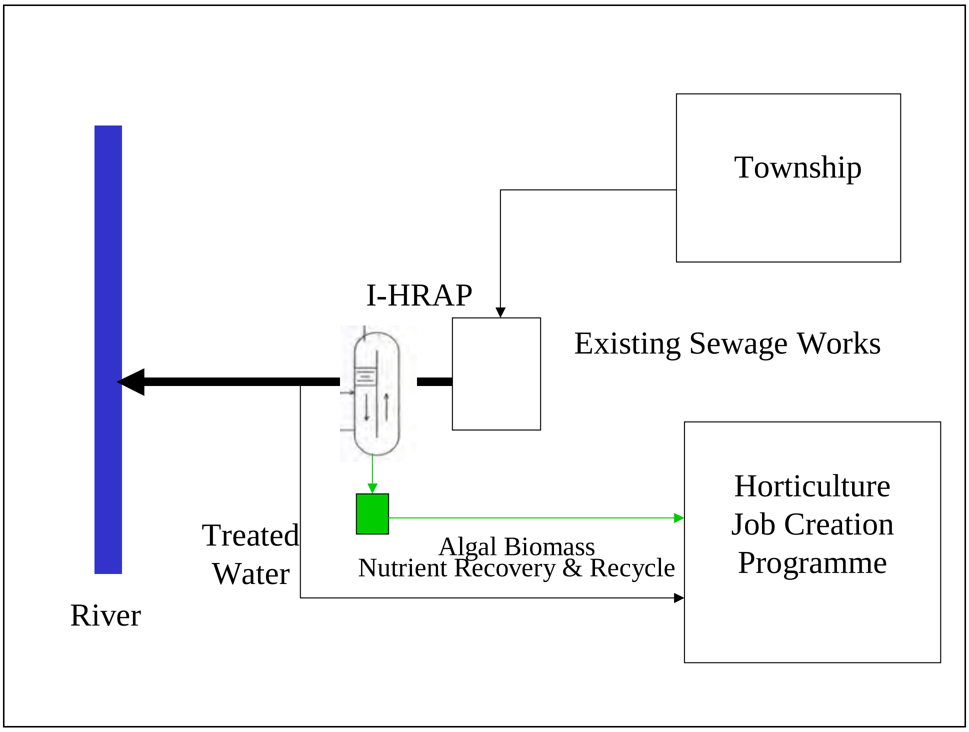

Figure 11.1: A schematic outline of the application of the I-HRAP as a retrofit to an existing poorly performing sewage works. Tertiary treatment, including disinfection, would enable the recovery and re-use of the water resource, and algal biomass as fertiliser, in community gardening or urban agriculture job creation projects.

ACKNOWLEDGMENTS

The authors wish to express their thanks and appreciation to the WRC for finance and support throughout the project. A wide range of people and organisations have been involve in the provision of generous support, advice, encouragement and other inputs associated with the execution of this study.

- The previous Executive and Deputy Executive Directors of the WRC, Messrs. P Odendaal and D van der Merwe who have provided substantial advisory inputs, moral support and encouragement throughout;
- WRC Research Manager Mr Greg Steenveld whose enthusiasm and engagement with the programme has added substantial value to the findings recorded here;
- Mr Roger Peter and Grahamstown Engineering Co. for sustained inventiveness in pilot plant construction. Mr Brian Ford and BJ Electrical for electrical and control elements;
- Rhodes University and the City of Grahamstown, who have generously provided facilities for much of the work undertaken

The Project Team:

- Prof PD Rose (Project Leader)
- Dr OO Hart (Researcher)
- L Dekker (Researcher)
- S Clark
- A Neba
- CD Wells

The Steering Committee Members for the WRC Projects K5/799, K5/1073 and K5/1362:

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- Dr G Offringa
- Dr B Petschke
- Mr HM du Plessis
- Mrs I de Moor (Water Research Commission)

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AIWPS	Advanced Integrated Wastewater Ponding System
ANTRIC	Anaerobic Trickle Filter
ASP	Algal Settling Pond
ASPAM	Algal Sulphate Reducing Ponding Process for Acidic and Metal Wastewater Treatment
BFB	Biological upflow Fluidised Bed
BOD	Biological Oxygen Demand
BODult	Ultimate Biological Oxygen Demand
CFU	Colony Forming Units
COD	Chemical Oxygen Demand
CODs	Soluble (filtered) Chemical Oxygen Demand
CODt	Total (unfiltered) Chemical Oxygen Demand
CPS	Capsular Polysaccahride
DAF	Dissolved Air Flotation
DEAT	Department of Environmental Affairs and Tourism
D-HRAP	Dunaliella – High Rate Algal Pond
DO	Dissolved Oxygen
DPB	Denitrifying Phosphorus removing Bacteria
DWAF	Department of Water Affairs and Forestry
EBEFS	Rhodes University Environmental Biotechnology Experimental Field Station
EBG	Environmental Biotechnology Group
EBPR	Enhanced Biological Phosphate Removal
EBRU	Environmental Biotechnology Research Unit (Rhodes University)
EPA	United States Environmental Protection Agency
EPS	Extracellular Polymeric Substances
GDW	Grahamstown Disposal Works
HAP	Calcium Hydroxyapatite
HRAP	High Rate Algal Pond

HRP	High Rate Ponds
HRT	Hydraulic Retention Time
IAPS	Integrated Algal Ponding System
I-HRAP	Independent High Rate Algal Pond
IWRM	Integrated Water Resource Management
MPN	Mean Probably Number
PAO	Phosphorous Accumulating Organisms
PAR	Photosynthetic Available Radiation
PE	Person Equivalent
PFP	Primary Facultative Pond
RO	Reverse Osmosis
RSDN	Rural Services Development Network
SBR	Sequential Batch Reactor
S-HRAP	Spiruline – High Rate Algal Pond
TCP	Tricalcium Phosphate
TDIS	Total Dissolved Inorganic Solids
TDS	Total Dissolved Solids
TKN	Total Kjeldal Nitrogen
TPS	Total Polysaccharides
UASB	Upflow Anaerobic Sludge Blanket
UV	Ultraviolet Light
VBNC	Viable but Non-Culturable Cells
VFA	Volatile Fatty Acid
WRC	Water Research Commission
WSP	Waste Stabilisation Ponds

NOTATION

A wide range of terms have been used over the years by different authors to describe various configurations of ponding systems used in wastewater treatment and in algal biotechnology applications. This has created certain confusion in the literature, and to avoid possible further confusion the following usage has been followed in this study:

- The term Advanced Integrated Wastewater Ponding System (AIWPS) refers to a specific trade-marked process application design. This ownership of name has been respected and care has been taken throughout not to use the term in a generic sense to cover the many forms of integrated ponding systems involving the use of algal photosynthesis. The term Algal Integrated Ponding Systems (AIPS), and Integrated Algal High Rate Oxidation Ponding Process (IAHROP) which was used in this sense in the earlier part of this study to describe the hybrid algal ponding systems, the development of which was under consideration in this programme, has been changed to Integrated Algal Ponding Systems (IAPS) to avoid confusion;

- The IAPS is used here to refer generically to combinations of ponding system unit operations involving an algal component in their operation;

- The term High Rate Algal Pond (HRAP) has been used here and replaces High Rate Oxidation Pond (HROP) used in some literature references, as it is not necessarily inclusive of the algal component;

- The terms algae or micro-algae are used for convenience in the more traditional sense broadly covering both the eukaryotic algae as well as the cyanobacteria.

1. THE DEVELOPMENT OF INTEGRATED ALGAL PONDING SYSTEMS AS A PLATFORM TECHNOLOGY IN INTEGRATED WASTEWATER BENEFICIATION: A WRC PROGRAMME

1.1 BACKGROUND

The continued availability of water to meet both environmental and human requirements presents one of the most serious challenges to the Sustainable Development project in South Africa (State of the Environment Report, Department of Environmental Affairs and Tourism, DEAT, 1999). The predicament of a rapidly accelerating demand, against the background of increasing pollution and degradation of the finite resource, has been the subject of growing concern over many years, and it has become evident that water may increasingly become the limiting resource restricting future socio-economic development of the country (Commission of Enquiry into Water Matters, 1970; Stander, 1987; DEAT 1999, 2000). In this regard, salinity and sanitation have been identified as six of the seven priority problem areas and also that the development of locally appropriate treatment technologies to deal with these problems requires urgent attention (DEAT, 2000). Water and sanitation also play a central role in the delivery, by 2015, of Millennium Development Goals in ensuring environmental sustainability, combating disease and the eradication of extreme poverty.

This report is part of a 15-year Water Research Commission (WRC) study undertaken by the Environmental Biotechnology Research Unit at Rhodes University (EBRU), on applications of biotechnology in an integrated management of the salinity and sanitation wastewater problems. These studies have been detailed in a 5-volume WRC report series titled ‘Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa’ (Rose *et al.*, 2002a)

A central thrust of this programme was the attempt to link technology innovation in the treatment of specific waste streams with the recovery of products of value that would provide both an incentive for treatment and, within the concept of sustainable resource recovery and reuse, to contribute through their beneficiation to wealth creation and the goal of poverty eradication. Fundamental work in the photosynthetic potential of the micro-algae, and the production of speciality biomass and high-value algal bio-chemical products, had developed rapidly since the 1970s, and the development of Integrated Algal Ponding Systems (IAPS), over much the same period, presented a technology platform that might be usefully explored in linking treatment of the salinity and sanitation wastewater problems.

The feasibility of an Algal Biotechnology approach to the problem was evaluated in 1990 in an initial study (WRC Project K5/410, 'A biotechnological approach to the removal of organics from saline effluents'). This led to a five-year follow-up project (K5/495 of the same title), during which a range of applications of the IAPS concept were explored and tested experimentally (Rose *et al.*, 2002b).

1.2 THE AIWPS – A MODEL SYSTEM

Early in the IAPS applications research programme, it was recognised that progress in the development of an algal-based wastewater treatment technology approach in South Africa would require a matching investment in the development of the engineering skills necessary for their construction and operation under local conditions. In this regard the Advanced Integrated Wastewater Ponding Systems (AIWPS) design, developed by Prof William Oswald over nearly 40 years, at the Sanitary Engineering and Environmental Health Research Laboratory, University of California, Berkley, USA, was identified as representing one of the most intensively engineered and developed of IAPS-type applications (Oswald, 1995, 1988 a&b). This development had been principally focussed on sewage treatment and was shown to be particularly applicable to low-cost community development projects. Numerous plants were operating successfully in both the USA and developing countries (see Notation for clarification of terminology used).

Following a visit to California by Professor Peter Rose, and then by WRC Research Manager Dr Oliver Hart, to investigate the AIWPS technology in the USA, Professor Oswald was invited to visit South Africa by the WRC, in May 1993, to lecture at various venues on the principles of the AIWPS concept. This interaction led to the WRC Project K5/651: 'Appropriate low-cost sewage treatment using the advanced algal high rate oxidation pond', which commenced in 1994. This project undertook the technology transfer exercise, in collaboration with Prof Oswald and Dr Bailey Greene, both of UC Berkley, whereby an AIWPS design was implemented as part of an IAPS research plant at the Rhodes University Environmental Biotechnology Experimental Field Station in Grahamstown.

The design, construction, commissioning and implementation of the first AIWPS plant in South Africa has been documented in detail in the WRC Report TT 190/02, 'Integrated Algal Ponding Systems and the Treatment of Domestic and Industrial Wastewaters, Part 1: The AIWPS Model' (Rose *et al.*, 2002c). This report should be read closely together with the current study which details the following 9-year evaluation of the system as the basis for developing low-cost wastewater treatment technology in a range of potential wastewater treatment applications. Research is also reported on the development of Tertiary Treatment applications of the system in improved disinfection and nitrogen and phosphorus removal unit operations.

1.3 IAPS RESEARCH DEVELOPMENT

The technology transfer initiative, which resulted in the construction of the AIWPS plant in Grahamstown, was undertaken as a component of an engineering support base in the development of an IAPS technology platform in the treatment of sewage and saline wastewaters. A number of applications grew out of this initiative which is the subject of the 'Salinity, Sanitation and Sustainability' report series. The main features of the programme are outlined below and the reports noted in Appendix 1:

- Development of IAPS in the treatment of saline tannery wastewaters. A full-scale system was constructed at Mossop-Western Leathers Co tannery in Wellington, Western Cape, South Africa (Report 2);
- Development of IAPS in the treatment of abattoir wastewaters. A pilot-scale plant was constructed at Cato Ridge Abattoir, Kwa Zulu-Natal, South Africa (Report 5);
- Development of an IAPS hybrid system in the treatment of saline winery and distillery wastewaters. A pilot plant was constructed at Brennokem Pty Ltd. Wine lees plant in Worcester, Western Cape, South Africa (Report 8);
- Development of the Process (Algal Sulphate Reducing Ponding Process for the Treatment of Acid Mine Drainage Wastewaters (ASPAM). A pilot plant has been constructed at EBRU Laboratories in Grahamstown (Report 6)
- Development of the Independent High Rate Algal Pond (I-HRAP) as a free-standing algal unit operation in waste water treatment. A number of applications have been developed including:

  The Spirulina-High Rate Algal Pond (S-HRAP) treating meso-saline wastewaters (<40 g.ℓ^{-1} Total Dissolved Inorganic Solids). A 2 500 m^2 industrial-scale HRAP was constructed at Mossop-Western Leathers Co. tannery in Wellington as part of the IAPS system (Report 2)

  The Dunaliella-High Rate Algal Pond (D-HRAP) treating hyper-saline wastewaters (>40 g.ℓ^{-1} TDIS). A pilot plant was constructed at Botswana Ash Co., Sua Pan, Botswana (Report 3);

  The I-HRAP developed for tertiary treatment operations in domestic wastewater treatment including disinfection and nutrient removal. A pilot plant has been constructed at the Environmental Biotechnology Experimental Field Station in Grahamstown (Report 7);

- Development of the Rhodes BioSURE Process®. This was based on research findings on the hydrolysis of complex carbon substrates in the anaerobic sulphate reducing compartments of IAPS treating sulphate saline wastewaters. The application of these finding to the treatment of acid mine drainage wastewaters led to the construction of pilot-, technical- and now a full-scale BioSURE plant in Springs, Gauteng South Africa, treating 10 Mℓ.day^{-1} mine drainage (Reports 9-12).

1.4 WRC PROJECTS K5/799, K5/1073 AND K5/1362

This report details the results of three WRC Projects undertaken by Rhodes University Environmental Biotechnology Research Unit as a follow-up to Project K5/651 in which the AIWPS plant was constructed at the Environmental Biotechnology Experimental Field Station in Grahamstown. The broad objectives of this work were as follows:

1.4.1 Monitoring of AIWPS Performance under South African Conditions

WRC Project K5/799, 'Development and Monitoring of Integrated Algal High Rate Oxidation Pond Technology for Low-cost Treatment of Sewage and Industrial Effluents.

- To undertake a comprehensive monitoring of the process
- To evaluate flexibility of the design in other IAPS applications
- To consider the value-added potential of algal biomass generated;
- To report on constraints and future research needs relating to the wide-scale application of the technology.

1.4.2 Extension of Applications and Optimisation of the Process

WRC Project K5/1073, 'Extension of Applications and Optimisation of Operational Performance of Algal Integrated Ponding Systems Technology in Appropriate Low-cost Treatment of Industrial and Domestic Wastewaters'.

WRC Project K5/1362, 'Development and Technology Transfer of IAPS Applications in Upgrading Water Quality for Small Wastewater and Drinking Water Treatment Systems'.

- To undertake scale-up of laboratory findings relating to improved performance of IAPS;
- To research and develop applications of the High Rate Algal Pond as a free-standing Tertiary Treatment unit operation;
- To survey and identify the potential for IAPS applications and technology transfer in low-cost upgrading of small community treatment works in the Eastern Cape Province;
- To evaluate applications of IAPS in the treatment of mine drainage and winery wastewaters;
- To maintain the operation of the Grahamstown IAPS installation as a demonstration unit for purposes of promoting the technology and the development of novel research products.

The results of these studies are detailed in Report 7 Part 4 a and b of the Salinity, Sanitation and Sustainability WRC report series.

2 THE ADVANCED INTEGRATED WASTEWATER PONDING SYSTEM

2.1 DEVELOPMENT OF THE AIWPS PROCESS

In the early 1960's, John Stauff, a design engineer with Carl Yoder and Associates, asked his former engineering classmate William J Oswald to suggest an innovative process design for the City of Helena's new wastewater treatment plant. Since their graduation from the Sanitary Engineering program at UC Berkeley in 1950, Oswald had become deeply involved in research on the role of microalgae in the treatment of wastewater. Based on his research conducted at UC Berkeley's Institute of Engineering Research (now the Sanitary Engineering and Environmental Health Research Laboratory of the Richmond Field Station), his consultations with Al Hyatt, the City Engineer for Woodland, where Oswald and his student Joe Bronson conducted some of the first biogas analysis in oxidation ponds, his work with sludge digestion at the Concord treatment plant, and his familiarity with the work of Guy Parker in Australia, Professor Oswald began to formulate a ponding system design that would provide for the removal of suspended solids, the growth of methane bacteria, photosynthetic oxygenation of primary effluent, and disinfection. Oswald recounts his thinking at the time:

“In conventional wastewater treatment, solids from primary and secondary sedimentation are put into a digester for 40 days. They are then removed, dried, and buried. Why not bury them in the first place? The conventional digester 40-day residence time does not permit complete digestion. It only conditions the sludge to drain and dry quickly. Why not build a Parker-type, deep pond with a volume big enough so that all the settled solids can remain and digest for hundreds of days and put that pond inside a bigger pond where algae could grow and produce oxygen to destroy odours? Why not then have a second pond where algae are grown under optimal conditions of light and mixing? When algae are grown under such conditions, they increase the pH and produce dissolved oxygen. So why not recycle this oxygen for disinfection and odour control? Then settle and remove the algae for use as a fertilizer or animal feed. Finally, why not add maturation ponds for further disinfection prior to discharge or reclamation?”(Oswald, 1987).

The above thinking of Professor Oswald resulted, after more than forty years of research and application, in the development of the AIWPS design. Although many variations of the concept have been investigated and used either separately, or in combination with Waste Stabilisation Pond systems (Mara and Marecos do Monte, 1987; Mara *et al*., 1996), the now trade-marked AIWPS provides possibly the most precisely engineered example of the IAPS approaches to wastewater treatment.

2.2 THE SYSTEM

The AIWPS design, in terms of its unit process operations, is similar to those of conventional wastewater treatment plants. These involve primary sedimentation, flotation, fermentation, aeration, secondary sedimentation, nutrient removal, storage, and final disposal. The methods by which these unit processes are fostered are, however, unique to AIWPS (Figure 2.1).

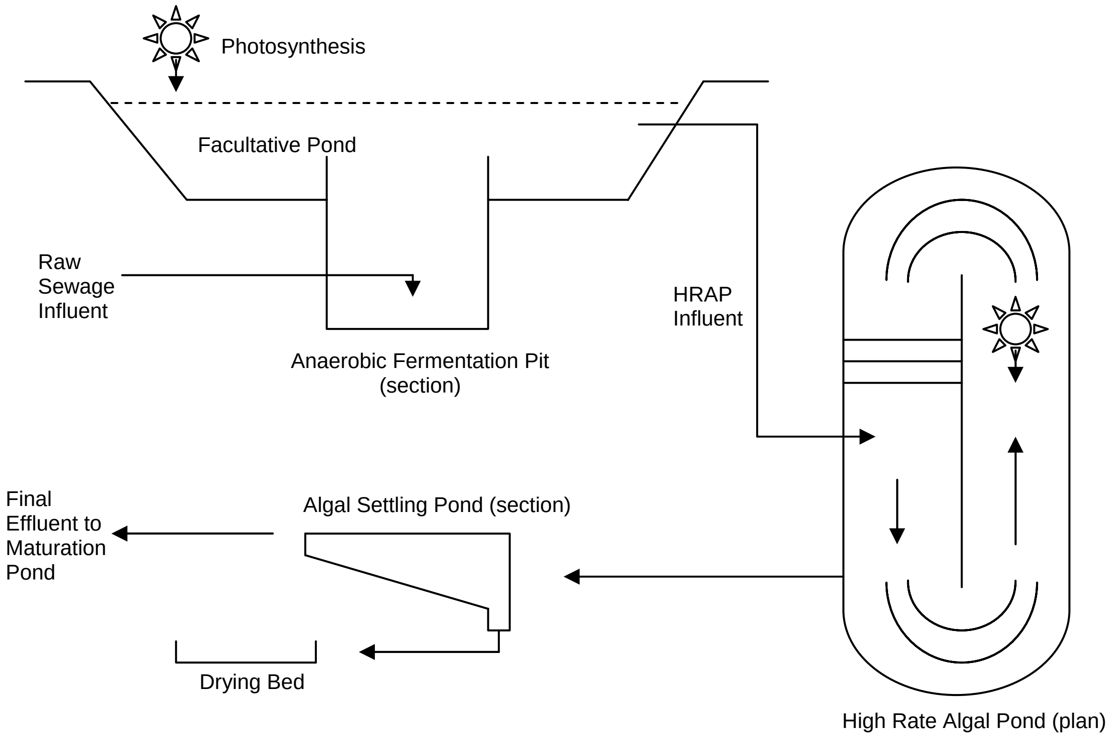


Figure 2.1 Schematic diagram of the principal unit operations associated with the AIWPS Plant design constructed at the Rhodes University Environmental Biotechnology Experimental Field Station, Grahamstown Disposal Works. After Rose *et al*. (2002c).

The system is composed of a four-pond series and for a full discussion of development, concept and design of the AIWPS, see Oswald (1988 a&b; 1995).

2.2.1 Primary Facultative Pond

The first of the four-pond series is the Primary Facultative Pond (PFP) in which the anaerobic bottom zone is overlaid with surface aerobic waters, creating thereby two functionally separate compartments in the pond. A chronic problem with conventional WSP has been that during windy periods, inversions and water mixing occurs, which carries oxygenated waters into the anaerobic zone, and inhibits the anaerobic digestion processes. Fermentation pits are constructed in the base of the PFP in which solids sedimentation and anaerobic processes take place, and these are designed with a surrounding berm or wall to prevent the intrusion of oxygenated water.

Raw waste is introduced near the bottom of these pits and, since they are deep, most of the settleable solids remain within the pits. Overflow velocity is designed to be low enough (less than 1.5 m/day) so that most helminth ova and other parasites also remain in the pit. Particulate solids are lifted by bubbles of biogas which form on and in them, but as these rise the bubbles expand and break away leaving the solids to resettle. The result is that solids accumulate within the pits creating an anaerobic sludge blanket through which all of the wastewater must flow.

Carbon dioxide in the biogas becomes available to support algal growth in the upper layers of the PFP, and the generation of photosynthetic oxygen provides, in part, for the aerobic function of this compartment. The upper aerobic layer is also responsible for particularly effective entrapment and oxidation of odour causing compounds. The low-odour characteristics of these systems has enabled their construction in close proximity to urban developments, and has focussed interest in their use in strategies where smaller dispersed wastewater treatment plants replace large centrally located treatment works.

To deal with floatables, PFP are designed with down-wind concrete scum ramps where floatable trash can be cast up by the wind to dry. These substances are largely inert, light in weight and low in odour. They can be collected periodically with a loader for burial or disposal as a solid waste. Since overflowing PFP is constant in depth, bank erosion is best controlled with a paved water line.

Depth is regulated in the PFP by the level of the invert of the outlet pipe. The inlet of this pipe should be located about 1 m below the surface to avoid transfer of floatables into the secondary pond.

2.2.2 High Rate Algal Pond

The HRAP provides the most effective design for the second unit operation of the AIWPS. This paddle-mixed raceway requires short retention times of 3-5 days, and produces much more dissolved oxygen than a conventional secondary Facultative Pond. Algae in these systems form stable flocs, which settle readily, and > 80% of the algal cells present in the system may be removed in a short residence time Algal Settling Pond (ASP). This biomass has a low respiration rate and may remain concentrated in the bottom of the ASP for a period of weeks or even months without releasing significant amounts of nutrients.

Algal photosynthesis in the HRAP tends to raise the pH of the treated waters, and a pH of 9.2 for 24 hours will provide a 100% kill of *E.coli* and most pathogenic bacteria and viruses. It is not uncommon for the HRAP to reach pH levels of 9.5 to 10 during the day, so a high rate of disinfection is normally achieved.

Because a surplus of dissolved oxygen is produced in the HRAP (usually several times the applied Biological Oxygen Demand), some of this partly-treated effluent is used to overlay the PFP with warm oxygen-rich water. This absorbs odour-causing compounds generated in the fermentation pits, and assures the presence of oxygen-producing algae in the surface waters of the PFP.

2.2.3 Algal Settling Pond

As indicated above, the HRAP should be followed by an ASP or some other method of removing algae. If the water from the HRAP is to be used for irrigation, algae need not be removed, but settling and storage should be sufficient to achieve a bacterial Mean Probable Number (MPN) $<10^3$ which is sufficient for irrigation waters.

Harvesting and removal of the algal biomass component of the treated water is required prior to discharge into public water bodies, although the algae may be beneficial to the food chain in the local ecosystem. There is little evidence that this biomass is harmful in moving waters. As uses for waste-grown algae become established, such as its use in animal feeds, fatty acid, plant hormone and pigment extraction, the potential exists for adding value to the treatment process and providing an incentive for integrated resource management in wastewater treatment.

Natural sedimentation of algae from a paddle-wheel-mixed HRAP is sufficient to remove >80% of the algae, but if higher degrees of removal are required then mechanical harvesting is indicated. In the Grahamstown AIWPS the settled algae from the ASP is pumped onto underdrained sand beds for drying and harvesting for evaluation purposes.

2.2.4 Maturation Pond

Where AIWPS-treated waters are to be discharged under conditions leading to possible human contact, storage for 10 to 20 days in a deep maturation pond may be used instead of chlorination, and will provide adequate reduction of the bacterial count. The maturation ponds provide valuable wetland environments and are often occupied by wild water fowl, and many other forms of wildlife. These may, however, also impart high but innocuous (MPN) loads to maturation pond effluents.

3. INVESTIGATION OF IAPS APPLICATIONS – STUDY METHODOLOGY

3.1 INTRODUCTION

The IAPS research and demonstration plant described in this study (Figure 3.1) was constructed at the Rhodes University Environmental Biotechnology Experimental Field Station, which is located at the GDW. The design rationale used for the plant was the AIWPS concept provided by Prof William Oswald and Dr. Bailey Green, consulting as Oswald Green, and is dealt with in detail by Rose *et al.*, 2002c. The plant was conceptualised to fill the requirements of both an experimental research facility and also for the demonstration of various IAPS concepts. (Given the hybrid operation the system, the use of the trade marked AIWPS term has been avoided). On Professor Oswald's advice the plant was sized at what he considered the minimum that would still be able to provide credible performance data suitable for engineering scale-up requirements. Hence it is considered that the 9-year performance report that follows is representative of what might be anticipated in a full-scale plant operating under comparable South African conditions.

Figure 3.1: Aerial photograph of the Integrated Algal Ponding System research plant at the Rhodes University Environmental Biotechnology Experimental Field Station, showing the primary facultative pond (top) and high rate algal ponds, algal settling ponds and drying beds below.

3.2 DESIGN

The plant was sized to treat the liquid wastes of 500 person equivalents (PE), and in this calculation average *per capita* water consumption and disposal figure of approximately 150 ℓ.day^{-1} was assumed. Accordingly, the design flow was calculated at 75 m^3.day^{-1}. With an ultimate Biological Oxygen Demand (BOD_{ult}) assumed to be 80 g BOD_{ult} per person per day, the organic loading to the system was taken as 40 kg.day^{-1}.

Apart from preventing possible intrusion of oxygen into the anaerobic layer of the PFP, the fermentation pit is located in the base of the PFP to ensure complete fermentation of solids and to eliminate sludge handling over a period of 20 to 30 years. With this in mind, the pit was designed using a volumetric capacity of 0.45 m^3 *per capita*, which is 15 times the standard *per capita* value (0.03 m^3) used in conventional sewage sludge digesters. The volume of the fermentation pit is therefore 225 m^3, giving a HRT of 3 days. A perimeter berm was constructed around the pit to preclude wind-induced mixing and therefore penetration of oxygenated water into the anaerobic fermentation zone. The depth of the pit is 4.5 m, 3 m below the bottom of the PFP and 1.5 m above the bottom. In order to ensure the settling of solids, helminth ova and other parasite cysts from the waste stream, the upflow rate should be less than 2 m per day. A daily flow of 75 m^3 over the pit area of 50 m^2 results in an upflow velocity of 1.5 m day^{-1}. This adds a 25% margin of safety over the recommended 2 m.day^{-1}figure.

The PFP, depicted in Figure 3.2, surrounds the fermentation pit and has a volume of 1500 m^3, to give a maximum HRT of 20 days. The surface area of this pond is 840 m^2, which, assuming a conservative BOD removal in the fermentation pit of 50%, will give an organic loading of 0.024 kg BODult/m^2/day. The water depth in the PFP is kept constant with a set overflow level and this water line is protected from erosion and weed growth by a concrete scum rack. The water transferred to the HRAP is taken from a depth of 1 m below the surface to avoid depletion of the surface algae layer and carry over of floating solids.

Figure 3.2: The primary facultative pond showing the access bridge constructed for access during monitoring and experimental work.

Using an assumed conservative BOD loading to the HRAP from the PFP, the depth in the two HRAPs was set at 30 cm (Figure 3.3). The total volume of these ponds is, therefore 150 m^3, with a surface area of 500 m^2. Using adjustable overflow weirs from the PFP off-take, the hydraulic loading, and thus HRT in the HRAP can be adjusted infinitely (up to a maximum of 75 $m^3.day^{-1}$) for experimental purposes, but was generally run between 3 and 6 days. The algae floc is kept in suspension in the HRAP by a paddle wheel that maintains a linear velocity of 30 $cm.sec^{-1}$ in the pond.

As was mentioned in chapter 1, Oswald (1990) recommends a recirculation of oxygen rich water from the HRAP back to the surface of the PFP. In the EBRU demonstration plant, however, it was found that this does not work as intended due to the algae from the HRAP settling to the bottom of the PFP when not mixed. This recirculation was, therefore, discontinued and the algae growth in the PFP was left to develop on its own, which resulted in an algae consortium better suited to the surface environment of the PFP. Odour control in the PFP has, nevertheless, been extremely effective.

The original design of the raceways provided only one semi-circular wall dividing the flow around the bends at the pond ends. This, however, caused a quiescent zone due to the water moving faster against the outer than against the inner dividing wall. This zone of reduced flow velocity results in settlement of algae floc on the bottom of the raceway at this point. For optimum performance, a constant linear velocity is required throughout the width of the raceway, including the ends. This is accomplished by separating the water stream by channel dividers as shown in Figure 3.3.

Figure 3.3: The high rate algal ponds showing channel dividers used to prevent quiescent zones of flow on the leeward side of the inner pond dividing wall.

The ASPs, shown in Figure 3.4, were designed to provide 0.5 day HRT, using a length to width ratio of 1:6.43, which is slightly more than the recommended 1:6 (Oswald, 1994). This HRT results in near complete removal of the algal floc and 80%+ of the planktonic algae which settle and form a slurry with solids concentration of 3-5%. At a 3% slurry concentration, the maximum daily volume slurry generated would be 0.25 m^3. At a depth of 5 cm on a sand bed surface, the daily area required for drying the slurry would be 5 m^2 (Figure 3.5). If a 7 day drying period is allowed, the drying bed surface area required would be 35 m^2. Another rule of thumb estimate used for sizing drying beds is 10% of the total HRAP area, which would give a sand bed area of 50 m^2. To compensate for these different size calculations, four 10 m^2 drying beds were built to give a total area of 40 m^2.

Domestic sewage flow will vary widely over the course of a day; however, because of long residence times and a large buffer capacity, IAPS have the advantage that they can be designed based on average flow rates. Considering the practical difficulties in attempting to simulate a variable daily flow, the plant was operated with a constant inflow to the PFP. This was achieved by pumping raw sewage from the GDW head of works. Figure 3.6 gives the orientation of the research plant and GDW. As the pump delivers approximately 165 $m^3.day^{-1}$, the excess wastewater, i.e. over and above the 75 m^3 required for the operation of the IAPS, needed to be returned to the GDW. This flow was regulated by an adjustable bypass line that is used to control the level of water in the splitter box, measured at a V-notch weir, and hence also the flow to the PFP.

Figure 3.4: Algal settling pond with 0.5 day hydraulic retention produces an algal slurry of around 3% solids.

Figure 3.5: The four algal drying beds located between the two algal settling ponds.

Figure 3.6: The primary facultative pond and the integrated algal ponds of the research plant shown at left in relation to components of the Grahamstown Disposal Works shown at right.

3.3 OPERATION OF THE PONDING SYSTEM

Operation of the system varied in some respects during the course of the study period depending on the specific research objectives under investigation. However, the hydraulic loading to the fermentation pit and PFP remained constant at 75 $m^3.day^{-1}$. There was no control over the chemical oxygen demand (COD) in the raw sewage and the organic loading did, therefore, fluctuate. Experimental adjustments were only made with investigations in the optimisation of HRAP operations. During commissioning and the first 4 years of the study these two ponds were operated in parallel, each unit taking half of the PFP effluent i.e. 37.5 $m^3.day^{-1}$ (Figure 3.7, A). This equates to a 4 day HRT in each HRAP.

In February 2000, the system was reconfigured and HRAP 1 was retrofitted to enable the investigation of the HRAP as an independent unit operation in the tertiary treatment and polishing of final effluent from any conventional or other sewage treatment facility. This application became known as the Independent High Rate Pond (IHRAP), with the final treated water used in the study being sourced initially from the GDW (Clark, 2001). Later, final treated water from the IAPS plant was also used in the evaluation of the IHRAP concept.

Initially a HRT of 5 days was used for this study (Figure 3.7, B), but this was also varied with the research protocol requirements as discussed in the results section. During this time the PFP effluent was split so that HRAP2 continued to operate as loaded from the PFP during the parallel, averaged configuration shown in Figure 3.7A, i.e. receiving its design load. The excess flow was wasted to drain. HRAP 1 was then operated as an IHRAP receiving final treated water from the GDW. The ponds were operated in this manner until June 2003, when the IHRAP experiment treating GDW final effluent was completed.

In June 2003 HRAP1 was reincorporated into the IAPS cascade but configured to operate in series with HRAP2 (Figure 3.7, C) rather than in parallel as in the earlier phases of the project. The performance of HRAP2 was evaluated, now operating as the IHRAP and receiving effluent from HRAP1 after settling algae in the ASP. Retention times during this last period were varied between 3 and 6 days.

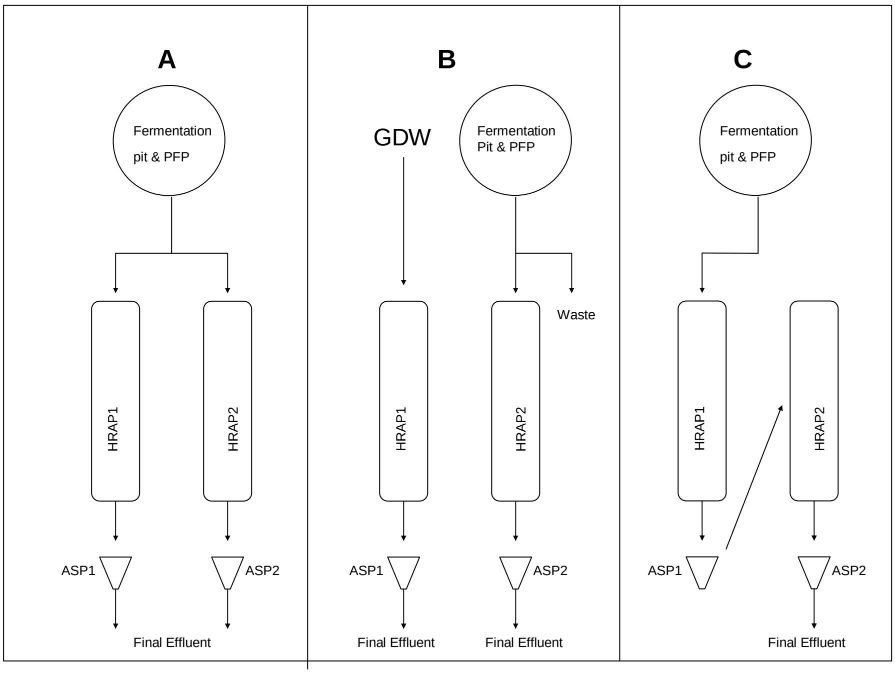


Figure 3.7: Various configurations of the HRAP unit operation evaluated in this study.

3.4 ANALYTICAL METHODS

Chemical Oxygen Demand, ammonia as NH_3-N, nitrate as NO_3-N and orthophosphate as PO_4-P were analysed using Merck Spectroquant SQ118 and Merck analysis kits. Samples were filtered through Whatman GF/C microglass filters. The pH was measured with a Cyberscan, 2500pH meter, standardised at pH 4.0 and pH 7.0 with SAARCHEM standard buffer solutions. Algae genera were identified using a light microscope at 10-100x magnification.

3.5 MICROBIOLOGICAL ANALYSIS

Bacteriological analysis was carried out using the filtration method described in Standard Methods (American Public Health Association 1998). Initially McConkey agar (Merck) was used as the culture medium. Whilst this gave clear results at high dilutions, at the low dilutions necessary for the final effluent, the residual algae in the water interfered with the clear differentiation of faecal coliform colonies. Using Chromocult agar (Merck), the enumeration of

Escherichia coli and total coliform colonies was successful and this method was, therefore, adopted for the analysis of indicator bacteria.

3.6 SAMPLING PROTOCOLS

Samples were drawn as follows: from the incoming raw sewage, from the PFP effluent, HRAP1, ASP1 effluent, HRAP2 and ASP2 effluent. These were sampled daily for nutrients and COD and twice weekly for bacterial assay. For some of the more short-term studies undertaken within the broader study, either composite or individual samples were taken over a 24 hour period. Analyses were generally carried out within half an hour of sampling. Where this was not possible, samples were refrigerated at 4˚C for no longer than 24 h.

3.7 DATA PROCESSING

The data to be described in the following chapters has been presented in several formats, depending on which aspect of the system was under investigation. Chapter 4 deals with the fermentation pit and PFP over the 9-year study. These units of the system underwent very little experimental manipulation during the study period and the results for the whole period have, therefore, been averaged, or reported as a continuous data set, where relevant. As will become clear from the results, there was a shift in COD strength in the raw wastewater and the data have, therefore, been divided into before and after this change occurred.

Chapter 5 details the performance of HRAP's 1 and 2, during the period when this operation was run as a single stage treatment with the performance averaged between the two high rate ponds operating in parallel.

From June 2003, the HRAP was reconfigured into a two-stage unit operation in order to optimise the tertiary treatment capacity of the system, particularly with respect to nutrient removal and disinfection. Because of this process modification, the results for this period have been excluded from the rest of the data and described separately in chapters 6-8.

Chapters 4 to 8 give an account of each individual unit in the IAPS. While the independent mechanisms are of interest in optimising their respective function, the concept behind the system is that a pond cascade provides the required complete treatment. The data have, thus, been depicted in chapter 8 in a manner as to describe the overall performance of the IAPS as an alternative, wastewater management technology.

4. A 9-YEAR STUDY OF IAPS OPERATION 1: THE PRIMARY FACULTATIVE POND

4.1 INTRODUCTION

The PFP in the IAPS concept, is designed to promote the growth of three distinct microbial consortia i.e. a deep anaerobic stratum supporting fermentative and methane producing bacterial consortia, an overlying facultative stratum supporting both aerobic and anaerobic growth, and finally by an aerobic surface layer heavily populated by green algae, cyanobacteria and photosynthetic bacteria. According to Green *et al*. (1995), methane fermentation is one of the most efficient ways to manage the influent organic load. The deep anaerobic pit is designed to prevent oxygen ingress and thus provide ideal conditions for methanogenesis and organic removal, measured here as COD reduction. While there are also mechanisms which act to decrease pathogen and nutrient loads in the PFP, the primary function of this unit operation remains COD removal.

Although different research students have been involved in the investigation of the various aspects of the IAPS over the 9-year study period, the operation of these units remained largely consistent. The results presented in this chapter are, therefore, a collation of all the data recorded by EBRU staff and students over the 9-year monitoring programme.

4.2 MATERIAL AND METHODS

The materials and methods for this section of the study are as described in chapter 3.

4.3 RESULTS AND DISCUSSION

4.3.1 Chemical Oxygen Demand Removal

The performance of the PFP during commissioning is discussed in more detail by Rose *et al*. (2002c) in the WRC report no: TT 190/02, "Integrated Algal Ponding Systems and the Treatment of Domestic and Industrial Wastewaters, Part 1: The AIWPS Model". Initially, the fermentation pit proved quite unstable with a low pH, high volatile fatty acid (VFA) levels and failure to accumulate a substantial sludge blanket. The stabilisation of the anaerobic processes was resolved by reseeding the pit with sludge from the GDW anaerobic digesters. Figure 4.1 shows the reduction of VFAs and subsequent consistency of these levels after about week 40. Despite the variable performance of the fermentation pit, a total (unfiltered) Chemical Oxygen Demand removal of between 60 and 85% was consistently achieved in the PFP during the first 50 weeks of operation (Rose *et al*., 2002).

Due to a large CODt variation in the sewage feed (raw), between 1997 and 1999, the results since commissioning in 1997 have been presented as two separate data sets: from commissioning until October 1996 in Figure 4.2, and 1999 until October 2004 in Figure 4.3. The results for the earlier period indicate an average raw CODt of 2 340 mg.ℓ^{-1}, while the average since the beginning of 1999 was 1 013 mg.ℓ^{-1}. During 1998, the Leather Industries Research Institute (LIRI) tannery was closed and, consequently, the effluent entering the sewer from this tannery was discontinued (Rose, pers. comm.) and this could possibly account for the drop in COD_t entering the system after that time.

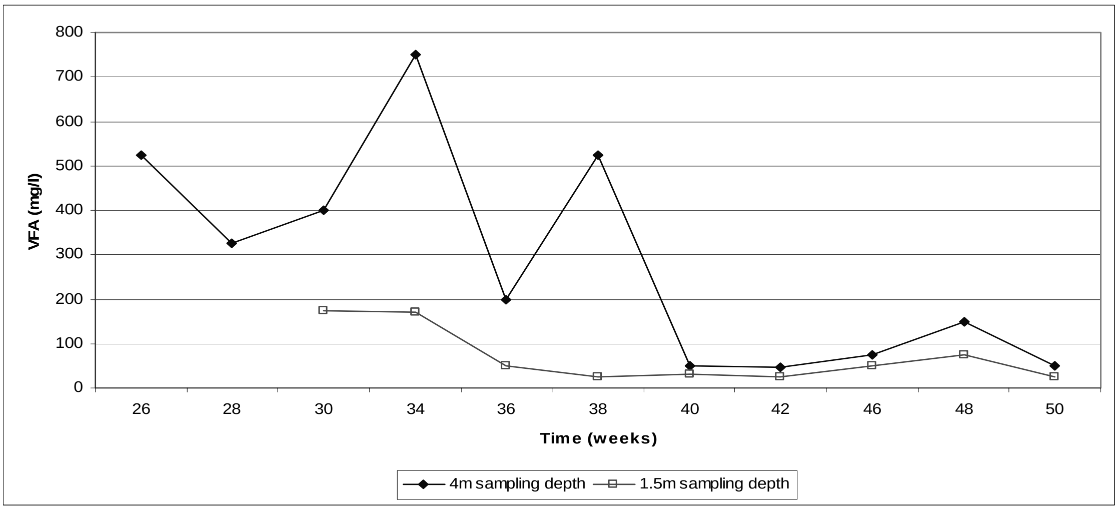


Figure 4.1: Stabilisation of volatile fatty acid levels in the fermentation pit, during commissioning and after reseeding with anaerobic sludge from the Grahamstown Disposal Works.

From Table 4.1, it is evident that, at times, the PFP effluent had high COD_t levels. The average COD_t removal during this time was 68.4%. As mentioned above, the high COD_t levels in the feed are possibly due to tannery effluent entering the treatment works. The low COD_t levels in the feed have also been correlated with incidents of high rainfall, during which storm-water enters the Grahamstown sewage system in large volumes (Dekker, 2002). The dilution effect of rainwater on sewage has been identified in other studies e.g. Schetrite and Racault (1995).

Table 4.1: Primary facultative pond unfiltered chemical oxygen demand removal from February 1996 to October 1997.

	Maximum ($mg.\ell^{-1}$)	Minimum ($mg.\ell^{-1}$)	Mean ($mg.\ell^{-1}$)	Standard Deviation	Percentage Removal
Sewage Feed	7 200	250	2 340	1 215	N/A
PFP effluent	5 040	88	741	628	68.4

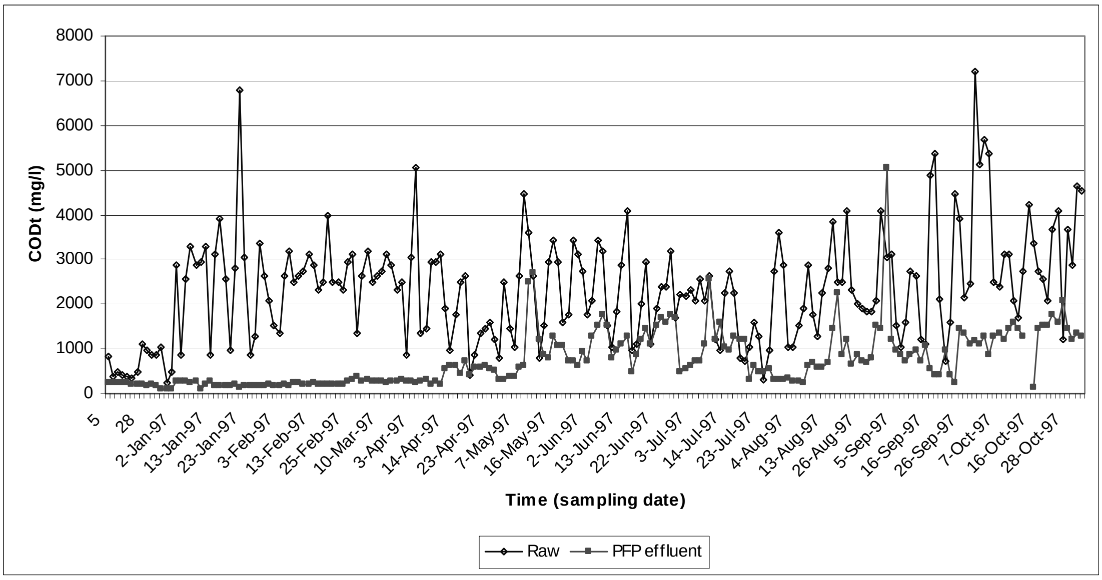


Figure 4.2: Unfiltered chemical oxygen demand removal in the primary facultative pond from commissioning until October 1997.

Although the high COD_t levels of 1997 are not repeated in the later data set, summarised in Table 4.2, there is still a wide COD_t variation in the sewage feed, with values fluctuating between 312 mg.ℓ^{-1} and 6 535 mg.ℓ^{-1}. Once again the low COD_t values can be correlated with rainfall incidents where dilution occurs, while the peak values can be attributed to the dumping of vacuum tanker loads into the GDW head of works from collection sumps in Grahamstown. The mean COD_t load of 1 162 mg.ℓ^{-1} was higher than the design load, but is comparable to levels measured in other studies in Africa (Horan, 1996) this is about double that found in areas of higher water consumption such as in parts of Europe and North America (Nurdogan & Oswald, 1995; Racault *et al*., 1995; Horan, 1996). Some of the drier areas of Spain have an average COD_t of 1 500 mg.ℓ^{-1} (Soler *et al*., 1995), while in the Yemen Republic the average COD_t in the raw sewage was reported by Veenstra *et al*. (1995) to be 1600 mg.ℓ^{-1}.

The mean percentage COD_t removal in the PFP between February 1999 and October 2004 was 73.5%. Oswald (1991a) reports a similar removal efficiency for the PFP at St. Helena, California. However, this performance is better than PFPs operating in Morocco (Ouazzani *et al*., 1995) and Tanzania (Kayombo *et al*., 2002), where COD_t removal rates of 50% and 66% respectively were recorded.

Table 4.2: Primary facultative pond unfiltered chemical oxygen demand removal between February 1999 and October 2004

	Maximum (mg O_2.l^{-1})	Minimum (mg O_2.l^{-1})	Mean (mg O_2.l^{-1})	Standard Deviation	Percentage Removal
Sewage Feed	6 535	312	1 162	643	N/A
PFP effluent	820	82	308	103	73.5

The buffering capacity of the PFP is illustrated in Figure 4.3. From the end of June 2004, high levels of up to 6 500 mg.ℓ^{-1} COD_t were measured in the sewage feed. These high levels did not appear to substantially affect the COD_t in the PFP effluent, which averaged 326 mg.ℓ^{-1} during this period, only slightly higher than the mean value recorded over the previous six years of monitoring. The spikes in the feed COD_t were also not reflected in the PFP effluent which provides an indication of the pronounced buffering capacity of these systems.

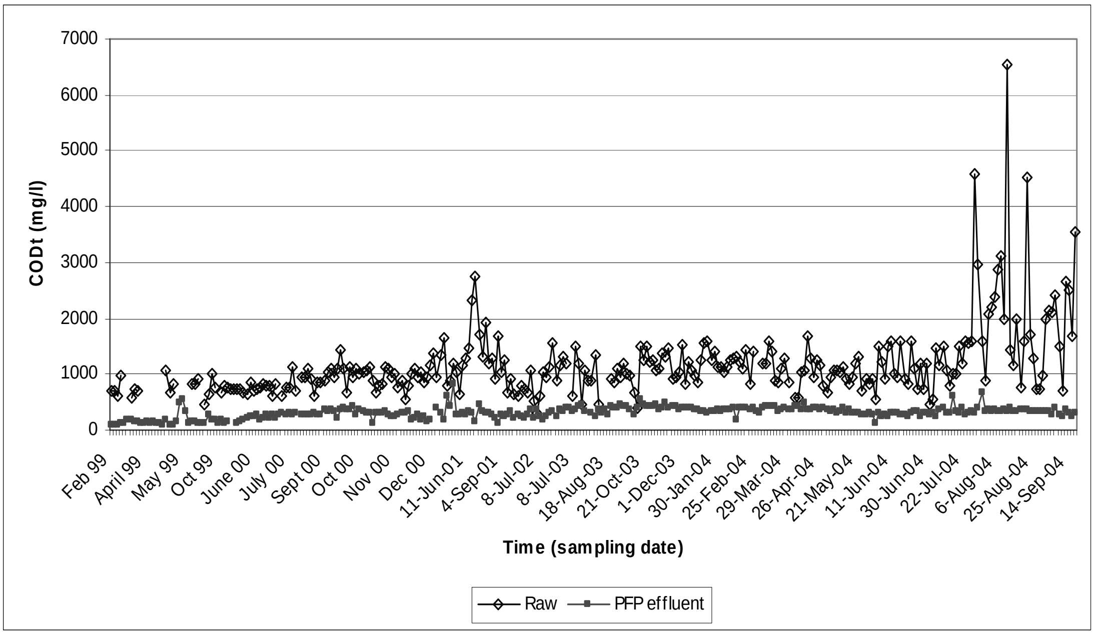


Figure 4.3: Unfiltered chemical oxygen demand removal in the primary facultative pond between February 1999 and October 2004

Removal figures for soluble (filtered) Chemical Oxygen Demand (COD_s) (Figure 4.4) were not as impressive as those for COD_t. During commissioning and the monitoring period up to October 1997, the ponds were not analysed for filtered COD_s. Data, therefore, only exist from February 1999 to October 2004. During this time the average percentage COD_s removal was only 20%. Peak loads in sewage feed COD_s were, however, effectively absorbed in the PFP and are not reflected in the PFP effluent. This difference in filtered and unfiltered removal is due to the large percentage of COD_t being in the form of suspended solids, which are settled out in the fermentation pit. This fraction would, however, be excluded by the 45µm filters used in the measurement of COD_s and the total removal rate is, therefore, not indicated in the COD_s figures.

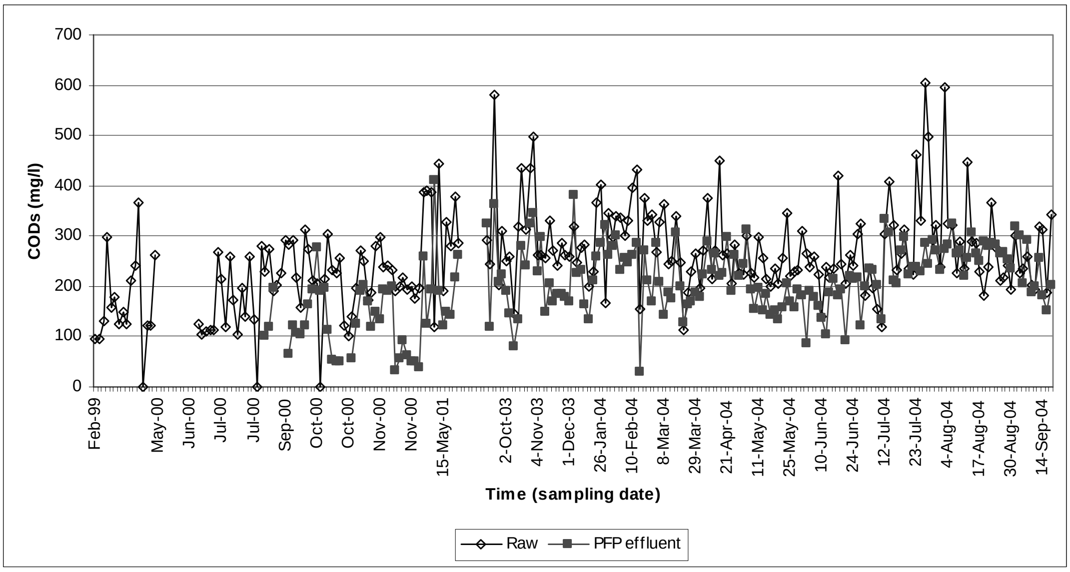


Figure 4.4: Filtered (soluble) chemical oxygen demand removal in the primary facultative pond for the period February 1999 to October 2004

The design of the fermentation pit takes advantage of the slow biomass generation, associated with anaerobic digestion, and provides an environment for an extended sludge age. This results in an infrequent desludging requirement. After nine years of operation, the Grahamstown demonstration plant had a non-degradable sludge build up of an average of only 300 mm in the fermentation pit.

Methanogens are very sensitive to environmental conditions, in particular to changes in pH, and generally tolerate of a pH range between pH 6.2 – 8.0. If the VFA production rate is greater than the rate of methanogenesis, the pH will fall, inhibiting and ultimately killing the methanogens (Horan, 1996). This will result in treatment failure and the pond becoming putrid and releasing objectionable odours (Oswald, 1994). Apart from a brief drop in pH during commissioning, the pH of the pit and the PFP has remained stable, and within the optimal methanogen range. The average of pH readings for the PFP effluent over the monitoring period was pH 7.7. Figure 4.5 illustrates the consistency in the pH of the PFP effluent, remaining between pH 6.5 and pH 8.5 over a nine month period, from January 2004 to September 2004.

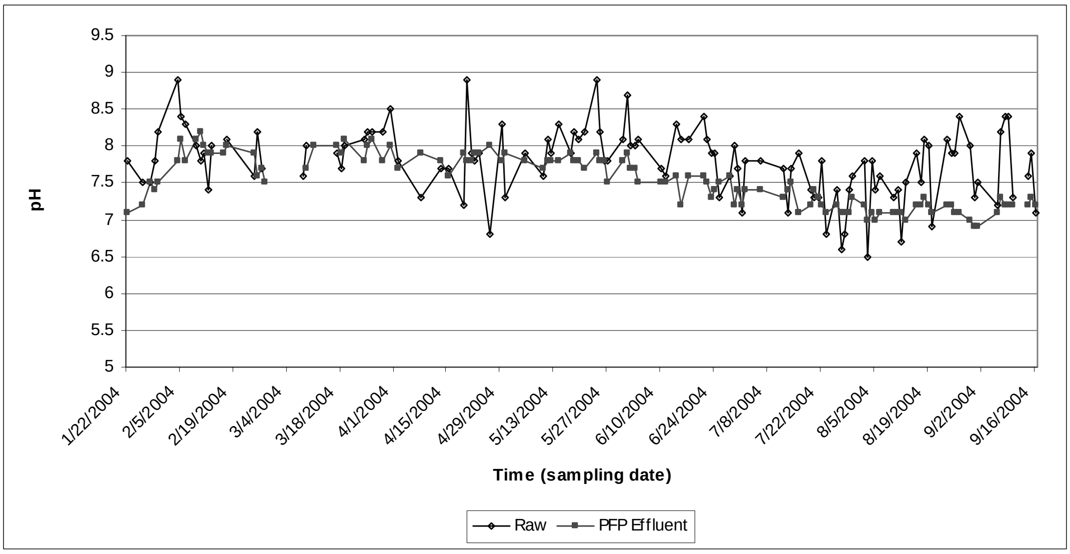


Figure 4.5: Comparison of raw and primary facultative pond effluent pH over the sampling period January to September 2004

Although the majority of the oxygen demand is eliminated in the fermentation pit, further removal also occurs in the oxygen-rich upper layers of the PFP, where organic material is either oxidised to CO_2 or assimilated into biomass (Henze *et al*., 2002). This is similar to any other aerobic treatment, with the exception that the oxygen is supplied by algal photosynthesis and not by mechanical means (Horan, 1996).

4.3.2 Nutrient Removal

The principle function of the PFP is the removal of the wastewater oxygen demand. Nutrients are, however, also cycled through this pond. Nitrogen and phosphate removal efficacy of the PFP was found to be poor, as has been the case in a number of other studies on primary ponds (Ouazzani *et al*., 1995; Soler *et al*., 1995; Veenstra *et al*., 1995). Table 4.3 and Figures 4.6 to 4.9 present a comparison of the nutrients present in the sewage feed (raw) and PFP effluent.

Table 4.3: Comparison of mean nutrient levels in the sewage feed (raw) and primary facultative pond effluent. Standard deviation shown in parentheses.

	P as mg.ℓ^{-1} PO_4^{3-}	TKN	N as mg.ℓ^{-1} NO_3^-	N as mg.ℓ^{-1} NH_4^+
Raw	17.8 (9.8)	128.8	6.9 (13.2)	11.4 (17.4)
PFP effluent	14.6 (10.1)	58.4	5.8 (9.5)	11.4 (13.2)
% Removal	18	54.6	15.9	0

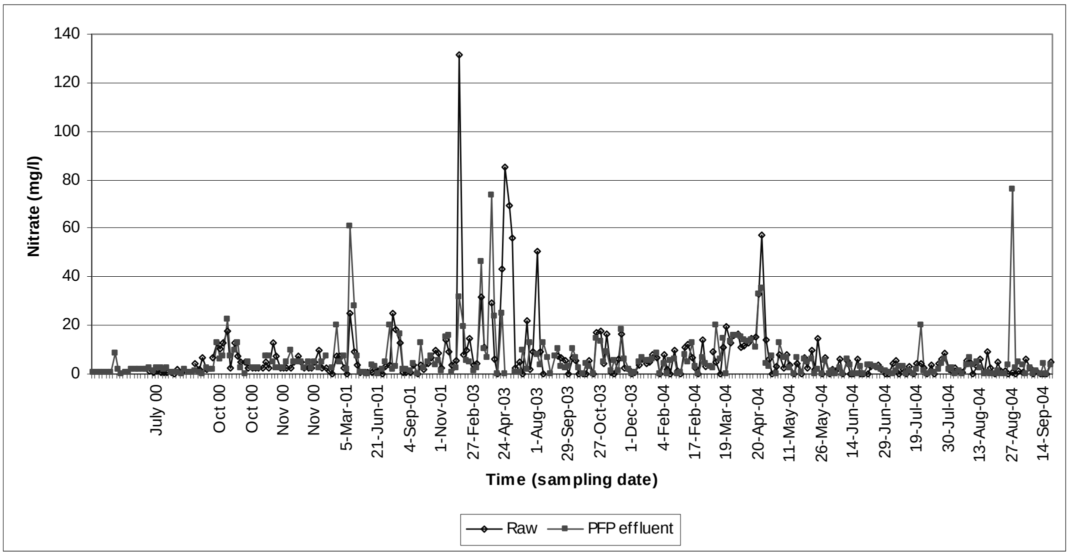


Figure 4.6: Comparison of nitrogen as mg.ℓ$^{-1}$ NO_3^- in the sewage feed (raw) and primary facultative pond effluent during the period May 2000 to September 2004.

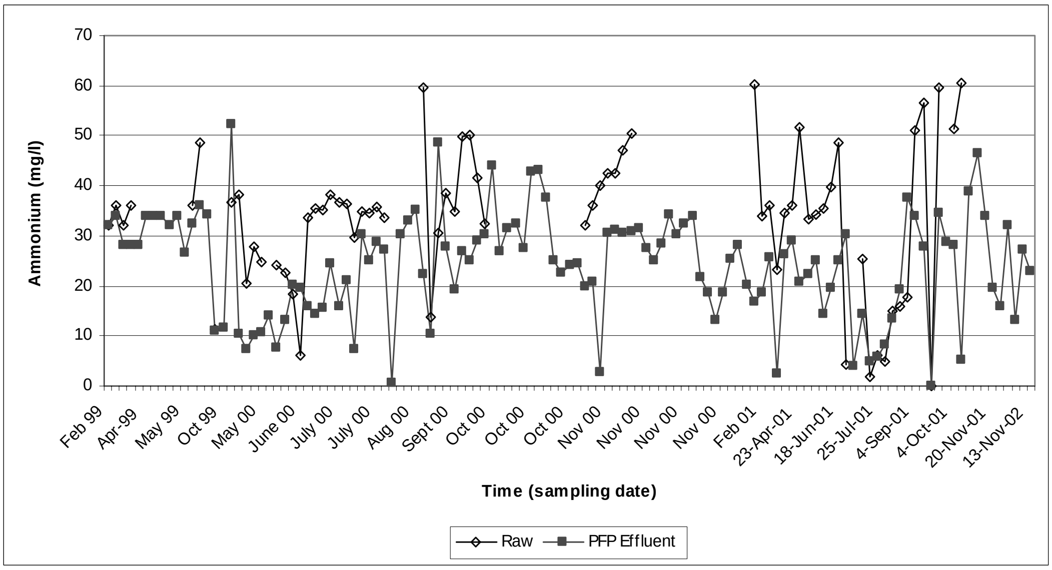


Figure 4.7: Comparison of nitrogen as mg.ℓ$^{-1}$ NH_4^+ in the sewage feed (raw) and primary facultative pond effluent during the period February 1999 to November 2002.

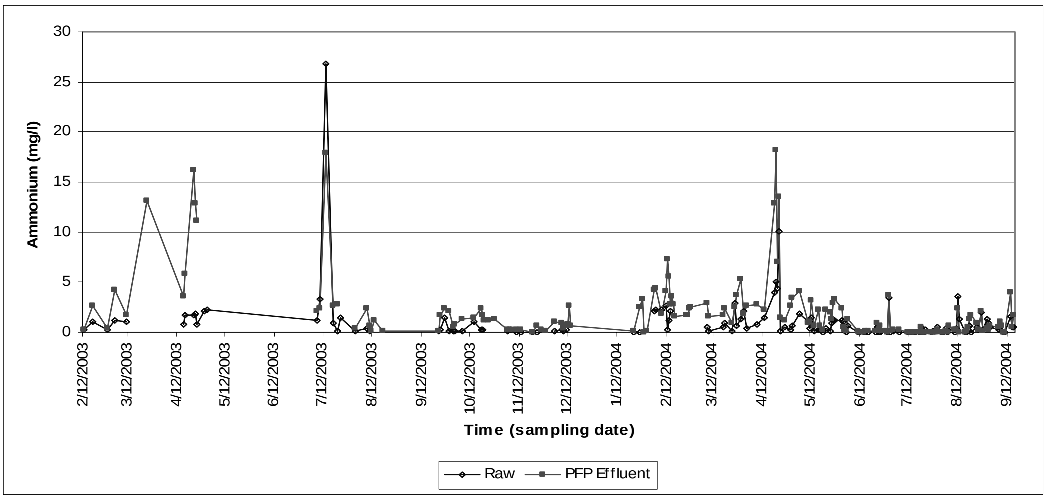


Figure 4.8: Comparison of nitrogen as mg.ℓ$^{-1}$ NH_4^+ in the sewage feed (raw) and primary facultative pond effluent during the period February 2003 to September 2004.

As can be seen from the results, very little of the incoming nitrogen, expressed as Total Kjeldahl Nitrogen (TKN) was in the form of either nitrate or ammonium and should, therefore, still be in the form of organic nitrogen. The ammonium results have been divided into two series as there appears to have been a significant reduction in the ammonia content of the sewage feed between the end of 2002 and the beginning of 2003. Although the mean values indicate that there was no change in ammonium levels between the raw and PFP effluent, during the earlier phase illustrated in Figure 3.7, there was an average reduction of 27.8%. The later period (Figure 4.8), however, indicates a mean increase of 10% across the PFP. Apart from infrequent peaks, the nitrate in the sewage feed was relatively low and there was only a 15.9% decrease in this level in the PFP effluent.

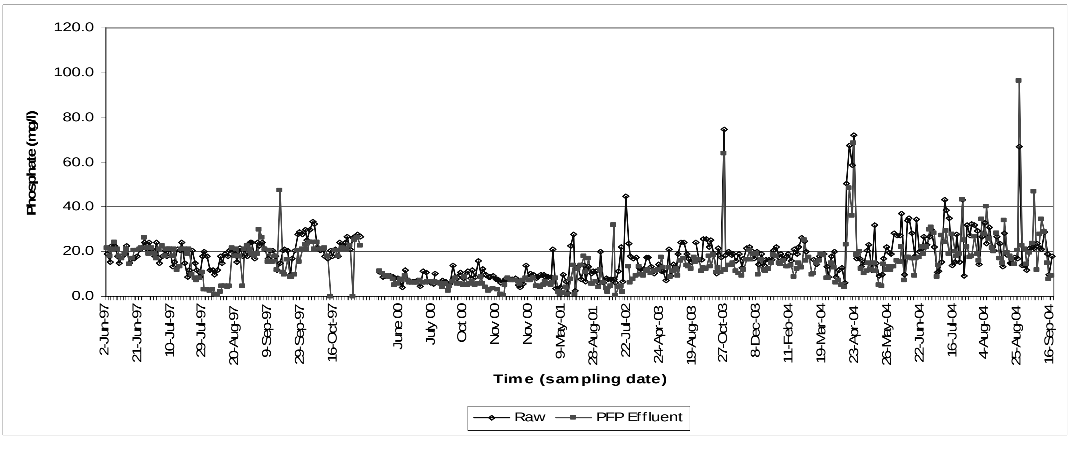


Figure 4.9: Comparison of phosphorous as mg.ℓ$^{-1}$ PO_4^{3-} in the sewage feed (raw) and primary facultative pond effluent during the period June 1997 to September 2004.

According to the mean figures, represented in Table 4.3, an 18% decrease in phosphate across the PFP was observed. From Figure 4.9, however, it can be seen that this decrease was possibly more a smoothing effect of the PFP on the peaks in the phosphate levels of the incoming sewage rather than continual removal. The measured phosphate levels in the PFP effluent followed those in the raw water closely, without major elimination taking place.

4.3.3 Nutrient Removal Mechanisms

Although over 50% of the incoming TKN was removed in the fermentation pit and PFP, there was no corresponding change in either the nitrates or ammonium levels. This could be attributed to a number of different factors. There is the possibility that nitrification, followed by rapid denitrification was taking place, with the nitrogen lost as gaseous N_2 (Mara & Pearson, 1986). Green *et al*. (1995b) found that up to 13% of the gas emitted from in-pond, anaerobic digesters, was in the form of N_2. High organic matter concentration and lack of oxygen in the fermentation pit would, however, inhibit the growth of nitrifying bacteria and it has, therefore, been suggested that biomass assimilation, sedimentation and ammonia volatilisation are perhaps more important removal mechanisms (Reed, 1985; Sezerino *et al*., 2003; Zimmo *et al*., 2003). During the period of higher ammonia levels in the sewage feed, there was also a higher percentage removal of ammonia. This was probably due to volatilisation of the ammonia as Zimmo *et al*. (2003), have reported higher ammonia volatilisation rates when influent ammonia was raised. Maynard *et al*. (1999) have claimed that pH >10 is required for ammonia volatilisation to take place and thus, as this pH level is seldom reached in the PFP, some of the ammonia was possibly also removed by algae and other microbe assimilation in the PFP (Horan, 1996). The period where there was a slight increase in ammonia levels, corresponds to higher organic nitrogen in the influent water. This is an indication that the increase was possibly due to ammonification, involving the degradation of proteins and urea, being more rapid than elimination by volatilisation, assimilation or nitrification (Maier *et al*., 2000).

The small amount of phosphate removed in the PFP can be attributed to assimilation into the biomass of algae and bacterial cells (Surampalli *et al*., 1995). As the required increase in pH for phosphate precipitation (Degrémont, 1991; Maynard *et al*., 1999) does not occur in the PFP, this mechanism presumably does not play a role in this stage of treatment.

4.4 CONCLUSIONS

The fermentation pit and PFP offer a satisfactory primary treatment operation, providing improvements over conventional anaerobic or facultative WSP. The advantages of this unit in the IAPS include the following:

- Effective removal of the incoming sewage organic content, with CODt removal rates of over 70%.
- A valuable buffering capacity, where peaks in CODt load of over 4 000 mg.ℓ^{-1} were absorbed in the PFP, while CODt in the effluent remained below 400 mg.ℓ^{-1}.
- The very slow build up of sludge in the fermentation pit is a major benefit of the system as the need for frequent sludge handling, with the associated costs typical of conventional treatment works, is eliminated.

- Good removal of total nitrogen was achieved in the PFP unit, although nitrate and ammonia levels were not affected.
- Limited phosphate reduction was measured in the PFP unit operation.

5. A 9-YEAR STUDY OF IAPS OPERATION 2: THE HIGH RATE ALGAL POND

5.1 INTRODUCTION

The component processes occurring in the HRAP are summarised in Figure 5.1. The HRAP are designed to optimise algal growth and, therefore, photosynthesis, with its consequent oxygen production and pH increase (Oswald, 1988a). The algal growth and oxygen production contribute to a certain amount of COD removal in the IAPS. In addition the HRAP environment facilitates the effective removal of ammonia, phosphate and pathogens. The parameters required for greatest algal activity include sufficient sunlight and nutrients. These are incorporated into the design by shallow, mixed raceways and an adequate supply of primary treated sewage.

As with the aerobic layers of the facultative pond, COD removal in the HRAP occurs by bacterial oxidation of the soluble organic matter, with the required oxygen supplied by algal photosynthesis. This process is, therefore, sometimes termed photosynthetic oxygenation (Oswald, 1988a). CO_2 produced by aerobic respiration is in turn utilised by the algae in photosynthesis. Although algae produce a net oxygen yield of 1.6 – 1.9 times their cell dry weight mass, the HRAP was designed to produce an algal concentration equal to the influent BOD_{ult} (Rose *et al*., 2002a).

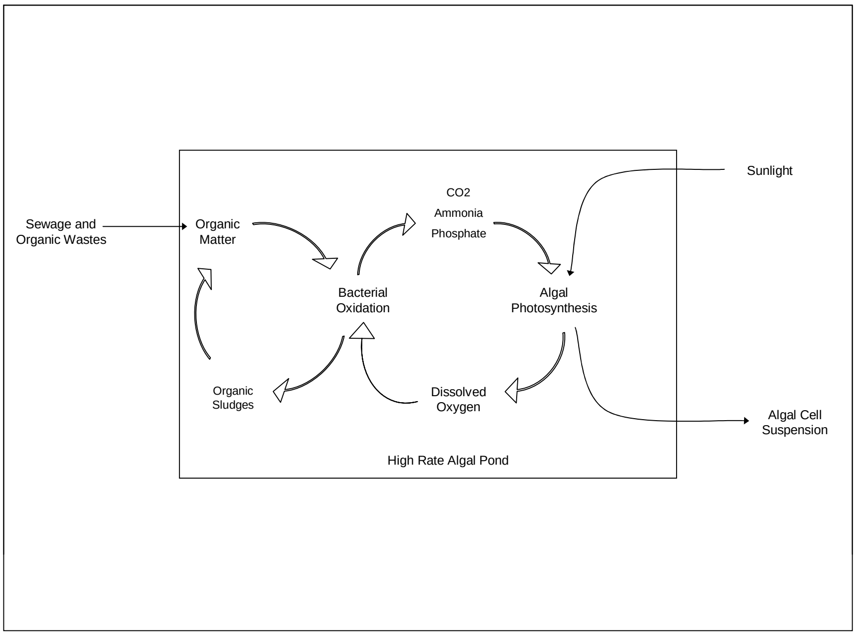


Figure 5.1: Cycle for photosynthetic oxygenation of wastewater. After Oswald (1988a)

5.1.1 Nutrient Removal Mechanisms in the High Rate Algal Pond

Maier *et al*. (2000) define nitrification as the microbially catalysed conversion of ammonia to nitrate, which is predominantly an aerobic, chemoautotrophic process. The best-known nitrifying bacteria are from the genus *Nitrosomonas*, which oxidises ammonia to nitrite and *Nitrobacter*, which oxidises nitrite to nitrate (Maier *et al*., 2000). The main steps are shown below:

$$NH_4^+ + O_2 + 2H^+ \rightarrow NH_2OH + H_2O \rightarrow N- + 5H^+ \quad (1)$$

$$NO_2^- + 2O_2 \rightarrow 2NO_3^- \quad (2)$$

Many biological systems remove nitrate by exploiting denitrification, where nitrate is microbially reduced, through various inorganic forms to gaseous N_2 that is then released to the atmosphere (Maier *et al*., 2000). Cromar *et al*. (1996), however, claim that nitrifying bacteria are susceptible to high irradiance and elevated temperatures, both typical of a HRAP, and these ponds therefore exhibit incomplete nitrification.

As ponds are oxygen rich, denitrification is often inhibited, resulting in a possible increase in nitrates. Another possible source for nitrate increase is nitrogen fixation which has been known to occur in certain cyanobacteria and by the nitrogen fixing bacteria, *Azotobacter*, symbiotic with some green algae (Gallon & Chaplin, 1988). In these cases, mechanisms have been developed by the microbes to protect the nitrogenase enzyme from oxygen damage (Maier *et al*., 2000).

There are two principle explanations for the removal of phosphate in the HRAP. The first is incorporation into the algal biomass (Mara & Pearson, 1986; Mesplé *et al*., 1996; Craggs *et al*., 1997). Craggs *et al*. (1997) found evidence for this in effluent from algae left in settling ponds show an increase in orthophosphate, possibly due to the release from the dead portion of the algal biomass.

The other likely removal mechanism is precipitation at high pH levels in calcium rich waters (Moutin *et al*., 1992; Mesplé *et al*., 1995). This is known to occur at pH greater than 8.5 (Moutin *et al*., 1992). According to Hartley *et al*. (1997) the effect of algae appears to be in raising the pH and initiating the precipitation reaction. This results in the co-precipitation of phosphate on calcium carbonate (calcite) (House, 1990) according to the equation:

$$Ca^{2+} + 2(HCO_3^-) = CaCO_3 \text{ (solid)} + H_2O + CO_2 \quad (3)$$

In abiotic experiments, the release of CO_2 resulted in a drop in pH, however, with algae present, this CO_2 is constantly utilised in photosynthesis thus maintaining the high pH (Hartley *et al*., 1997).

An alternative mechanism for phosphate precipitation in HRAP has been investigated by Moutin *et al*. (1992). They found that calcium and phosphate removal occurred in the form of calcium hydroxyapatite precipitation:

$$5Ca^{2+} + 3PO4^{3-} + OH^- = Ca_5(PO_4)_3OH \text{ (solid)} \quad (4)$$

There was evidence of a pH related, and therefore a precipitation removal mechanism active in the demonstration HRAP. A significant drop in phosphate in the second HRAP without a corresponding increase in algal biomass was observed (dry-weight, results not shown), suggesting that precipitation rather than algal growth is responsible for the removal.

5.2 MATERIALS AND METHODS

The materials and methods used in the study are as described in Chapter 3.

5.3 RESULTS AND DISCUSSION

5.3.1 Chemical Oxygen Demand Removal

The COD removal performance of the IAPS was monitored from commissioning until the end of 2004. CODs, however, was only analysed from September 2000. Between the end of 1997 and the beginning of 1999, there was a drop in the CODt in the PFP effluent, resulting from a similar drop in the sewage feed, discussed in Chapter 4. Due to this variation, the data has been separated into two periods: January to October 1997 (Figure 5.2) and February 1999 to September 2004 (Figure 5.3).

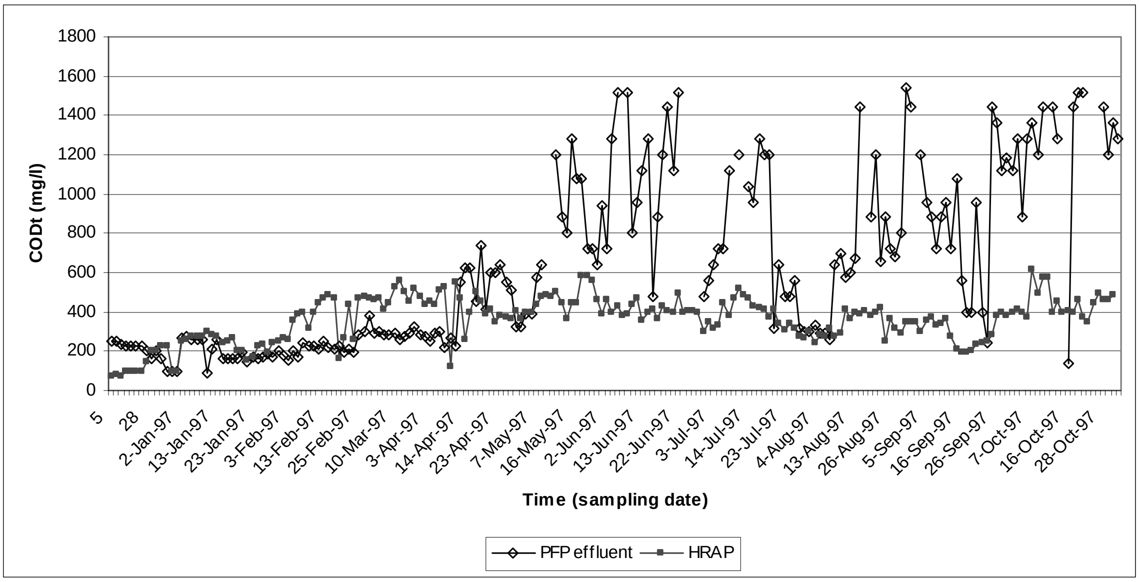


Figure 5.2: Unfiltered chemical oxygen demand removal in the first high rate algal pond, during 1997**.**

From Figure 5.2, it is evident that during commissioning and the first half of 1997, there was, at times, an increase in CODt in the HRAP. Where analysis was carried out on the ASP effluent, it is obvious that this excess organic matter was then removed. Unfortunately there are no ASP results from 1997 and the effect of the settling pond during this period is, therefore, not evident. What is apparent from this data set is the capacity of the HRAP to absorb relatively high COD levels. From May 1997 CODt values of up to 1 500 mg.ℓ^{-1} were recorded. These high levels are, however, not reflected in the HRAP, where the mean removal over the year was 42.8%.

Table 5.1 summarises the COD performance data from the HRAP for the period February 1999 to September 2004. During this period, the CODt in the PFP effluent was more stable than the earlier period, with a mean effluent CODt value of 308 mg.ℓ^{-1}. Consequently, there was actually an increase in the organic content of the HRAP water. This was then removed in the settling pond, indicating an algal contribution to the CODt. Interestingly, the rate of CODt and CODs removal was exactly the same at 42.9%.

Table 5.1: Chemical oxygen demand removal in the high rate algal pond during the period February 1999 to September 2004

	PFP Effluent		HRAP1		ASP1	
	CODt	CODs	CODt	CODs	CODt	CODs
Mean (mg.ℓ$^{-1}$)	308	203	343	128	176	116
Std Deviation	103	74	151	57	83	51
% Removal	N/A	N/A	+11.4 (increase)	37	42.9	42.9

Figure 5.3 shows the real time performance data for the period February 1999 to September 2004. Although a second HRAP was operated in series from August 2003, this data has not been included in the graph as very little additional CODt removal takes place in HRAP2. It is clear from the figure that the CODt in the HRAP follows that of the PFP effluent, with the removal taking place in the settling pond, once again indicating the algal contribution to CODt.

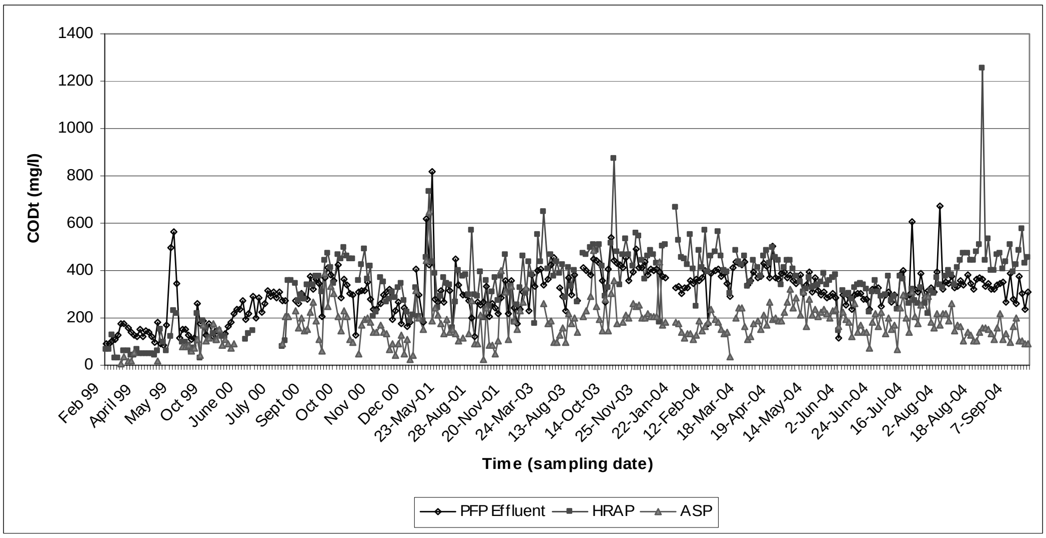


Figure 5.3: Unfiltered chemical oxygen demand removal in high rate algal pond 1 and algal settling pond 1 for the period February 1999 to September 2004.

Analysis of the COD_s, illustrated in Figure 4.4, reflects a different scenario, where there is a 37% reduction in the HRAP, with only a further 6% removal taking place in the ASP (not shown in the figure).

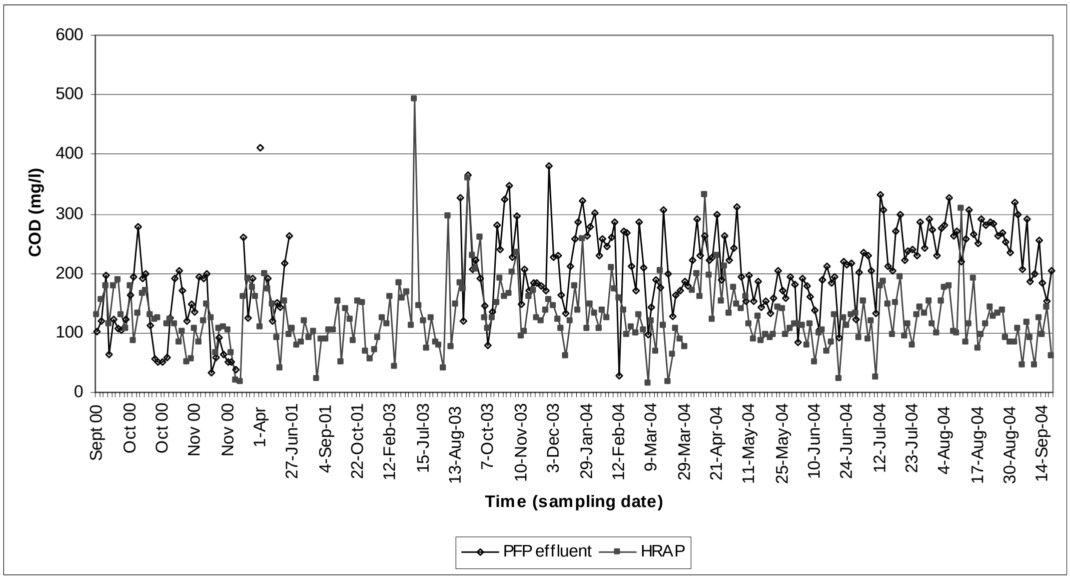


Figure 5.4: Soluble chemical oxygen demand removal in the high rate algal pond for the period September 2000 to September 2004.

Because a number of different researchers were involved in investigating various aspects of the IAPS, there were periods between specific studies when operational control of the system was not as tight as it could have been. Although data was collected for these periods, the system was left to function without any intervention. The system was then the subject of more rigorous management during 2003 and 2004. COD removal performance for this period was compared with the mean results obtained over the nine-year study period. While the CODs removal improved by 9%, the CODt removal rate for this period was 46%, which is only 3% better than that achieved for the entire monitoring period. This observation has both positive and negative implications. Firstly, it indicates the system has limitations in terms of COD removal and, although the 75 $mg.\ell^{-1}$ standard (South African National Water Act No. 36 of 1998) was met for short periods, the mean data over a longer interval show that it is difficult to maintain this level. The more positive conclusion that can be drawn from these data is that the system performs equally well under poor management as under stringent control, making it suitable for areas where technical skill is lacking.

The 43% CODt elimination in the HRAP/ASP unit operation is better than the 31% achieved in a HRAP studied by El Hamouri *et al*. (1995) in Morocco but is not as effective as the St Helena plant in California, where a rate of 53% was reported (Oswald, 1990).

5.3.2 Nutrient Removal

Oswald (1990) claims that significant nutrient removal can be achieved in a HRAP. In the current studies, this was found to be true with regard to ammonium and phosphate but nitrate removal was less consistent. The following figures illustrate the nutrient removal performance of the HRAP.

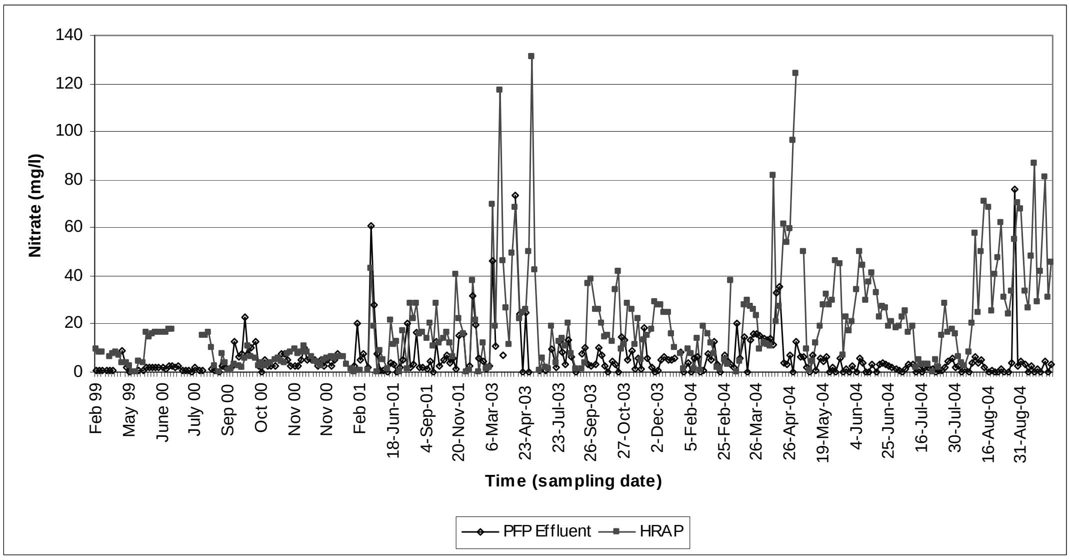


Figure 5.5: Nitrate in the primary facultative pond effluent and high rate algal pond for the period February 1999 to August 2004

During most of the period depicted in Figure 5.5, an apparent increase in nitrates in the HRAP is observed. The mean figures support this observation, indicating a mean increase in nitrate concentration of 47% between the PFP and the HRAP.

These results, based on the IAPS, contradict the findings of a number of other studies, where nitrate reductions in algal ponds have been reported (Tam & Wong, 1989; Cromar *et al*., 1996; Green *et al*., 1996; van der Steen *et al*., 1998). Green *et al*. (1996), however, point out that the large fluctuations in nitrates in the HRAP may be a shortcoming of the system.

From the beginning of 1999 to mid 2002, relatively high concentrations of ammonia were observed in the PFP effluent. This was effectively removed in the HRAP (Figure 5.6). During 2003 and 2004, these ammonia levels dropped, corresponding to periods of high ammonia in the HRAP (Figure 5.7).

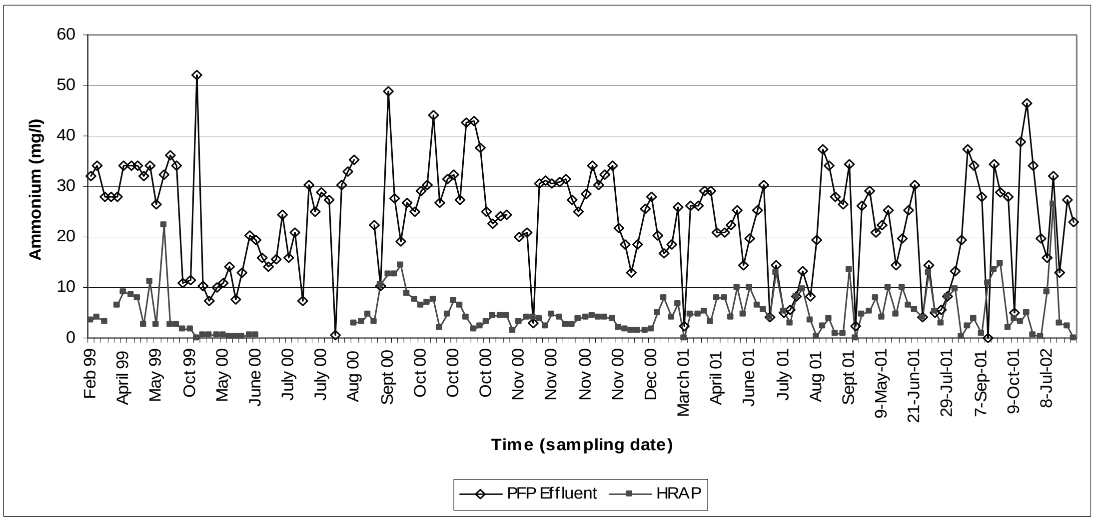


Figure 5.6: Ammonia removal in the high rate algal pond for the period February 1999 to July 2002**.**

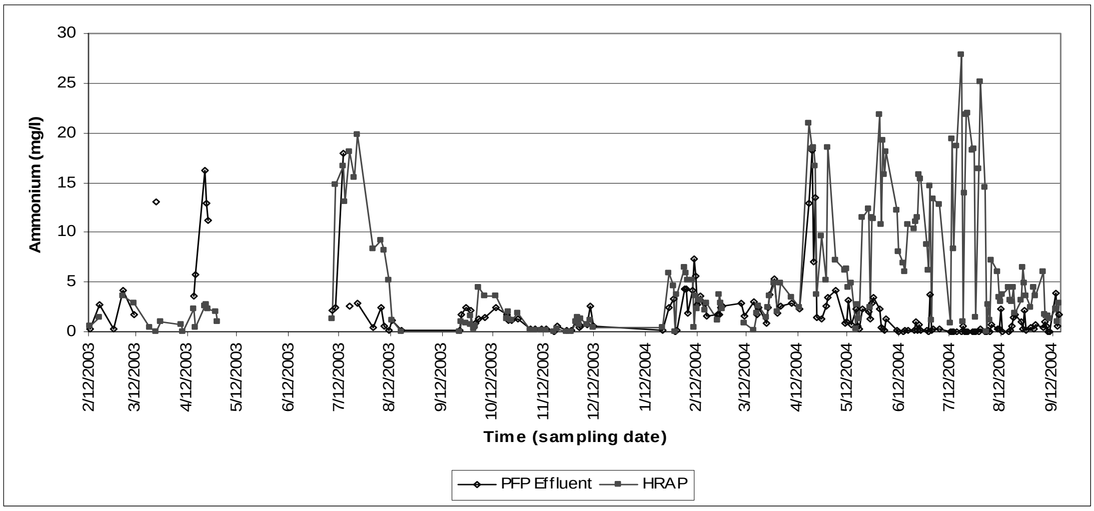


Figure 5.7: Increase in ammonia in high rate algal pond 1 during 2003 and 2004.

The increase in nitrates in the HRAP is difficult to explain conclusively. As there is a 36% decrease in TKN in the HRAP (results not shown), nitrification is a possible explanation.

Organic nitrogen in the PFP effluent is broken down, releasing ammonia, evidenced by the increase shown in Figure 5.7. Some of this ammonia is then oxidised to nitrate by the nitrifiers under the aerobic conditions present in the HRAP. As denitrification is inhibited by oxygen (Maier *et al.*, 2000), the abundance of oxygen in the HRAP would prevent this process from taking place, hence the increase in nitrate. The case for nitrate increase due to nitrification is supported by the data from an experiment undertaken in the HRAP, where the organic load to the HRAP was increased for a ten day period, thereby reducing the amount of oxygen present. During this time the nitrate levels in the effluent decreased considerably (Figure 5.8). During the day enough oxygen was produced to facilitate nitrification but at night this was quickly used up, allowing denitrification to take place with the nitrates thus lost to the atmosphere as N_2 gas as per equation 5.

$$NO_3^- + 5H_2 + 2H^+ \rightarrow N_2 + 6H_2O \qquad (5)$$

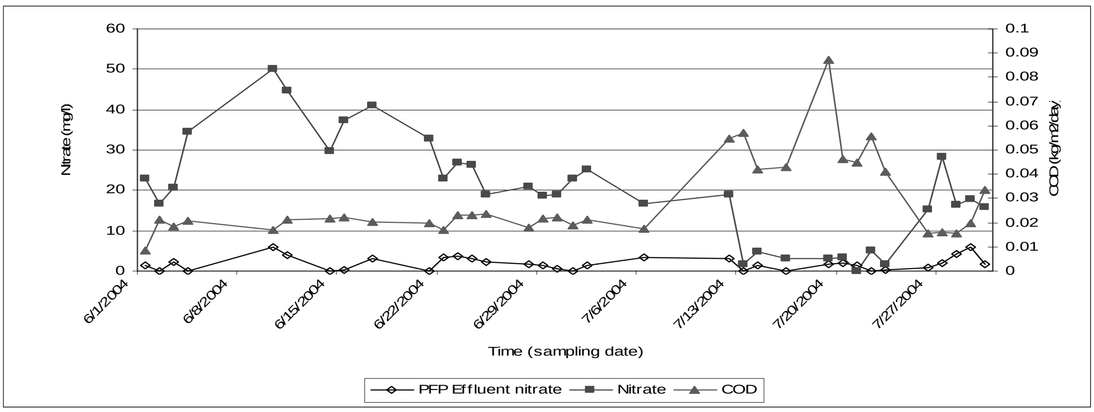


Figure 5.8: Nitrate in the high rate algal pond plotted against chemical oxygen demand load to the high rate algal pond, and illustrating a drop in nitrate levels at high organic loading.

When the COD load was again decreased, there was a corresponding increase in nitrates. This possible nitrogen pathway is defended by Zimmo *et al.* (2003), who suggest that nitrification/denitrification is important in overall nitrogen removal in algal ponds. If nitrogen fixation is taking place, the resulting ammonia would still have to be oxidised to the nitrate present in the water. Further research is, however, required to fully understand nitrogen cycling in the HRAP.

Figure 5.9 illustrates the fate of phosphate in the HRAP. Phosphate removal in the HRAP was not particularly effective, with the variation in effluent levels following those of the influent closely. The mean phosphate reduction in the HRAP was 26%. Dekker (2002) reported a 30% removal during his monitoring of the system in 1997. The work by Rose *et al.* (2002c), during the commissioning phase, also found similar phosphate levels in the PFP and HRAP effluents. Comparable removal efficiencies of 38% and 34% were found by El Hamouri *et al.* (1995) and Cromar *et al.* (1996) in their respective studies.

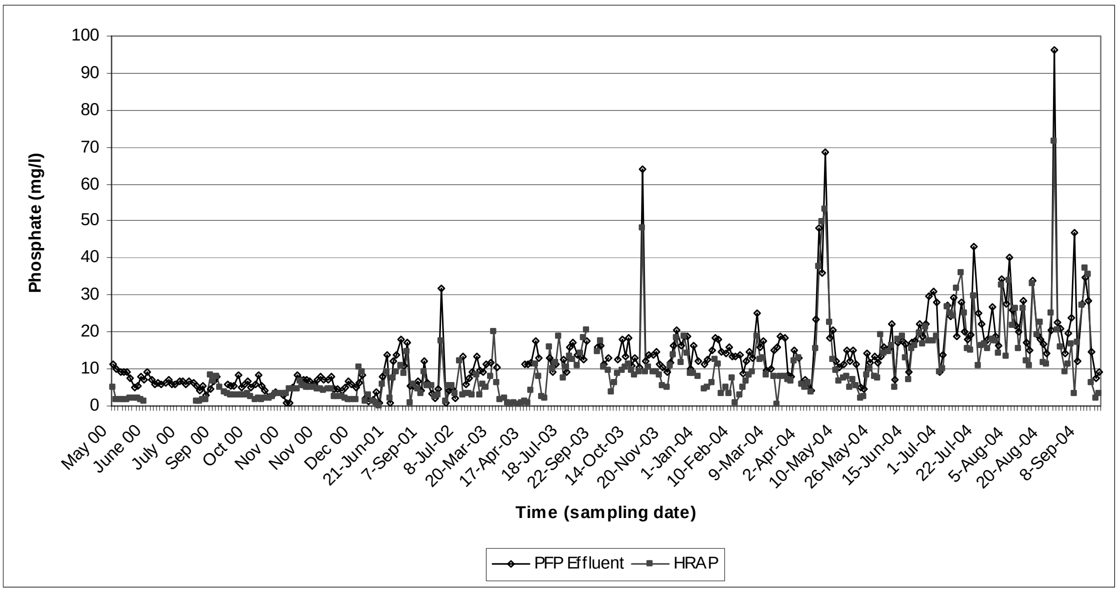


Figure 5.9: Phosphate in the primary facultative pond effluent and high rate algal pond during the period May 2000 to September 2004.

5.3.3 Algae in Integrated Algal Ponding System

The term algae, as commonly used, does not refer to a formal taxonomic grouping but rather to an array of generally aquatic, photosynthetic organisms. This diversity ranges from microscopic prokaryotic cells to large seaweeds such as the giant kelp *Macrocystis*, which can reach lengths of 70 m (Chapman & Gellenbeck, 1989). Algal forms important in wastewater ponds are mostly from the classes Chlorophyta and Euglenophyta, which include non-motile and flagellate green algae (Gloyna & Tischler, 1979; Mara & Pearson, 1986). Other, less-dominant, phytoplankton includes genera from the Cyanophyta (blue-green algae/cyanobacteria) and Chrysophyta (Mara & Pearson, 1986). Although the Cyanophyta are prokaryotic, they generally fill the same ecological niche and are thus grouped with the microalgae (Chapman & Gellenbeck, 1989). This also applies to their use in algal biotechnology and wastewater treatment.

According to Mara and Pearson (1986) species diversity in conventional WSP generally increases as the organic load decreases and consequently fewer species are found in facultative ponds than maturation ponds. The continuously mixed conditions provided by the HRAP, however, favour the culture of specific algae types i.e. relatively larger, non-motile chlorophytes that are able to form commensal or symbiotic flocs (Oswald, 1988c). Due to this design characteristic, a lower species diversity was present in the decreased organic loading of the HRAP than in the more heavily loaded PFP under investigation in this study. Because different algal types flourish in conditions of varying water quality, knowledge of the algal genera present and their biomass concentration provides a useful indication of pond status and wastewater treatment efficiency (Mara & Pearson, 1986).

As has been observed by other authors (Gloyna & Tischler, 1979; Mara & Pearson, 1986; Oswald, 1988c), genera occurring in the PFP were either flagellates such as *Euglena* and *Chlamydomonas* or very small and, therefore, buoyant species such as *Chlorella* sp. It has been suggested that their motility and buoyancy give these types the ability to keep to the surface and thus secure more light exposure, giving them a competitive advantage over the non-motile forms (Mara & Pearson, 1986). *Oscillatoria* sp. was also found periodically in the PFP but did not form thick, malodorous mats as reported by Oswald (1988c).

A number of studies have established that the conditions in the HRAP favour the dominance of *Scenedesmus* sp. and *Micractinium* sp. (Oswald, 1988c; Green *et al*., 1995a; Nurdogan & Oswald, 1995; Zulkifi *et al*., 1996; Craggs *et al*., 2003). Both these genera predominated in the earlier phases of monitoring (Dekker, 2002), however, as operation continued, *Micractinium* sp. together with *Actinastrum* sp. were prevalent during winter, while in summer, *Pediastrum* sp. formed the principle species in the climax culture, especially in the second HRAP. *Pediastrum* sp. was also recorded by Potts (1998) during the commissioning phases in 1996. *Dictyosphaerium* sp. was also common in both HRAPs. All the observed algae formed flocs that were readily settled once introduced into the quiescent settling pond.

Grazing rotifers and ciliates were present throughout the IAPS but appeared to decrease in the second HRAP, possibly due to the lack of suitable food i.e. smaller algae, not possessing protective spines or setae. Grazing is important in the physical functioning of the system as it removes the small, non colonial algae, which are difficult to separate from the water, thereby facilitating the clarity of the final effluent (Benemann *et al*., 1980).

5.4 CONCLUSIONS

- CODt increased in the HRAP due to the growth of algal biomass, indicated by the similarity in soluble HRAP COD and settled effluent COD, from the ASP.
- Algal biomass was effectively removed in the ASP, giving a net CODt reduction during the HRAP/ASP stage of 43%.
- Although residual COD was mainly in the stabilised form of algal material, the full IAPS system (HRAP operated as a single stage unit operation) did not achieve the 75 mg l^{-1} discharge standard.

- While an average 26% phosphate reduction was observed in the HRAP, this was not sufficient to bring effluent levels to within the 10 $mg.\ell^{-1}$ required by the discharge standards.
- Despite good ammonia removal, residual levels also at times exceeded the 3 $mg.\ell^{-1}$ ammonia discharge standard.
- Nitrate removals were somewhat erratic, with the levels of these nutrients increasing at times. However, the manipulation of organic load to the HRAP showed promise for denitrification and this should be pursued more thoroughly in future.
- Algal genera found in the PFP were similar to those recorded in conventional WSP. The operation of the HRAP results in a lower species diversity, dominated by strongly floc-forming green algal forms which provides a good settling characteristic.

6. THE I-HRAP AS A TERTIARY TREATMENT UNIT OPERATION 1: DISINFECTION

6.1 DEVELOPMENT OF THE IHRAP CONCEPT

As was noted in the previous chapter, effective removal of COD and nutrients was achieved through the IAPS (HRAP operated as a single stage unit operation), and the final discharge values were comparable to those anticipated from optimally performing conventional WSP systems. It was, however, also evident from the study up to that point that the South African discharge standard requirements (Department of Water Affairs and Forestry, 2002), particularly for phosphate was not achieved. Of greater concern was the observation that *E.coli* removal performance, also in common with most WSP, seldom met the relaxed standard of 1000 cfu.100 $m\ell^{-1}$ (data reported below). Although many WSP in South Africa perform substantially worse than the IAPS results reported here, these, nevertheless, raised concern about the wide application potential of the IAPS which had been anticipated with the inception of the project.

The problems noted above were essentially those of tertiary treatment, and it was evident that the system, nevertheless, performed well in the primary and secondary treatment functions. It was thus decided to focus the attention of the project on adaptations to the system that might enhance tertiary treatment operations. Two strategies were the subject of preliminary investigation (results not shown). Firstly the retention time in the HRAP was extended and, while phosphate removal showed an improvement, disinfection remained a problem. It was apparent that the averaging of secondary and tertiary treatment functions in the HRAP, while yielding a performance superior to most WSP operations, did not enable the separate optimisation of the different treatment objectives.

In the second study the objective was to see whether the HRAP could be optimised as a separate tertiary treatment unit operation. In the original plant design two HRAP units had been provided (Figure 3.1), and these had been operated in parallel up to this point to provide the appropriate capacity for full flow conditions. The two HRAP were then reconfigured to operate in series (Figure 3.7c), without changing the design loading. This was done by wasting the flow to the second unit, and passing the settled flow from HRAP1 to HRAP 2. The system was operated in this format for a short period and it immediately became apparent that a very different pond algal population structure established in the HRAP2. While the algal growth in this system was also pronounced, it was not dispersed and aggregated into a tight heavy floc with substantially improved settling characteristics. The interstitial liquor, however, remained quite clear with few unattached planktonic algal cells present and a low colloidal content. With this form of algal floc, light penetration into the pond was much improved which suggested possibilities for enhanced disinfection in the HRAP series configuration.

Based on these preliminary observations a fully structured study was set up to investigate the use of the HRAP as a tertiary treatment operation which became known as the IHRAP.

Research undertaken by Clark (2001), Dekker (2002), Neba (2003) and Wells (2005) showed the effectiveness of the IHRAP design approach in both N and P removal and disinfection. Final treated effluent from both the IAPS and from the Grahamstown conventional sewage works were used in these studies. These results indicated the potential widespread use of the IHRAP system as a final polishing operation that might be appended to many existing small plants, especially in rural areas, where failure to meet nutrient discharge standards is not uncommon. Figure 3.7 illustrates the various HRAP configuration investigated in these studies.

As the IAPS process is intended for use in smaller and rural communities, there is the high probability that final water treatment effluent will, either intentionally or unintentionally, be used for irrigation, livestock watering, washing or recreation. It is therefore imperative that this water is rendered pathogen free by any installed treatment system (van Leeuwen, 1996). As these areas also generally have a shortage of skills, any disinfection mechanism used needs to be relatively maintenance free, with a low level of operator skill inputs, and a high level of reliability required for it to function. Monitoring and possibly enhancing the removal of faecal indicator organisms in the IAPS, and specifically the IHRAP system, was thus considered a critical area requiring focus during the project studies.

While the nutrient removal results from these studies have been reported in Chapters 6 and 7 of the report, the investigation of disinfection effects achieved during the study of IHRAP operations are presented in this chapter.

6.2 DISINFECTION IN PONDING SYSTEMS

Many different mechanisms have been reported to play a role in disinfection achieved in ponding systems including predation, sunlight, temperature, dissolved oxygen, pH, sedimentation and starvation (Fallowfield *et al*., 1996, Davies-Colley, 2003). There has, however, also been much debate over which of these factors is the most important. Parhad and Rao (1962) had reported that *E. coli* could not grow in wastewater with a pH higher than 9.2 and this was considered for some time to be the major factor involved in effective pond disinfection. Pearson *et al*. (1987) had also noted that faecal coliforms were adversely affected by pH and Dissolved Oxygen but not necessarily by high light intensities.

As the COD of the ponds is relatively low towards the end of treatment, it is possible that a certain amount of starvation does take place. Sedimentation has also often been given as one of the reasons for pathogen reduction in conventional WSP (Almasi & Pescod, 1996; Maynard *et al.*, 1999).

Although an up to 2 log reduction can be observed in facultative ponds, many of the factors normally responsible for the elimination of indicator faecal coliforms are not present in this system. These include a pH greater than pH 9 (Parhad & Rao, 1962; Sebastian & Nair, 1984; Pearson *et al*., 1987), good sunlight penetration (Craggs *et al*., 2004) and high levels of dissolved oxygen (Davies-Colley *et al*., 1999). The *E. coli* reduction in these systems could therefore be attributed to other factors such as sedimentation and starvation, as the organic content of the water is removed (Almasi & Pescod, 1996; Maynard *et al*., 1999).

The action in disinfection of a combination of factors has been supported by Davies-Colley *et al*. (1999), who found that photo-oxidation caused damage to *E. coli* cell membranes but that this was not sufficient in itself to cause inactivation. Elevated pH is, however, rapidly lethal to *E. coli* cells with damaged cell membranes (Davies-Colley e*t al*., 1999). An increasing body of research has, however, indicated that light might be the single most important factor causing disinfection in ponds (Maynard *et al.*, 1999; Davies-Colley 2003). The study by Craggs *et al*. (2004), claimed that sunlight was the single most important factor, and that pH and DO were only second order factors in disinfection.

The above findings seemed to suggest that the substantially improved light penetration observed in the IHRAP configuration might enable enhancement of the disinfection process occurring in this system. The investigation of this hypothesis formed the basis of the studies reported below (Wells, 2005).

6.3 MATERIALS AND METHODS

Materials and methods are as described in Chapter 2

6.4 RESULTS AND DISCUSSION

6.4.1 Pathogen Removal in the Primary Facultative Pond

From Figure 6.1 it can be seen that there is a constant, high level knockdown of the pathogen indicator *E. coli* in the PFP. The mean removal rate over the two-month monitoring period was 90%. The total coliform analysis gave similar results (not shown), with a 1 log reduction observed. The *E. coli* levels in the PFP effluent were, however, still far higher than those required for safe discharge into a natural watercourse. These levels are consistent with a number of other pathogen removal investigations reported for primary ponds (Pearson *et al*., 1995; Jagals & Lues, 1996; Rangeby *et al*., 1996; Almasi & Pescod, 1996)

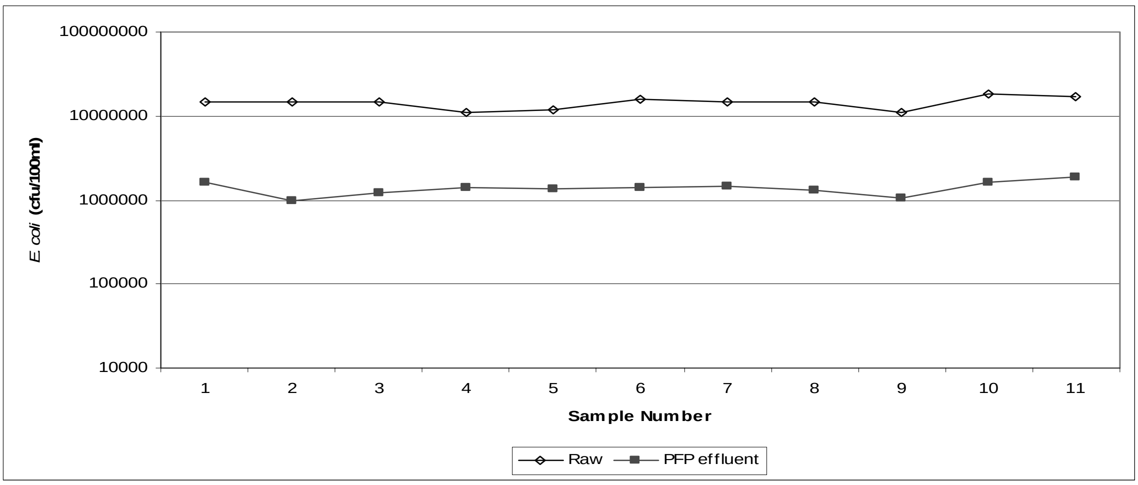


Figure 6.1: *E. coli* removal in the primary facultative pond monitored over a period of 2 months.

Because the PFP effluent does not meet irrigation or discharge standards, with regard to indicator organisms, further treatment is required. In terms of the Grahamstown IAPS, this function is provided by the HRAP unit operations.

6.4.2 Disinfection in the High Rate Algal Pond

Figure 6.2 illustrates the *E. coli* removal performance of the high rate ponds over the period of 12 months operation.

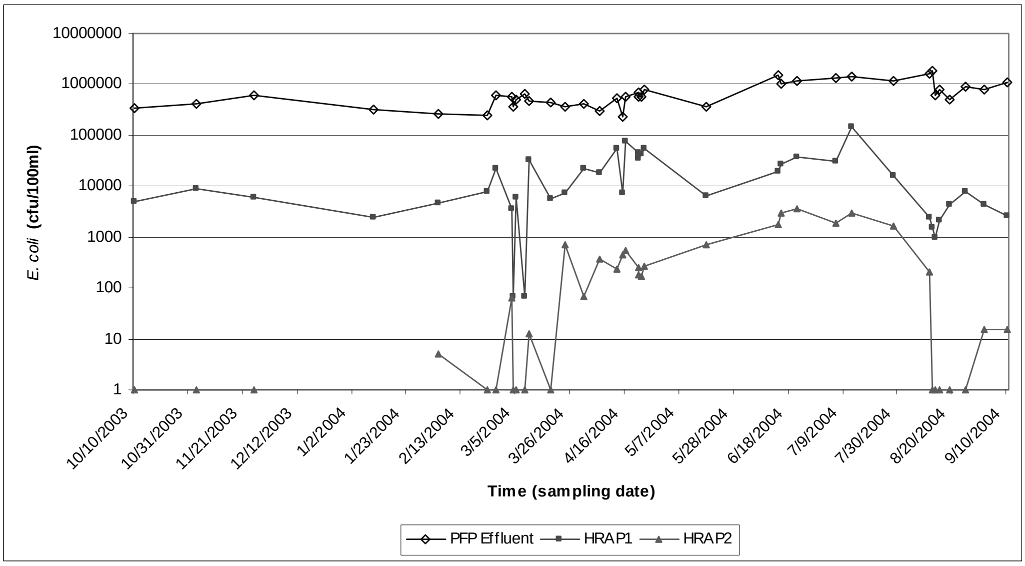


Figure 6.2: *E. coli* removal in high rate algal pond 1 and high rate algal pond 2, during the 12-month period October 2003 to September 2004.

From Figure 6.2, it can be seen that there is a 2 log knockdown of *E. coli* in the first HRAP. The effluent from this pond, however, still contains *E. coli* cfu counts in the region of 10^4 colony forming units (cfu)100 mℓ^{-1}. It is in the second HRAP (IHRAP) that complete removal of the remaining *E. coli* and faecal coliforms was observed and a cfu count was sustained over the 6-month summer period (3-day HRT). As the season moved into winter, it is evident in Figure 6.2 that the indicator count in HRAP2 increased, although, apart from June and July (mid-winter) where the standard was slightly exceeded, the faecal coliform remained below 1000 cfu.100 mℓ^{-1} - the South African guideline for irrigation (DWAF, 2001).

Studies were then undertaken to evaluate variation of the HRT in order to control the level of disinfection achieved in HRAP2. Results shown in Figure 6.3 indicated that it was possible to achieve a count of cfu.100 mℓ^{-1} for both *E. coli* and faecal coliform, even under low-temperature conditions.

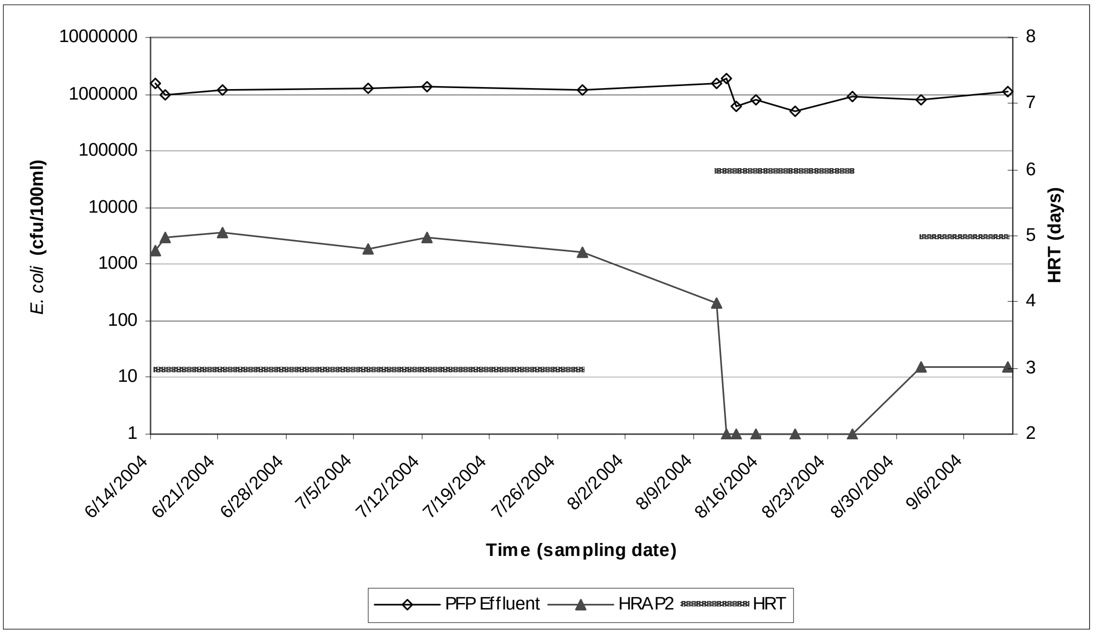


Figure 6.3: *E. coli* removal in the IHRAP under winter conditions and with the operation of the system at different hydraulic retention times.

It was shown that with a HRT of 6 days, under winter conditions, a 100% removal of *E.coli* cfu was achieved. To determine the cut-off point of the observed effect, the retention time was decreased again to 5 days. Under these conditions the *E.coli* cfu count rose to 15 cfu.100 mℓ^{-1}. This setting still achieved a 99.999% removal but, in order to attain an indicator free effluent, a HRT between 5-6 days was found to be necessary. During the 6 day HRT regime, total coliform counts were also reduced to cfu.100 mℓ^{-1} (data not shown).

The seasonal adjustments to HRT are consistent with those found by El Hamouri *et al*. (1994) in Morocco, where a winter 6 days HRT was required to obtain the same results as a 3 day HRT in summer. In this study, however, they were unable to bring the faecal coliforms in the final effluent to under 10^3 cfu.100 mℓ^{-1}. Bahlaoui *et al*. (1997) in France and Garcia and Bécares (1997) in Spain, were also unable to achieve a removal efficiency of greater than 99.1% and 98.05% respectively in their study of HRAP. Other studies have shown comparable disinfection rates, with faecal coliform or *E. coli* levels in HRAP effluents, less than 100 cfu.100 mℓ^{-1} (Sebastian & Nair, 1984; Davies-Colley *et al*., 2003).

From the observations made in this study, it is also clear that a compelling inverse correlation (r = -0.75) exists between pH and *E. coli* inactivation and this appears to be one of the key agents in disinfection (Figure 6.4). As the pH drops below pH 9.5 in the HRAP, a sharp rise in *E. coli* count is observed. When this pH is again increased to above pH 10, 100% removal was once again attained.

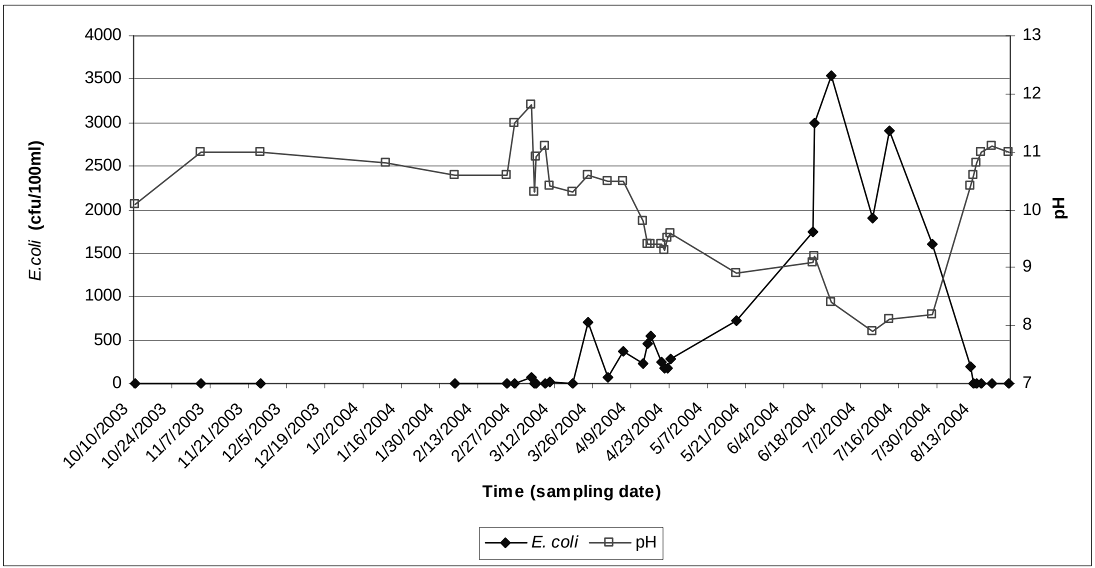


Figure 6.4: Change in *E. coli* counts in the high rate algal pond as a correlate of decrease and increase in pH.

It should be noted that the HRAP is not a controlled environment and it is, therefore, difficult to attribute the pathogen fluctuation to pH alone. pH, temperature, DO and solar radiation all vary diurnally almost in phase with each other and must thus be considered as closely inter-related factors operating in the system (Craggs *et al.*, 2004). For instance, an increase in irradiation might be directly responsible for a pathogen decline or it may simply encourage algal metabolism with a subsequent increase in pH and DO which are ultimately the cause of *E. coli* inactivation.

Although only *E.coli* and total coliform counts were monitored in this study, previous work has shown that *E.coli* removal provides a good indicator for the disinfection of most bacteria and viruses including *Salmonella*, *Shigella*, *Campylobacter* and rotavirus (Davies-Colley, 2003). The only notable exception is *Vibrio cholerae* which has been shown to have a different disinfection response in WSP and caution should be applied when using *E.coli* removal in WSP to infer its removal (Davies-Colley, 2003). Whether the IHRAP would achieve a better response in this respect than the WSP would be speculative at this point but, given the results acquired for the system to date, it would seem worthwhile further investigating this potential. Given the poor correlation of laboratory experiments, and the ethical problems associated with the spiking of an open system with this pathogen, a satisfactory methodology for such an investigation would seem to require location of an IHRAP disinfection unit in an area in which cholera is endemic. This has been identified as a subject of future work in the group.

6.5 CONCLUSIONS

- The observation of complete removal of both *E.coli* and coliforms at HRT of 3 days in summer, and 6 days in winter conditions, is a result outside the normal capability of WSP operations.
- The operation of the IHRAP as a tertiary treatment unit has been demonstrated and, although in this study final treated water from the IAPS was examined, subsequent work on nutrient removal in the final water stream from a conventional sewage treatment plant (reported in Chapters 7 & 8), indicates that a similar performance could be anticipated.
- These results hold important implications for water recycle and reuse strategies especially in rural areas. This is discussed further in Chapter 10.

7. THE I-HRAP AS A TERTIARY TREATMENT UNIT OPERATION 2: PHOSPHATE REMOVAL

7.1 INTRODUCTION

Eutrophication of freshwaters, particularly due to excessive phosphorus concentrations is a problem experienced worldwide, and the reduction of inorganic phosphate concentration in wastewater before final discharge is therefore an inevitable preoccupation for modern society (Arnz *et al*., 2001).

Biological phosphate removal from wastewater is generally achieved in two ways: stoichiometric coupling to microbial growth or enhanced storage in the biomass as polyphosphate (poly-P). The latter was known as “luxury uptake” (Levin and Shapiro, 1965) and is the key mechanism in the enhanced biological phosphate removal (EBPR) process. The EBPR process is primarily characterised by circulation of activated sludge through anaerobic and aerobic phases, coupled with the introduction of influent wastewater into the anaerobic phase (Barnard, 1975). By this anaerobic-aerobic configuration, micro-organisms which accumulate poly-P, and thus have a high phosphorus content, are selected and grow to dominance in the process (Van Loosdrecht *et al*., 1997). Since pure cultures that possess complete characteristics of a Phosphate Accumulating Organism (PAO) have not been isolated yet, the biochemical mechanism cannot be definitely described (Mino *et al*., 1998). The EBPR process is well established in practice and many full-scale plants are in operation world-wide (Arnz *et al*., 2001).

Removal of phosphorus from wastewater by chemical addition has also found widespread application and the most commonly used chemicals are lime, alum and ferric chloride. Here the dissolved orthophosphates are precipitated and removed by solids separation processes such as sedimentation, flotation or filtration (Maurer and Boller, 1999). The quantity of chemical required is determined by the concentration of phosphorus species in the wastewater and the degree of purification desired. A number of other factors such as pH, alkalinity, ratio of metal salt to phosphorus, intensity of mixing and the presence of interfering substances will also affect the actual quantity of chemical required (Thomas *et al*., 1996). The transfer of dissolved phosphorus species (present as H_3PO_4, $H_2PO_4^-$, HPO_4^{2-}, PO_4^{3-} depending on pH) into particulate form includes three mechanisms: (1) chemical precipitation of metal-hydroxy-complexes of low solubility; (2) selective adsorption of dissolved phosphorus species onto freshly precipitated metal-hydroxy-complex surfaces; and (3) flocculation and co-precipitation of finely dispersed colloidal matter. The latter mechanism is independent of the phosphorus speciation in the water but depends mainly on size and surface chemical properties of the phosphorus containing colloids. These mechanisms are not independent of each other but take place simultaneously when precipitation chemicals are added to the wastewater.

Biologically mediated chemical precipitation of phosphorus in EBPR plants has previously been reported (Fuhs and Chen, 1975; Arvin, 1983; Mino *et al*., 1985; Appeldoorn *et al*., 1992). The relative amount of phosphorus precipitated was found to be as high as 80% and the influence of high pH and high concentrations of calcium was pointed out. The chemical precipitation of phosphorus in conventional biological nutrient removal plants may be mediated in at least two ways: First the elevated P-concentrations created by anaerobic phosphate release from PAO can initiate and accelerate calcium phosphate precipitation. Secondly, biological denitrification in fixed biofilms and possibly also in bacterial flocs can lead to phosphate precipitation due to the elevated pH-conditions inside the biofilms (Arvin and Kristensen, 1979). In both cases the precipitation conditions must be generally favourable, i.e. the calcium concentration should be reasonably high, roughly above 50 mg.ℓ^{-1}, and the concentration of precipitation inhibitors low; magnesium, pyrophosphates and bicarbonate (alkalinity). It is also essential that the pH is relatively high, preferably above 7.5. Struvite (magnesium ammonium phosphate hexahydrate), is a mineral that often precipitates from wastewater during anaerobic biological treatment of hog wastes (Maqueda *et al*., 1994), poultry wastes (Manninen *et al*., 1989), wine distillery effluents (Loewenthal *et al*., 1998), and biosolids from biological phosphorus removal processes (Fujimoto *et al*., 1991).

The concentration of orthophosphate may depend on the content of calcium in calcium rich waters (Hepher, 1958; Golterman and Meyer, 1985) including lime-treated wastewaters (Banister *et al*., 1998). It therefore appears that the equilibrium between the orthophosphate and the solid phase determines the concentration of phosphate in solution. Although the relationship between phosphate and calcium is unquestionable, different authors disagree on the composition of the solid phase (Arvin and Kristensen, 1979; Golterman and Meyer, 1985; House, 1990). It seems that apatite, $Ca_5(PO_4)_3OH$, amorphous tricalcium phosphate, $Ca_3(PO_4)_2$, brushite, $CaHPO_4{\cdot}2H_2O$, as well as octacalcium phosphate, $Ca_4H(PO_4)_3$, may determine the equilibrium phosphate concentration.

Hartley *et al*. (1997) reported the co-precipitation of phosphate with calcite in the presence of photosynthesising green algae where the algae played a central role in raising the pH and initiating the precipitation reaction. In a solution containing calcium bicarbonate the rising pH, due to algal uptake of dissolved CO_2, resulted in the precipitation of calcite and during this process orthophosphate is incorporated in the calcite crystals. The precipitation of calcium phosphate minerals in a high rate algal pond was reported by Moutin *et al*. (1992), although phosphorus removal was only 40%, with more than 4 mg.ℓ^{-1} dissolved phosphorus remaining in the treated wastewater. In the literature, two possible reaction mechanisms for calcium phosphate precipitation are proposed. Firstly, the co-precipitation of inorganic phosphorus with calcium carbonate (calcite) is a phenomenon that occurs naturally in hardwater lakes. According to Hartley *at al.* (1997) calcite starts precipitating from a solution containing calcium bicarbonate at pH >8.8, which cause the production of carbon dioxide:

$$Ca^{2+} + 2(HCO_3^-) = CaCO_3\,(solid) + H_2O + CO_2 \qquad (6)$$

In abiotic experiments a decrease in pH was observed during the precipitation reaction which was due to CO_2 release. However, in the presence of algae the process of photosynthesis constantly utilised the CO_2 produced during calcite precipitation. It is believed that phosphate first adsorbs to the calcite crystal surface and then a fraction of this becomes incorporated into the crystal at active kink sites during crystal growth (House and Donaldson, 1986; House, 1990).

Secondly, the removal of calcium and phosphate from slightly alkaline wastewater can be approximately represented as the precipitation of calcium hydroxyapatite (HAP):

$$5Ca^{2+} + 3PO_4^{3-} + OH^- = Ca_5(PO_4)_3OH \text{ (solid)} \qquad (7)$$

Although HAP is the thermodynamically stable state, the phosphate concentration is determined by the solubility of the amorphous tricalcium phosphate [$Ca_3(PO_4)_2$], which was confirmed by the calculation of the theoretical predicted solubility as well as various experiments by Moutin *et al.* (1992).

The development of the algal turf scrubber for phosphorus removal from secondary treated wastewater resulted in effective removal of phosphate through combined biological and chemical precipitation induced by a microalgal colony growing on an inclined flow way (Craggs *et al.*, 1996).

Against the above literature background it was hypothesised that the IHRAP engineered as a free-standing tertiary unit operation could be applied for the removal of phosphates from conventionally treated sewage wastewaters (Dekker, 2002).

7.2 MATERIALS AND METHODS

7.2.1 Pilot Plant

The IHRAP2 used for this study could receive final effluent from either the conventional GDW-final or from the AIWPS-final as described in Figure 3.7. The HRT in HRAP2 was held at 4 days with an influent loading rate of 37.5 $m^3.d^{-1}$.

7.2.2 Laboratory Experiments

Laboratory experiments were conducted to determine the likely method of Phosphate removal occurring in the HRAP. The culture apparatus used for the algal-mediated experiments consisted of a 1-litre cylindrical glass vessel with two ports (Figure 7.1). Each batch precipitation experiment was repeated at least three times, temperature 25ºC, and chlorophyll-a concentrations in the glass vessel were similar to that of the I-HRAP medium.

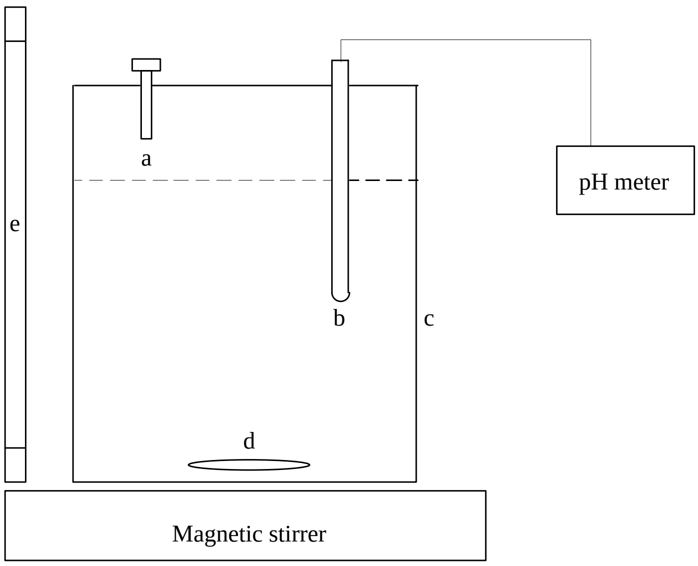


Figure 7.1: Culture apparatus used for algal mediated experiments with the following components: (a) small sampling port; (b) pH electrode; (c) 1-litre cylindrical glass vessel; (d) magnetic follower; (e) fluorescent tube x 6.

A pH electrode connected to a Cyberscan 2500 pH meter was inserted through one of the ports and the remaining port was used for sampling or the addition of algal culture. The culture apparatus was surrounded by 6 fluorescent tubes, giving a total photosynthetical available radiation (PAR) of 158 µmol.m^{-2}.s^{-1}, as energy source for photosynthesis. A magnetic stirrer and follower stirred the solution. Precipitation of calcium phosphate was investigated using fresh algal biomass from the I-HRAP and washing the algae with distilled water. Each batch precipitation experiment was repeated at least three times. Temperature was held at 25^{o}C, and chlorophyll-a concentrations in the glass vessel used were similar to that of the I-HRAP medium.

7.2.3 Analytical Methods

Analysis of all parameters was undertaken as previously described in Chapter 3.

7.3 RESULTS AND DISCUSSION

7.3.1 Evaluation of I-HRAP for phosphate removal

Observations of the continuous operation of the 500 m^2 I-HRAP receiving GDW-final as influent are shown in Figure 7.2. The removal of soluble phosphate was around 90% resulting in effective levels of less than 1 mg P.ℓ^{-1} being achieved in the algal pond. Table 7.1 summarises the I-HRAP performance, where the loss of phosphate by mineralisation with calcium is in accordance with that of the flask studies.

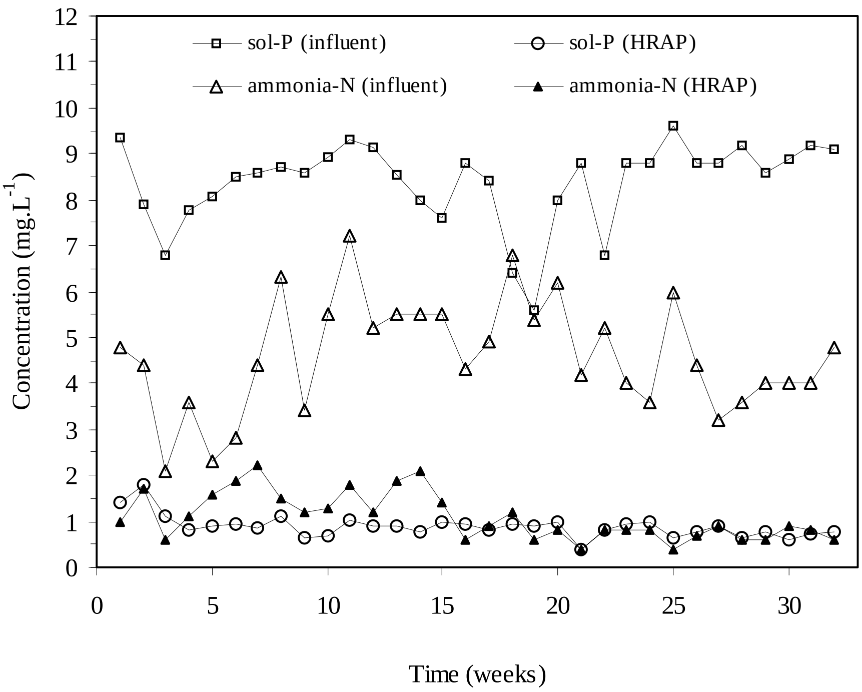


Figure 7.2. Treatment of GDW-final water in the I-HRAP over a 32-week period, indicating the phosphate removal effect.

Table 7.1: Performance of the I-HRAP and its associated ASP during treatment of the GDW-final water. Standard deviations in brackets.

Parameter		Influent		I-HRAP		I-ASP *	
pH		7.2	(0.2)	10.5	(0.6)	10.4	(0.7)
sol-P	$mgP.L^{-1}$	8.39	(0.91)	0.88	(0.25)	0.92	(0.24)
NH_3	$mgN.L^{-1}$	4.6	(1.22)	1.09	(0.51)	1.12	(0.52)
NO_3	$mgN.L^{-1}$	14.7	(1.92)	15	(1.7)	15.1	(1.6)
Calcium	$mg.L^{-1}$	29	(2)	17	(1)	17	(1)
SCOD	$mg.L^{-1}$	60	(16)	54	(14)	55	(15)
TCOD	$mg.L^{-1}$	88	(17)	335	(21)	59	(32)
$Chlorophyll_a$	$\mu g.L^{-1}$	–	–	478	(120)	48	(29)

*ASP overflow results shown for period week 23-32.

During this 32-week study period different procedures for harvesting of settled algal biomass from the ASP were used (described below) and the release of phosphate in the ASP into the final discharge is evident in Figure 7.

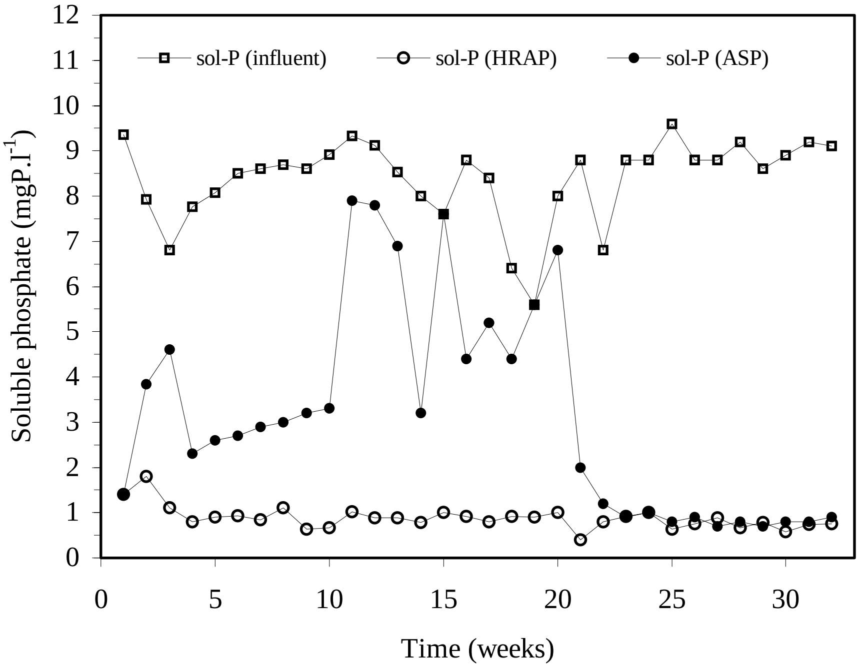


Figure 7.3. Release of phosphate from I-ASP during different biomass harvesting procedures.

Week 1 to 10:
During this period the ASP was cleaned weekly by withdrawing the slurry from the bottom without first decanting the supernatant. This procedure seemed to result in a degree of mixing between the bottom phase (where P-release took place) and the phosphate free ASP overflow.

Week 11 to 20:
From week 11 daily recycling of algal biomass from the ASP was initiated in an attempt to increase the chlorophyll concentration in the I-HRAP for possible enhanced phosphate removal. During this period algal biomass remained in the system with no harvesting taking place. This operating procedure did not have a measurable effect on the phosphate level inside the I-HRAP; however, it resulted in a higher degree of P-contamination of the ASP overflow. The slurry pump probably did not remove all the biomass during each cycle and again mixing between the bottom and surface took place. Flask studies (reported below) showed that under the 4-day hydraulic retention time in the I-HRAP, the algal biomass concentration was not the limiting factor for phosphate removal, but rather the low calcium level. The addition of ± 5 mg Ca.ℓ^{-1} in the form of calcium chloride resulted in further calcium phosphate precipitation to achieve a level of less than 0.5 mg P.ℓ^{-1}.

Week 21 to 32:
Daily biomass recycling stopped and the method for harvesting returned to the standard operating procedure, where the ASP was cleaned properly on a weekly basis while preventing phosphorus contamination of the final discharge. The standard procedure for settled biomass removal was to first decant the supernatant liquid down to the concentrate and then remove the concentrate with a slurry pump.

The performance of the I-HRAP was also evaluated over a 24-hour period during week 24 to examine the possible relationship between photosynthesis and phosphate removal. The results are shown in Figure 7.4, and during this time the influent characteristics were very similar to the average values shown in Table 7.1. Note that for the time of the year, during which this study was undertaken, sunrise was at about 06:00 and sunset at 19:00. In Figure 7.4 (a) the graphs for DO, temperature and pH follow the same profile, where DO and pH appears to be directly linked to algal photosynthetic activity.

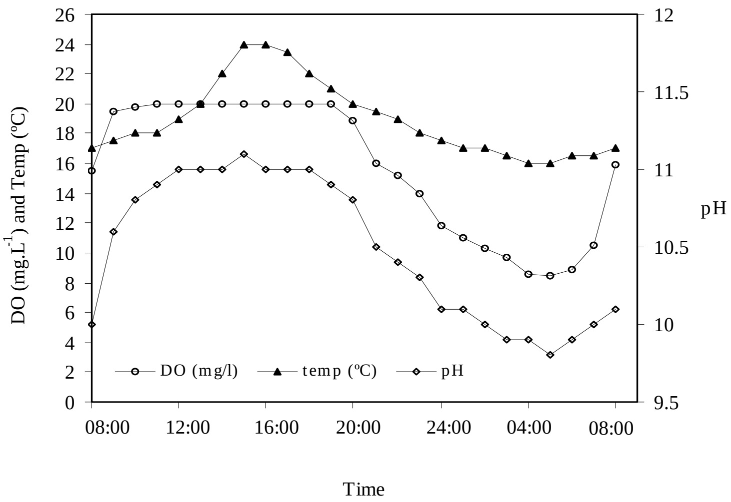


Figure 7.4a. The performance of the I-HRAP monitored over a 24-hour period.

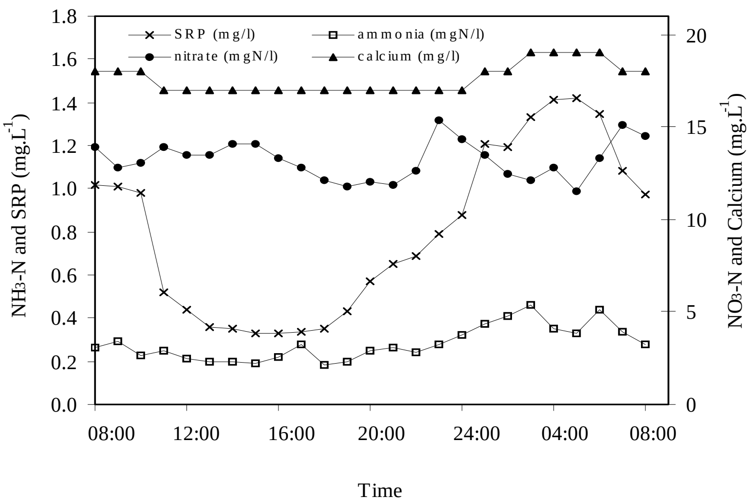


Figure 7.4b. The performance of the I-HRAP monitored over a 24-hour period.

In the idealised photosynthetic equation

$$6CO_2 + 12H_2O \xrightarrow[\text{Algae}]{\text{light}} 6(CH_2O) + 6O_2 + 6H_2O \qquad (8)$$

CH_2O is regarded as the organic matter fixed in algal material and O_2 on the right side of equation 11 has been shown to come entirely from the water on the left hand side (Rabinovitch and Govindjee, 1969). The algae are primarily dependant on dissolved carbon dioxide to fill their photosynthetic carbon requirements. In waste treatment the major source of inorganic carbon for algal growth is the organic carbon in the waste, which must first be released as CO_2 by bacteria decomposing the waste. Another source is the bicarbonate ion, HCO_3^-, from which most algae have a mechanism for extracting CO_2. Another source is CO_2 that can be absorbed directly from the air, and at a pH of 8 the rate of absorption is about 10^{-11} mol.atm^{-1}.cm^{-2}.s^{-1}. But at a pH of 10 the rate is 100 times greater (Oswald, 1994). Given the low COD (consisting mainly of slowly biodegradable organics) in the I-HRAP influent, it is believed that the source of CO_2 is mainly from atmospheric absorption. Algae have an inorganic carbon assimilation mechanism involving the enzyme, carbonic anhydrase. The HCO_3^- is converted to carbon dioxide, which is internalised and incorporated into the photosynthetic pathway, and hydroxide ions remain in the medium. The hydrogen ions generated are removed from the solution, it is believed, by accumulation into the algal cells. The resultant increase in hydroxide ion concentration is therefore responsible for the rise in pH of the I-HRAP medium during the daytime.

During night-time respiration occurs which is basically the reverse of photosynthesis, where dissolved oxygen is absorbed and carbon dioxide released by the algal cells and thereby causing lowering of the pH. From Figure 7.4(b) it is evident that soluble phosphorus removal from the I-HRAP medium is highest when the algal photosynthetic activity is at its maximum during daytime and the opposite for night-time. Again, the correlation between calcium and phosphate levels reflects the laboratory results.

7.3.2 Calcium Phosphate Precipitation Study

The removal of phosphate from GDW-final water in the presence of green algae was evaluated first. The induction period before the onset of precipitation lasted for more than 2 hours, during which time there was a steady rise in pH from the starting pH of 7.8 to 9.4 (Figure 7.5). This rise in pH was due to the microalgal photosynthetic activity, as it did not occur in the absence of algae or when the algae were maintained in darkness. The correlation between phosphate and calcium levels appears evident with no change in the phosphate concentration observed prior to precipitation.

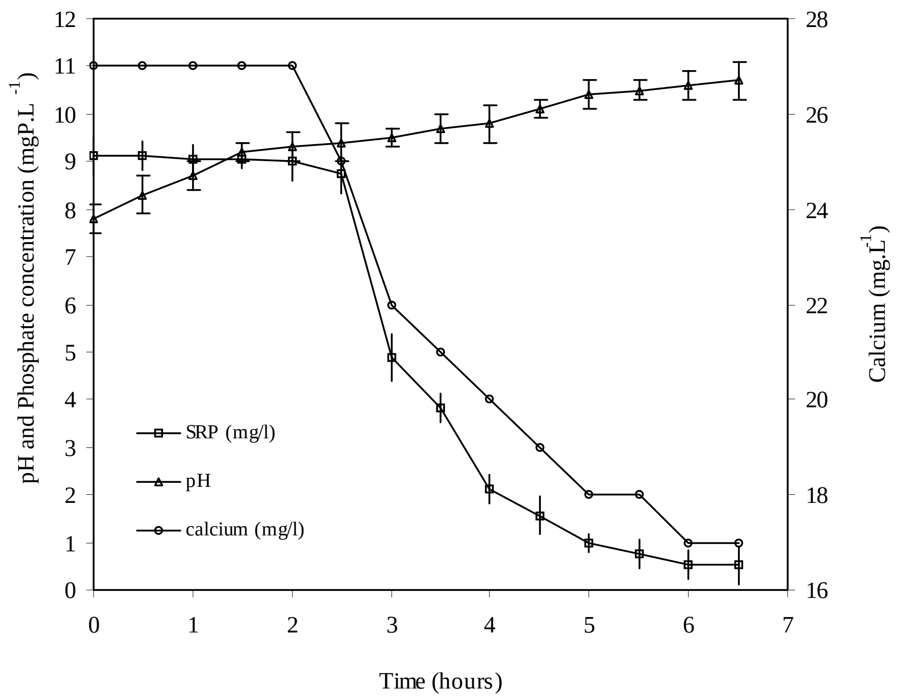


Figure 7.5. Precipitation of calcium phosphate minerals from GDW-final in the presence of photosynthesising algae.

It appears from these results that the mechanism for calcium-bound removal is regulated not only by the pH and various other ionic species, but more importantly by the ratio of calcium to dissolved phosphorus (Ca/P). In the case of calcite co-precipitation the Ca/P molar ratios before the onset of precipitation was in excess of 525, where the phosphate concentration (< 0.3 mg P.ℓ^{-1}) was low enough not to inhibit calcite crystal formation. According to Hartley *et al.* (1997) soluble reactive phosphorus concentrations near 0.33 mg P.ℓ^{-1} caused inhibition of calcite precipitation in abiotic conditions. In the case of tricalcium phosphate (TCP) formation the Ca/P molar ratios before the onset of precipitation was not more than 20, which ruled out the formation of calcite due to the high phosphate inhibition factor. Precipitation of magnesium phosphate and especially struvite ($MgNH_4PO_4$) was not considered in this study because the ratio of Mg/Ca is about 0.2 in GDW-final, which is expected to rather favour calcium phosphate phase precipitation (Abbona *et al.*, 1986). Also the Ca/P for GDW-final was 3.5 which suggests that the formation of HAP and/or TCP is the most probable mechanism involved in phosphate removal from this type of wastewater.

Precipitation of ionic crystals from solution is notoriously difficult to describe rigorously. However, many such reactions follow a general pattern where a period of very slow precipitation or reactant removal is followed first by rapid removal and then by further slow removal as the reactant concentration approaches an equilibrium value (Ferguson *et al.*, 1973). The same general pattern is followed in Figure 7.4 where the graphs of calcium and soluble reactive phosphorus seems to indicate that a precipitation mechanism is indeed involved. During the precipitation reaction, the culture medium cleared as the algal cells settled to the bottom of the flask. This was demonstrated by a rapid decrease in suspended chlorophyll-a concentration before and after precipitation (from 413 to 126 μgl^{-1}). The algal cells self-flocculated in a manner similar to that described by Koschel *et al*. (1983), in studies of algal assemblages in Lake Breiter Lucin. It is likely that precipitation occurs on or in close vicinity of the algal cells where the highest pH gradients are expected (Hartley *et al*., 1996). Table 7.2 shows the fractionation of phosphorus in a mixed sample from the same experimental apparatus, which indicates that the majority of phosphorus is indeed in the chemical precipitate form.

Table 7.2: The fractionation of phosphorus after calcium phosphate mineralisation.

Parameter	Phosphorus ($mgP.L^{-1}$)	Percentage
Soluble reactive phosphorus	0.62	5.5
Iron-bound phophates	0.02	0.2
Calcium-bound phosphates	9.81	87.3
Organic phosphates	0.79	7
Total phosphorus	11.24	-

Iron-bound phosphate levels were very low which would be expected as sewage contains relatively low concentrations of iron. The organic phosphate fraction is probably incorporated with the algal biomass present in the culture medium. A small fraction still remains in the soluble form (5.5% of total P), which was expected given the presence of slowly biodegradable organic compounds (CODs 50-60 mgl^{-1}). Organic compounds such as humic, fulvic and tannic acids are known inhibitors of calcium hydroxyapatite precipitation (Inskeep and Silvertooth, 1988). When the medium was acidified with dilute hydrochloric acid the release of phosphate coincided with a rise in calcium concentration (Figure 7.6) similar to what it was before the precipitation experiment, thereby emphasising the pH-dependence of the reaction mechanism.

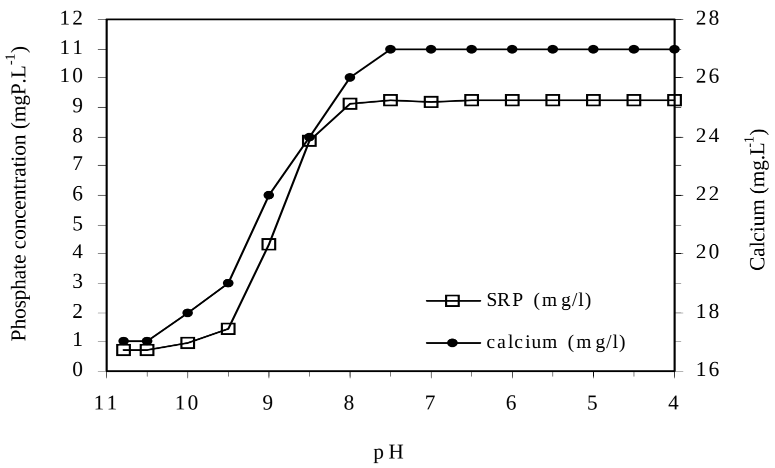


Figure 7.6. The release of chemically bound phosphate during the addition of acid, showing the relationship between pH and soluble reactive phosphorus concentration.

7.3.3 Phosphate removal in IAPS Final Water

Where Dekker (2002) had used final water from the GDW to demonstrate the phosphate removal potential of the I-HRAP, Wells (2005) repeated the study using the final water from the IAPS system. Figure 7.3 shows a consistent removal to levels below 5 mg.ℓ$^{-1}$ which is well below the 10 mg.ℓ$^{-1}$ discharge standard.

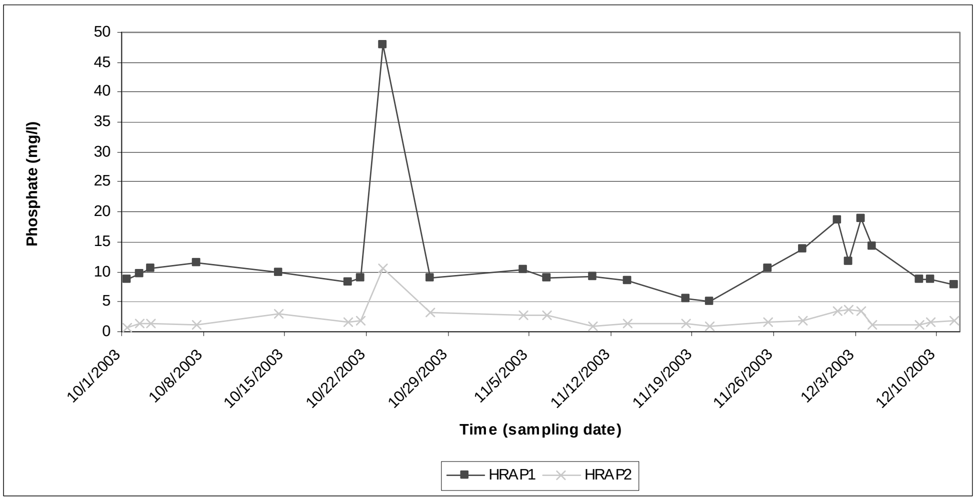


Figure 7.7: Phosphate removal in the IHRAP treating an IAPS final water**.**

In an attempt to confirm the phosphate removal mechanism proposed by Dekker (2002), Wells (2005) used *in situ* pH adjustment of the I-HRAP rather than the flask study methodology. This was done by adding high COD effluent directly to the I-HRAP. The results in Figure 7.8 clearly show that during periods of low pH, there is a corresponding increase in phosphate levels and this is reversed when the pH is allowed to rise again. A correlation coefficient of r = -0.99 for this observation indicates an extremely good inverse correlation in this relationship. These observations support Dekker's (2002) *in vitro* findings, and the theory that phosphate is removed in this system by precipitation. It should, however, also be borne in mind that a higher pH could also be indicative of increased algal activity and, thus possibly greater phosphate uptake.

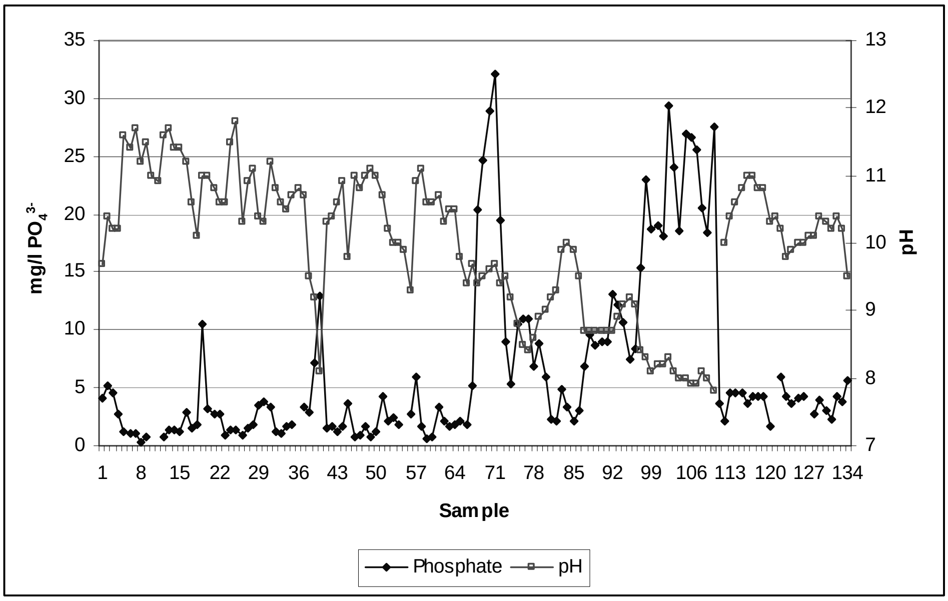


Figure 7.8: Phosphate removal showing an inverse relationship with pH.

7.4 CONCLUSION

- This research has demonstrated that it is possible in the I-HRAP to achieve average phosphate residuals of below 1 mg P.ℓ$^{-1}$ by calcium phosphate precipitation at pH >10.
- The I-HRAP can be used as a free-standing unit operation.
- Where the calcium concentration is a limiting factor, dosing with small amounts of lime and/or calcium chloride might be considered.
- The process offers considerable hope for savings in the chemical and capital costs associated with phosphate removal because of the possibility of utilising passive algal photosynthesis in a low-cost operation. This process could potentially be applied for treatment of final discharge waters emanating from a sewage treatment works, or indeed other treatment systems, such as for wine industry wastewater.

8. THE I-HRAP AS A TERTIARY TREATMENT UNIT OPERATION 3: NITROGEN AND COD REMOVAL

8.1 INTRODUCTION

Nitrogen can cause various problems in receiving water bodies depending on what form it is present. Ammonia is toxic to aquatic organisms, especially the higher forms such as fish at concentrations as low as 0.5 $mg.\ell^{-1}$ (Barnes & Bliss, 1983). In addition, excess ammonia can lead to oxygen depletion in water when nitrifying bacteria utilise oxygen for the oxidation of ammonium ion to nitrate, during nitrification (Horan, 1996). High nitrate levels can cause infant methaemoglobinaemia (Barnes & Bliss, 1983). Up to the age of about six months, infants have an incompletely developed digestive system and accumulate nitrite ions which enter the bloodstream. In the blood the nitrite is reoxidised to nitrate, using haemoglobin as the oxidising agent. This reduced form of haemoglobin (methaemoglobin), lacks the ability to bind with oxygen, effectively leading to oxygen starvation (Barnes & Bliss, 1983; Horan, 1996).

Perhaps the most widely known effect from the discharge of high nitrates and phosphates is eutrophication. While the presence of a small amount of diverse algal species is beneficial to a healthy aquatic ecology, high levels of nutrients often stimulate problematic algal blooms. During the day, these blooms contribute oxygen to the water body but at night, when photosynthesis stops, respiration results in a high oxygen demand. This fluctuation in dissolved oxygen is often to the detriment of other life forms (Horan, 1996). Seasonal death and decay of large masses of plants and algae may also lead to exhaustion of DO and odour generation (Barnes & Bliss, 1983).

Nitrogen control in treatment plants focuses on ensuring that nitrogen appears in the effluent in the desired form and concentration, and providing for nitrification activity is often sufficient to alleviate ammonia toxicity and oxygen demand in receiving waters (Barnes & Bliss, 1983). Where more complete nitrogen removal is required, additional or alternative procedures need to be employed. The most commonly used method being the coupling of nitrification to denitrification processes. As these two reactions occur under different physical conditions, they must either be separated in a multi-stage system or in different zones within the same reactor (Horan, 1996). Plant types used for nitrification include trickling filters, rotating disc filters, activated sludge and two stage activated sludge while denitrification takes place in systems such as the anaerobic filter, anaerobic fluidised bed and combined sludge system with anoxic zones (Barnes & Bliss, 1983). Nitrogen removal may also take place in waste stabilisation ponds (Gloyna & Tischler, 1979; Mara & Pearson, 1986).

While the operation of the IAPS had shown generally good removal of ammonia, levels exceeding the 3 $mg.\ell^{-1}$ were monitored. Nitrate removal was found to be quite erratic. With this in mind the I-HRAP system was investigated as described previously to determine what level of removal could be achieved where fitted as a polishing unit attached to the IAPS operation

8.2 MATERIALS AND METHODS

Materials and Methods used were as described in Chapter 3.

8.3 RESULTS AND DISCUSSION

8.3.1 Nitrogen Removal

It is evident from results reported in Chapter 5 that at times an increase in ammonia levels was observed in the HRAP of the IAPS. However, in a 12 month study of nitrogen removal in the I-HRAP (Figure 8.1) it was shown that ammonia levels were successfully decreased to below South African discharge standards of 3 mg.ℓ$^{-1}$ ammonia (South African National Water Act No. 36 of 1998).

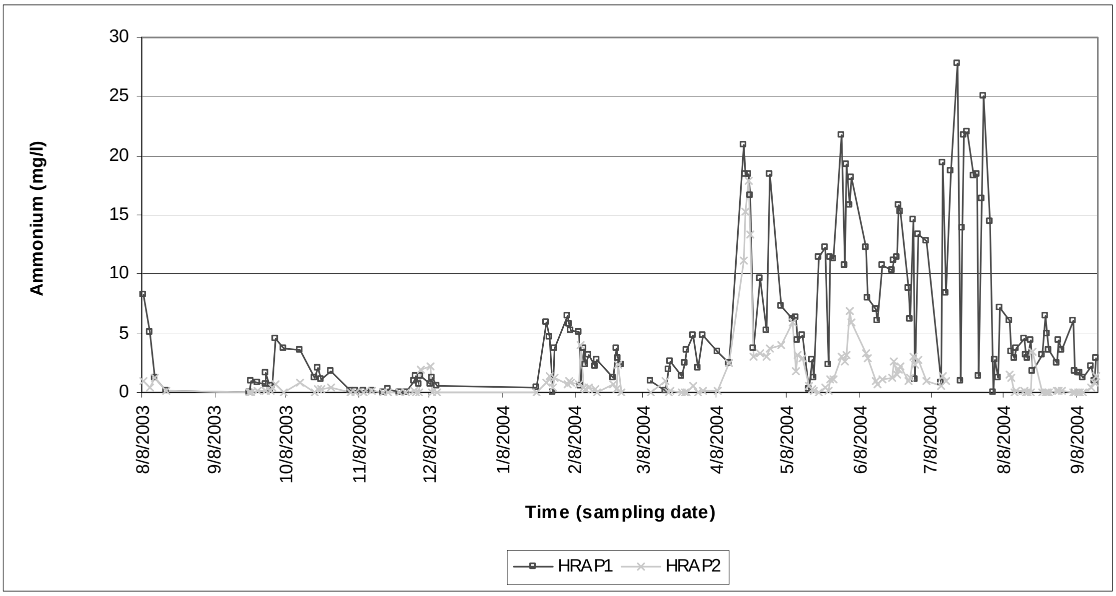


Figure 8.1: A 1-year study of ammonia removal in the I-HRAP treating final water from the IAPS plant.

As this water is continuously mixed and also attains pH values of up to pH 11, the most probable cause of ammonia removal in the I-HRAP is volatilisation (Reed, 1985; Gómez *et al.*, 1995; Nurdogan & Oswald, 1995; van der Steen *et al.*, 1998). Shilton (1996) also found ammonia volatilisation made a significant contribution to the removal of nitrogen from ponds treating piggery wastewater. Algal uptake may also be responsible for some of the ammonia removal (Nurdogan & Oswald, 1995; van der Steen *et al.*, 1998).

The mean ammonia level in the final effluent over the study period was <1.5 mg.ℓ$^{-1}$. This low level was consistently achieved in the system and remaining under the DWAF standard (3 mg.ℓ$^{-1}$) 92% of the time and under 0.5 mg.ℓ$^{-1}$ 68% of the time. This is a considerably better performance than activated sludge or trickle filters where ammonia levels of between 10 and 40 mg.ℓ$^{-1}$ are quite common (Horan, 1996). It also appears to be better than ordinary WSP, where effluent ammonia values of between 5 and up to 50 mg.ℓ$^{-1}$ have been reported (Racault *et al.*, 1995; Mendes *et al.*, 1995;

Ceballos *et al*., 1995). High rate ponds studied by El Hamouri *et al*. (1995) in Morocco and Green *et al*. (1996) in California also had mean ammonia levels of no lower than 7.8 and 5.3 mg.ℓ^{-1} respectively. With seasonal CaO addition and algae separation units in an advanced tertiary high rate pond, Nurdogan and Oswald (1995) were able to obtain effluent ammonia levels of between 2 and 3 mg.ℓ^{-1}.

Figure 8.2 shows that the high levels of nitrate recorded in HRAP of the IAPS were not effectively removed in the I-HRAP tertiary treatment operation and the results were quite variable. In some cases a small removal function was followed by an accumulation of nitrate in the I-HRAP. This was probably due to the aerobic conditions of the HRAP allowing nitrification to take place but inhibiting denitrification.

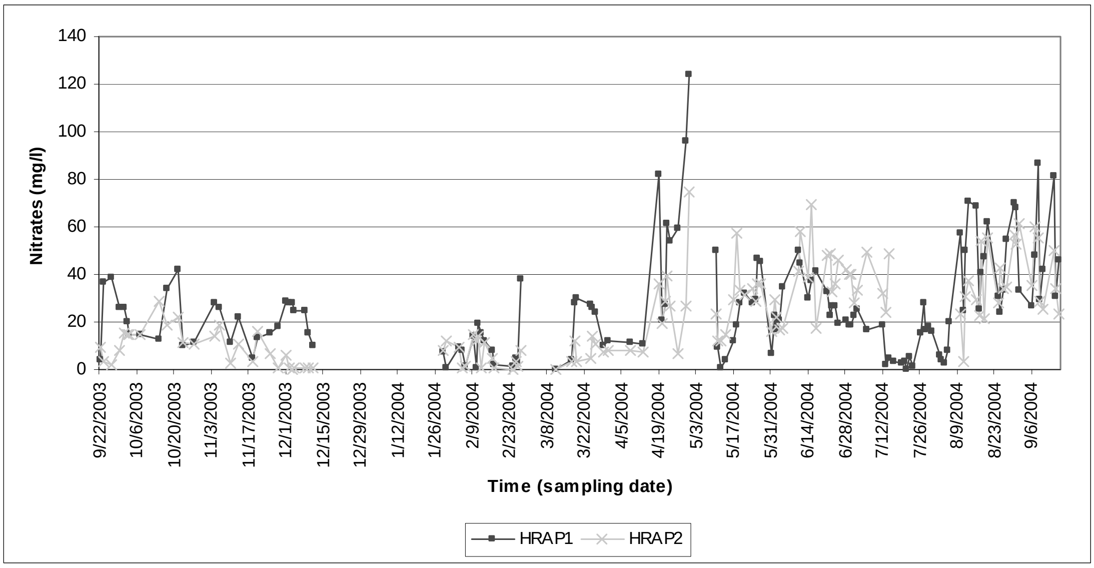


Figure 8.2: A 1-year study of nitrate removal in the I-HRAP treating final water from the IAPS plant.

8.3.2 Chemical Oxygen Demand Removal

Although the principle objective of the I-HRAP operation was enhanced nutrient removal and disinfection, the system was also monitored for its effect on COD removal. A pattern of CODt addition and removal, similar to that reported in Chapter 5 was also observed in the I-HRAP and its ASP operated in the IAPS series, with only a slightly improved COD level in the final effluent (Figure 8.3). Once again, the COD present in the I-HRAP can mostly be attributed to algal biomass.

It is evident from Figure 8.3 that the operation of the I-HRAP had very little effect on the further removal of CODs. This was true for CODt as well, where only an additional 6% reduction in CODt was recorded for the I-HRAP and ASP operation.

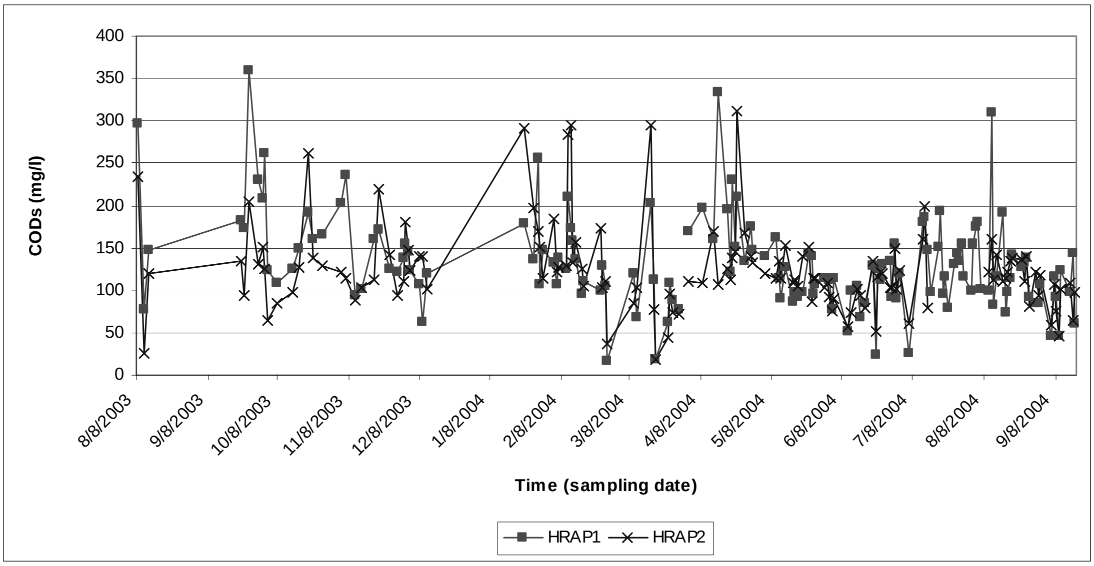


Figure 8.3: A 1-year study of soluble chemical oxygen demand removal in the I-HRAP.

8.4 CONTROL EXPERIMENT FOR I-HRAP VS HRAP RETENTION TIME

An experiment was undertaken to control for whether the I-HRAP was really necessary or whether simply increasing the retention time in a single HRAP as a unit of the IAPS would provide the same effect. The retention time in HRAP1 was thus doubled to 6 days HRT and the results compared to the average effluent quality obtained with the HRAP followed by the I-HRAP configuration where each was operated at a 3-day HRT.

Phosphate results for monitoring the HRAP at the 6-day HRT was as shown in Figure 8.4.

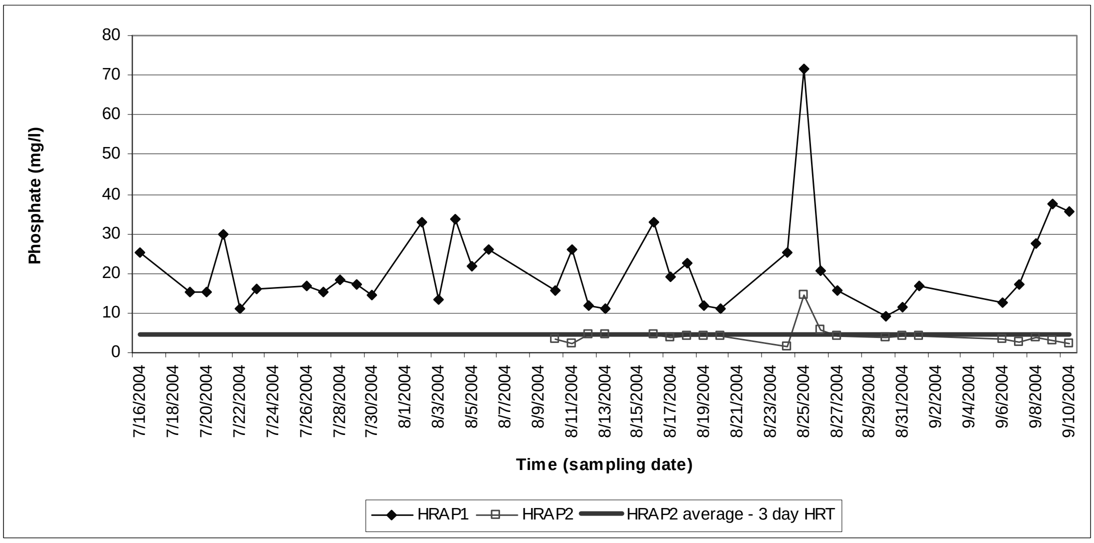


Figure 8.4: Comparison of phosphate removal in the HRAP at a 6 day hydraulic retention time (HRAP1) compared to the I-HRAP following the IAPS configuration (HRAP 2) operated at a 3 day HRT.

The results show that even with a 6 day HRT, the HRAP unit of the IAPS was still not able to reduce phosphate levels to that of the I-HRAP operated at 3-day HRT. Immediately the second HRAP was brought on line, phosphate removal improved. A similar effect was observed for *E. coli* removal, where one HRAP was unable to achieve 100% inactivation even with the doubling of HRT. The reason for this enhanced treatment requires further investigation. It appears, however, that the gradual improvement of water quality through progressive steps (ponds) is the most effective way to manage water in a ponding system.

8.5 SUMMARY OF I-HRAP PERFORMANCE AS A TERTIARY TREATMENT UNIT OPERATION

Table 8.1 shows the results for the most stable operating conditions achieved over a period of approximately three months, within the eighteen month study on the operation of the I-HRAP system.

The phosphate reduction of 86% achieved was similar to the average for the whole study. An ammonia removal rate of nearly 90% was monitored during this period. The nitrate removal was better than indicated by the 18 month mean, with the final effluent containing a mean nitrate of 11 mg.ℓ^{-1}, which is within the South African discharge standards (South African National Water Act No. 36 of 1998). Although the *E. coli* figures indicate a mean of 4.8 cfu.100 mℓ^{-1}, a 0 cfu.100 mℓ^{-1} count was observed over 80% of the time, with 1 result of 65 cfu.100 mℓ^{-1} inflating the mean.

Table 8.1: A summary of the performance of the I-HRAP tertiary treatment unit operation (HRAP2) and following the HRAP operated as a component unit of the IAPS (HRAP 2)
.

	CODt (mg.ℓ^{-1})	CODs (mg.ℓ^{-1})	NO_3^- - N (mg.ℓ^{-1})	NH_4^+ - N (mg.ℓ^{-1})	PO_4^{3-} - P (mg.ℓ^{-1})	*E. coli* (cfu.100 mℓ^{-1})
PFP effluent	307	203	5.6	12.1	15.7	5.8×10^5
HRAP1	175	128	19.3	5.6	12.1	6.7×10^3
HRAP2	169	124	11	1.4	2.3	4.8

9. I-HRAP-LINKED DENITRIFICATION USING ALGAL BIOMASS AS THE CARBON SOURCE

9.1 INTRODUCTION

Although effective disinfection and phosphate and ammonia removal had been demonstrated for the I-HRAP tertiary treatment operation, nitrate removal performance was found to be variable in both the IAPS and the I-HRAP experimental investigations outlined in previous chapters, with little effective overall change in nitrate status being effected. The addition of high COD wastewater to the HRAP had been shown to successfully induce anoxic conditions and denitrification at night (results not shown), but this was also found to adversely impact the nutrient removal and disinfection performance achieved in the I-HRAP.

In approaching this problem it was decided to undertake a more detailed investigation of the potential for denitrification linked to I-HRAP operation using the algal biomass produced in the system as the carbon source. This work was initiated by Dekker (2002), and then extended and followed up Clark (2001) and Neba (2003).

9.1.1 Denitrification

Denitrification is the reduction of nitrate to nitrogen gas and this process occurs in four steps:

$$\underset{\text{Nitrate}}{NO_3^-} \longrightarrow \underset{\text{nitrite}}{NO_2^-} \longrightarrow \underset{\text{nitric oxide}}{NO} \longrightarrow \underset{\text{nitrous oxide}}{N_2O} \longrightarrow \underset{\text{nitrogen gas}}{N_2}$$

Each step is catalysed by a reductase enzyme system. Denitrification requires an electron donor which can be organic material or reduced compounds such as sulphide or hydrogen. Heterotrophic denitrifying bacteria require an organic carbon source for respiration and growth. A variety of organic compounds have been used, such as methanol (Borregaard, 1997) and formate (Soares *et al.*, 1991) as well as different industrial wastes including fusel oil (Klapwijk *et al*., 1981) and primary sewage sludge (Moser-Engeler *et al.*, 1998). Numerous species of facultative denitrifying bacteria, including *Pseudomonas, Micrococcus, Achromobacter* and *Bacillus* are capable of converting nitrate to nitrogen gas. Nitrate replaces oxygen in the respiratory process of the organisms capable of denitrifying under anoxic conditions. Autotrophic denitrifying bacteria utilise hydrogen or sulphur as electron donors, and carbon dioxide or bicarbonate is used as a carbon source for microbial cell synthesis (Anderson and Levine, 1986; Flere and Zhang, 1998). Generally, autotrophic denitrifiers grow slowly and denitrification rates are lower, whereas contamination of denitrified water with organic materials requires extensive post-treatment in the heterotrophic processes. Most full-scale applications make use of heterotrophic processes due to their efficiency, high specific denitrification rate and operational simplicity.

9.1.2 Biological Denitrification Processes

Biological upflow fluidised bed (BFB) technology has been used since the 1970s for industrial waste treatment and more recently for denitrification of contaminated groundwater and municipal waste (Sutton and Mishra, 1990). The BFB technology has the potential to remove large quantities of nitrogen with relatively small space requirements, which is of particular importance in densely populated cities with limited space for expansion. Semon *et al*. (1997) described the results of a BFB with sand media that was operated at a flow rate of 1 296 $m^3.d^{-1}$, and an average loading rate of 1 843 kg $NO_3.m^3.d^{-1}$, treating final effluent from a conventional activated sludge treatment facility. Approximately 3 mg $methanol.mg^{-1}$ influent nitrate was used as the external carbon source. Denitrification in the reactor was very rapid resulting in average effluent nitrate levels of 0.4 $mg.\ell^{-1}$ (influent 7.7 $mg.\ell^{-1}$) with reaction times of less than 5 minutes. Borregaard (1997) reported another high rate process where a combination of nitrification-denitrification was achieved in a fixed-film system utilising methanol as carbon source. The Biostyr unit has polystyrene granules which offer a high specific surface area and are therefore very compact. This process removed at least 70% total nitrogen in order to comply with Danish final discharge standards.

The sequential batch reactor (SBR) can be modified to provide advanced secondary treatment, nitrification, denitrification and biological nutrient removal. An SBR treatment cycle consists of timed sequences which typically includes the following steps: fill, react, settle, decant, idle (Arora *et al*., 1985). When biological nutrient removal is desired, the steps in the cycle are adjusted to provide anoxic or anaerobic periods within the standard cycles. Surampalli *et al*. (1997) reported complete nutrient removal using a SBR that followed a specific sequence: Aerated fill (COD removal, nitrification, and phosphorus uptake), react (COD removal, nitrification, phosphorus uptake), settle (waste P-containing sludge), idle (denitrification, growth of P-removing bacteria). Biological dephosphatation by activated sludge under denitrifying conditions has been optimised for a full-scale activated sludge treatment plant, where denitrifying phosphorus removing bacteria (DPB) were cultivated in an anaerobic-anoxic SBR (Kuba *et al*., 1997). A problem with the conventional system is the competition for COD between phosphorus and nitrate removing organisms, since organic substances in municipal wastewater are often limiting. After determining the culture conditions for DPB it was clearly shown that 50% of the phosphorus removal occurs via denitrifying activities, resulting in less competition for organic substrate.

Semi-passive treatment of wastewater to achieve denitrification in the form of subsurface flow constructed wetlands has also been reported, where hydroperiod manipulation and vegetation presence/absence in two-stage treatment systems were applied (Kemp and George, 1997).

9.1.3 Algal Extracellular Polymeric Substances as Carbon Source

Changes in the composition of extracellular polymeric substances (EPS) in activated sludge during anaerobic storage have been reported due to microbial degradation (Nielsen *et al*., 1996). The results showed that a fast decrease in total sludge protein and carbohydrate took place within 3 days of anaerobic storage as a result of degradation processes, which accounted for approximately 20% of the organic fraction. Stress production of EPS by microalgae and cyanobacteria is known to respond to changes in several external factors, such as nitrogen concentration, irradiance or temperature (Arad *et al*., 1992; Moreno *et al*., 1998) where carbohydrate and protein are found to be the major components (Flaibani *et al*., 1989).

In this study the following was considered:

1. Does stress induced release of EPS occur in I-HRAP algal biomass?;
2. Can EPS be effectively used as a carbon source for biological denitrification?;
3. Can these reactions be scaled up into an effective bioprocess unit operation attached to the I-HRAP operation?

9.2 MATERIALS AND METHODS

9.2.1 Laboratory experiments

Batch denitrification using algal biomass was conducted in a covered 2-litre flask at 25ºC while mixing with a magnetic follower (Figure 9.1). Fresh algal biomass was harvested from the I-HRAP and concentrated to give a final total COD value of about 2000 mg.ℓ^{-1}. Nitrate was supplemented in the form of KNO_3 to give a final concentration of 20 mg N.ℓ^{-1}. Samples were taken every 12 hours and analysed for nutrient levels and polysaccharides (with the addition of KNO_3 when NO_3-N<1 mg.ℓ^{-1}).

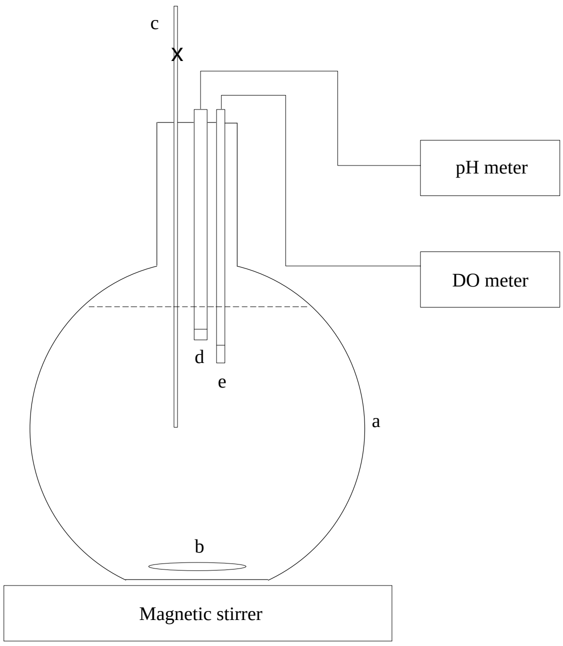


Figure 9.1: Experimental apparatus used for batch denitrification experiments. (a) Foil covered glass flask (b) magnetic follower (c) sampling port (d) pH-electrode (e) DO-electrode.

Continuous denitrification was conducted in a 7.4 l covered upflow glass column with a height of 1.17 m and internal diameter of 9.0 cm (Figure 9.2). The reactor was seeded with 5 litres concentrated algal biomass (COD 37 500 mg.ℓ$^{-1}$, chlorophyll-a 53.8 mg.ℓ$^{-1}$) obtained from the I-HRAP and the feed consisted of settled I-HRAP medium with KNO_3 supplementation to a final concentration of 20 mg NO_3-N.l^{-1}. The NO_x-N mass balance was determined by setting up another smaller column (volume 540 mℓ) that operated in parallel to the 7.4 l column under the same conditions. Gas produced in the reactor was collected in a gas meter filled with dilute hydrochloric acid solution. Both systems were operated in at a temperature of 25 ± 2 ºC.

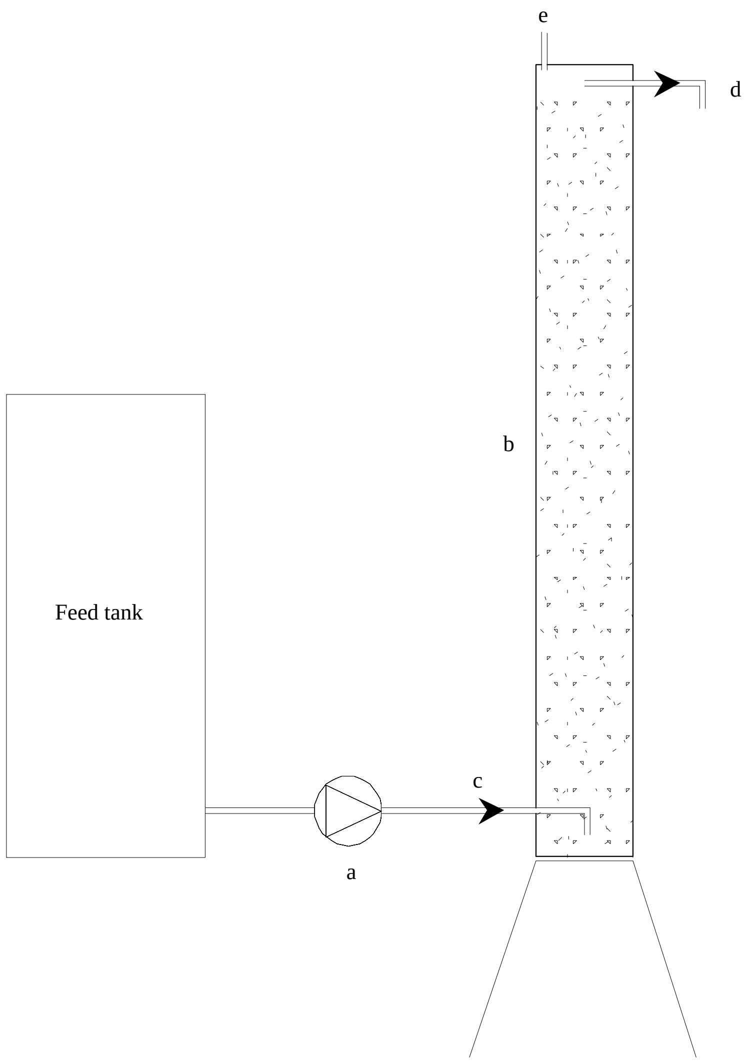


Figure 9.2. Upflow anoxic column with settled algal biomass. (a) Peristaltic pump (b) covered cylindrical glass column (c) influent (d) overflow (e) gas vent.

9.2.2 Analytical Methods

Analysis of all parameters was undertaken as previously described in Chapter 3.

Biogas production was measured by acid solution displacement and the gas content was determined by gas chromatography using a thermal conductivity detector. Total polysaccharides (TPS) in the cultures and EPS in the supernatants, resulting from culture centrifugation at 5 000 G for 15 min, were determined by the phenol-sulphuric method (Dubois *et al*., 1956). Capsular polysaccharide (CPS) was determined similar to EPS, but after stirring the sample in distilled water at 50ºC for 30 min (Vincenzini *et al*., 1990). The Lowry method (Lowry *et al*., 1951) was applied for protein determination.

9.3 RESULTS AND DISCUSSION

9.3.1 Nutrient Release in Algal Settling Ponds

A depth profile in the ASP and I-ASP was undertaken to monitor nutrient release and nitrate removal at the bottom of these ponds. The results are shown in Table 9.1 where the ‘middle samples’ were drawn just above the algal sludge bed over a period of one week.

Table 9.1. Depth profile in each algal settling pond showing the correlation between phosphate and calcium levels. Note that ASP follows HRAP, and I-ASP follows I-HRAP.

		ASP			I-ASP		
Parameter		Surface	Middle	Bottom	Surface	Middle	Bottom
pH		8.7	7.6	5.7	10.4	8.9	8.3
sol-P	$mgP.L^{-1}$	3.9	10.1	58	0.91	1.21	9.73
Calcium	$mg.L^{-1}$	26	27	27	18	18	32
NH_3	$mgN.L^{-1}$	3.04	10.2	18.3	0.54	1.77	4.98
NO_3	$mgN.L^{-1}$	5.3	0.4	0.1	14.3	10.7	1.2

Note that HRAP had been treating the overflow from the facultative pond and I-HRAP received GDW-final for phosphate removal as described in Chapter 7. For I-ASP there was a correlation between the phosphate and calcium concentrations, but this time the calcium phosphate precipitate seemed to re-solubilise due to a decrease in pH, which is in agreement with results already reported. Fermentative phosphorus release is unlikely because green microalgae are known to degrade very slowly under anaerobic conditions. The P-release by PAO is also unlikely due to the operation of I-HRAP, which did not meet the requirements for PAO cultivation according to Mino *et al*., 1998. Even more nutrients were release from the ASP than I-ASP, possibly due to a higher sludge volume accumulated. There is no evidence to suggest that the released phosphate was from chemical origin as there was no simultaneous increase in calcium concentration. It is quite possible that PAO were present in large numbers and under these circumstances are likely to accumulate and release phosphate in an anoxic environment when there is organic substrate available in the form of fermentable products. The fact that nitrate removal and ammonification takes place in the algal sludge beds suggests that fermentable products are indeed available for denitrification and/or biological phosphorus release. These results suggest that when performing harvesting and drying of the algal biomass care should be taken to collect the drainage from the drying bed, which may be returned to I-HRAP for improving NH_3 and PO_4 removal.

9.3.2 Batch Denitrification Using Algal Biomass as Reagent

Accumulation of internally stored carbon in the form of starch and other products may occur in the microalgae during periods of high carbon flux (Preiss and Romeo, 1989; de Philippis *et al*., 1992). Under stress conditions, such as nitrogen starvation, the stored material may be released as polysaccharides, among other compounds, and serve as a carbon source for denitrifying organisms. Nitrate removal observed in the bottom of both the ASP and within the I-ASP indicates that easily biodegradable carbon is available to denitrifying organisms. When algal biomass was harvested from the I-HRAP and submitted to anoxic dark conditions, polysaccharide release was stimulated. Figure 9.3a shows a relationship between COD_t consumption and denitrification, and also the production and utilisation of polysaccharides within the system. Nitrate replenishment and removal is shown against the decline in available COD. Four phases were apparent in the removal process:

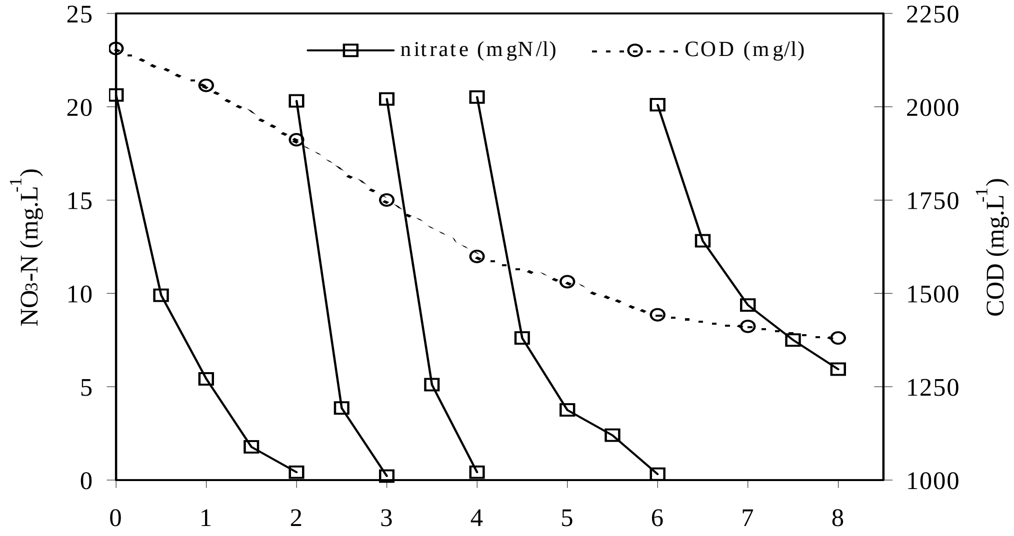
Day 0 to 2:
The system became anoxic within 2 hours (dissolved oxygen reduced from 14.5 to < 0.2 mg.ℓ^{-1}). De-repression of nitrate reductase enzymes has been reported to occur within a period of 40 minutes to 3 hours (Payne *et al*., 1971; Baumann *et al*., 1996). The rate of nitrate reduction may also be dependant on the rate of EPS hydrolysis into more easily accessible products. During the first 24 hours, polysaccharide production was higher than its rate of consumption (Figure 9.3b). It was assumed that fermentative organisms are responsible for the breakdown of released complex carbohydrate into more easily accessible carbon, which in turn is utilised by the denitrifying organisms. The algal biomass sourced from the I-HRAP appeared to contain a sufficient seed of facultative denitrifying organisms.

Day 2 to 4:
The rate of nitrate reduction appears to be directly linked to NO_3-N concentration. After each replenishment, when nitrate decreased to less than 5 mg.ℓ^{-1}, the rate of reduction also decreased as it apparently became the limiting factor. Between days 3 and 4 the rate of polysaccharide production appeared to equal its consumption.

Day 4 to 8:

The rate of nitrate removal slowed, possibly due to the decrease in TPS production rate, with available COD becoming the limiting factor.

A



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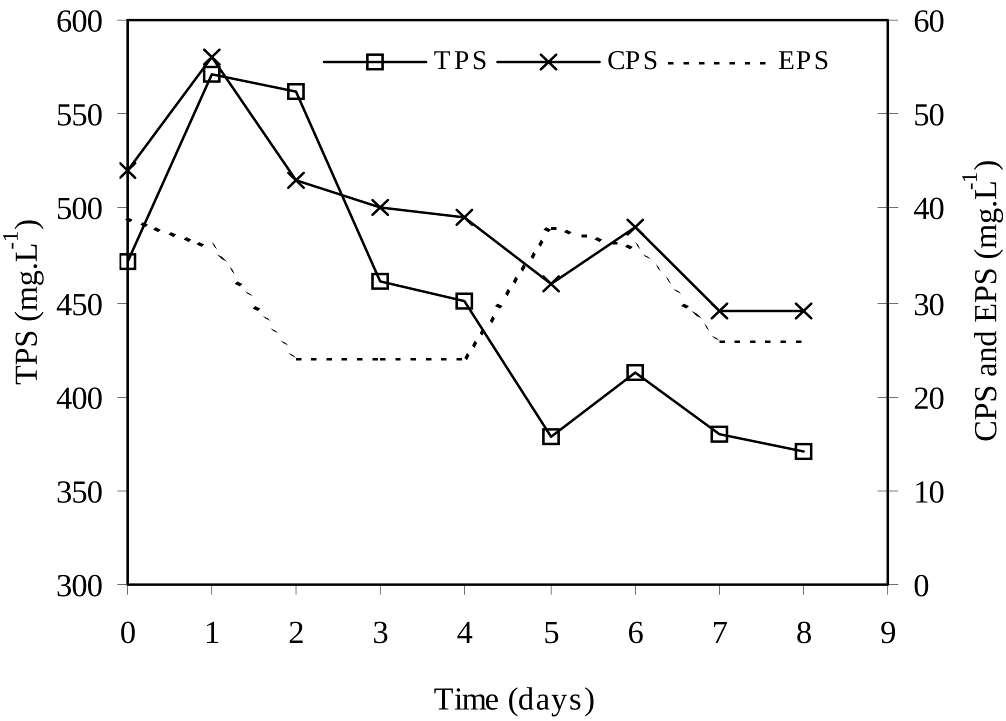

Figure 9.3: Batch denitrification using algal biomass as the carbon source and showing (A) repeated replenishment of nitrate against COD consumption and (B) polysaccharide release and utilisation.

Towards the end of the experiment the conversion of internally stored carbon into polysaccharide appeared to become exhausted. In this experiment 776 mg COD.ℓ^{-1} (36% of COD_t) was consumed while eliminating 94 mg NO_3-N, giving an average COD:NO_3-N removal ratio of 8.26:1 (mg/mg) over the 9 day period. The calculated stoichiometric value of TPS, which is expressed in glucose equivalents, gave a COD: TPS ratio of 1.07:1. Therefore the COD:NO_3-N ratio for the first 6 hours was 8.0:1, and indicates that 90% of the COD consumed was thus from polysaccharide origin.

It was calculated that 121 mg COD.ℓ^{-1} of I-HRAP-medium would be available for removing 15 mg NO_3-N.ℓ^{-1} and, according to the COD:NO_3-N ratio of 8.26:1, approximately 98% nitrate could theoretically be removed by utilising the produced and harvested algal biomass in an appropriate denitrifying reactor design.

Clark (2001) investigated the release of polysaccharides by HRAP harvested algal biomass under a range of stress conditions. In a series of detailed flask studies he showed the above results of polysaccharide accumulation were repeatable under a range of operating conditions, of which holding the biomass under anoxic conditions in the dark for 12 hours would provide the greatest release of TPS and EPS. He showed that maxima of 800 – 1000 mg.ℓ^{-1} TPS may be produced under these conditions and also confirmed the COD:NO_3-N ratio of 8:1 for the consumption of algal TPS in nitrate reduction.

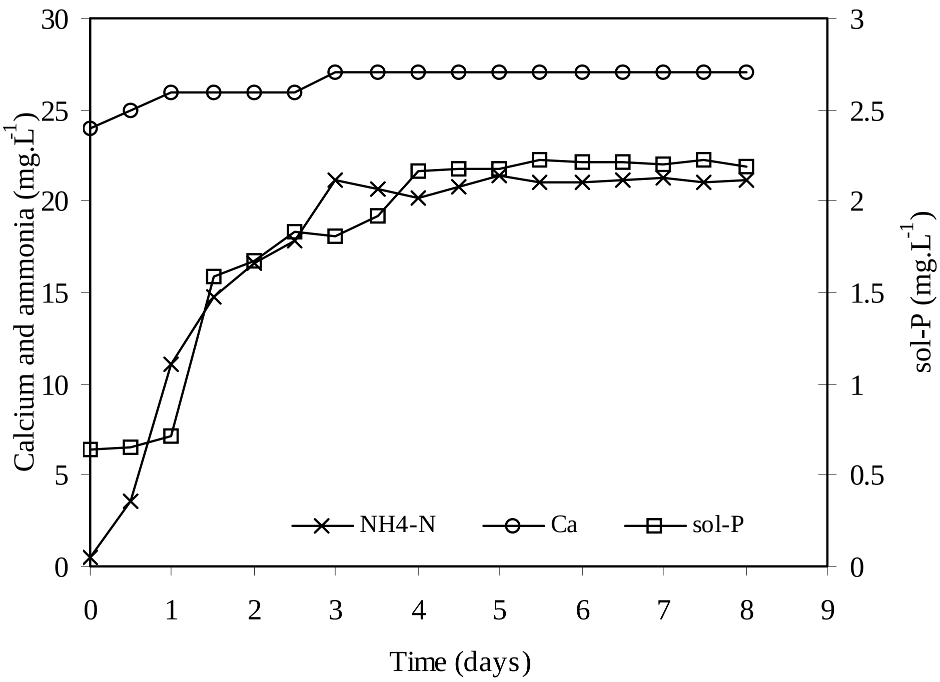

Figure 9.4: The release of phosphate and ammonia during denitrification

In the batch experiments, the release of phosphate and ammonia was also observed (Figure 9.4). During substrate fermentation ammonification occurred, presumably due to the breakdown of microalgal protein. The ammonia concentration rose to 20 mg N.ℓ^{-1} within 3 days, but then remained constant, possibly due to ammonia stripping at the elevated pH of the medium (pH>9.5). Phosphate release (from 0.6 to 2.2 mg P.ℓ^{-1})

correlates well with dissolved calcium release (from 24 to 26 mg Ca.ℓ$^{-1}$) and change in pH (from >10. to 9.6).

9.3.3 Continuous Denitrification in an Upflow Reactor

The upflow reactor column was loaded with settled algal biomass and operated in UASB mode for 45 days. The results are reported in Figure 9.5 with four stages of the experiment described below.

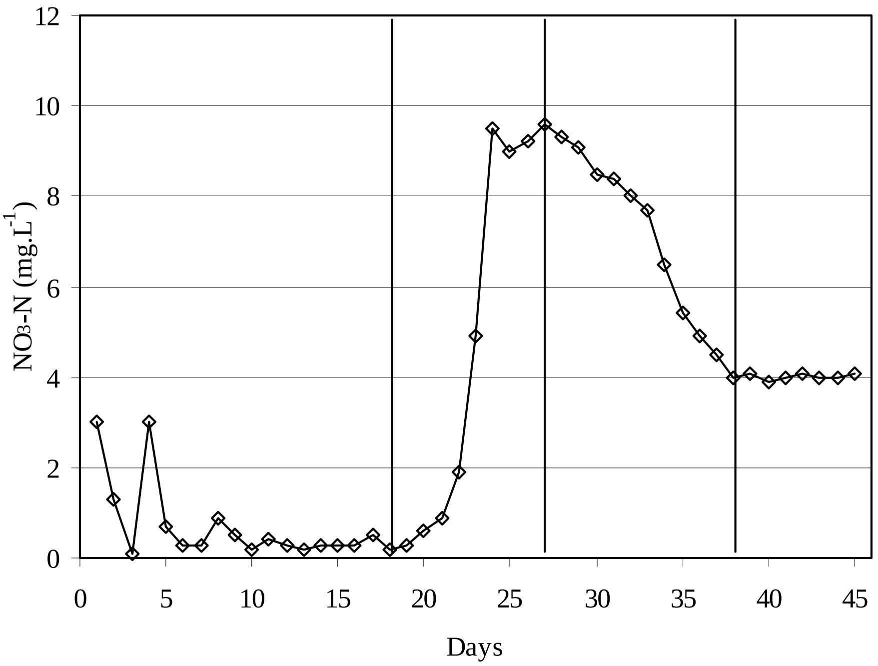

Figure 9.5 Performance of the denitrifying upflow column using settled algal biomass as reagent.

Day 1 to 18:

Denitrification started immediately with the HRT set at 1.03 days. The feed rate was increased from day 4 to give a retention time of 0.6 days. During the first 5 days the fermentative gas production was high, resulting in sludge piston formation, where the dense algal sludge bed caused considerable gas entrapment at the bottom of the reactor. When the buoyancy of the accumulated gas was high enough, a sporadic flotation of sludge occurred resulting in subsequent biomass washout. The retention time was adjusted to 0.5 days (from day 8) and the sludge bed stabilised with no more serious loss in algal biomass observed. Nitrate removal efficiency of 99% was achieved.

Day 19 to 27:

As the rate of stress induced release of algal polysaccharides declined, the rate of nitrate reduction decreased from 99% to 55%.

Day 28 to 38:

From day 27 fresh algal biomass was included in the feed. The COD_t in the feed was 335 mg.ℓ^{-1} and according to the batch experiment more or less 36% of the COD should be available for denitrification, thereby removing 14.6 mg N.ℓ^{-1}. Nitrate reduction improved with the addition of fresh algae and the performance increased gradually from 52 to 80% nitrate removed.

Day 39 to 45:

The column had stabilised with nitrate eliminated at a loading rate of 32 mg NO_x-N.ℓ^{-1} reactor volume per day.

The overall performance results for the upflow reactor denitrification experiment are presented in Table 9.2. The biogas analysis showed that only nitrogen was present and no carbon dioxide or methane was detected. Carbon dioxide would tend to remain in solution at pH above 9.5. The experimental study showed that this reactor removed 56.6 mg NO_x-N.d^{-1} of which 88% was recoverable as nitrogen gas, and indicates that anoxic denitrification is mainly responsible for nitrate removal.

Table 9.2: NO_x-N balance for the continuous upflow reactor denitrification experiment.

Hydraulic load	L.d^{-1}	3.7
Influent NO_3	mgN.L^{-1}	20
Influent NO_2	mgN.L^{-1}	3.2
Effluent NO_3	mgN.L^{-1}	0.8
Effluent NO_2	mgN.L^{-1}	2.3
NO_x removed	mgN.d^{-1}	56.6
N_2 gas collected	mgN.d^{-1}	50

In terms of practical application of these findings to nitrate removal in the I-HRAP, a denitrification unit of 18 m^3 would be required to treat the GDW-final water (15 mg NO_3-N.ℓ^{-1}).

9.3.4 Nutrient Removal Following Denitrification

Increases in phosphate due to pH reduction in the column, and in ammonia levels due to protein breakdown, could present a problem for the denitrification strategy using algal biomass as the carbon source. This may, however, be dealt with where the treated denitrification liquor passes to the I-HRAP. This was examined in a flask study where the column effluent was mixed with algae at the same concentration as the I-HRAP. The results are shown in Table 9.3.

Table 9.3: Removal of phosphate and ammonia from the denitrified medium in the presence of photosynthesising algae.

Time (hours)	pH	sol-P ($mgP.l^{-1}$)	Ca ($mg.l^{-1}$)	NO_3 ($mgN.l^{-1}$)	NH_3 ($mgN.l^{-1}$)
0	9.4	4.02	23	0.3	6.21
6	10.6	0.43	15	0.3	2.09

9.4 DENITRIFICATION AT PILOT-SCALE

Following the above studies by Dekker (2002) and Clark (2001), Neba (2003) undertook the scale-up evaluation of the algal denitrification column reactor at the EBRU Environmental Biotechnology Experimental Field Station in Grahamstown. This was located directly after the trickle filter humus tank. He used a 3 m^3 column packed with stone media that was fed directly from GDW final effluent (Figure 9.6). Nitrate levels ranged between 60 - 80 $mg.\ell^{-1}$.

Figure 9.6: Stone packed column reactor used for the scale-up evaluation of denitrification studies.

The reactor was operated at HRT ranging between 7 and 48 hours and a steady state period of 20 days allowed between each change of HRT. Algal feed was provided together with the I-HRAP feed water to the reactor and the active mass was that detained on the stone packing. Given the nature of the operation it was not possible to establish the algal load within the system at any one time.

Peak nitrate removal of 93.2% was recorded, but the best sustained average removal of 80.6% was recorded for HRT of 48 hours. Shorter retention resulted in reduced levels of removal. Results for the 48 hour HRT are shown in Figure 9.7.

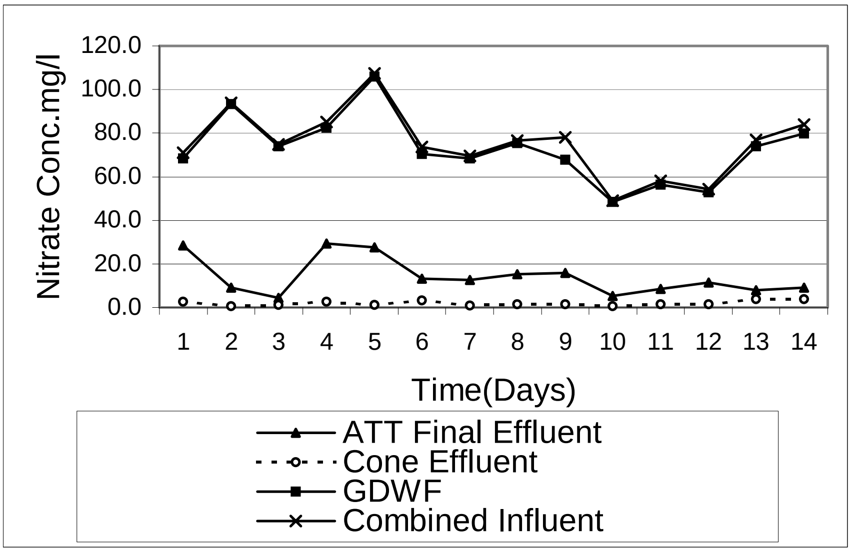

Figure 9.7: Nitrate removal in the algal denitrification packed column reactor showing effluent nitrate and algal load drawn from the I-HRAP and set at a hydraulic retention time of 48 hours.

Although no gas collection apparatus was fitted to the column reactor, Figure 9.8 shows that the nitrate was most likely denitrified to nitrogen gas since no accumulation of nitrite could be observed in the system.

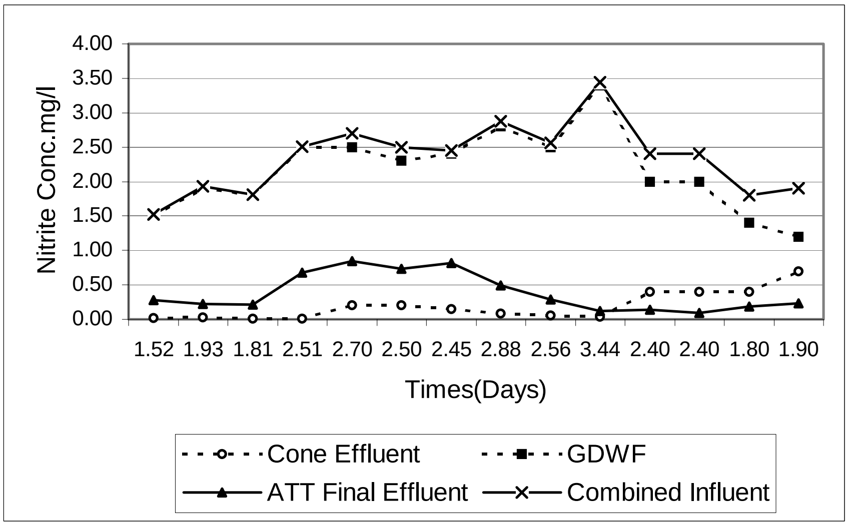

Figure 9.8: Nitrite removal in the algal denitrification packed column reactor showing influent nitrate and algal load drawn from the I-HRAP and set at a hydraulic retention time of 48 hours.

Although it was anticipated that the anoxic conditions in the reactor would result in a reduction in pH and hence the release of phosphate (Figure 9.9), the extended retention of algal biomass in the packed bed system resulted in the digestion of algal cellular matter and the release of larger amounts of ammonia than had been encountered in the laboratory experiments (Figure 9.10)

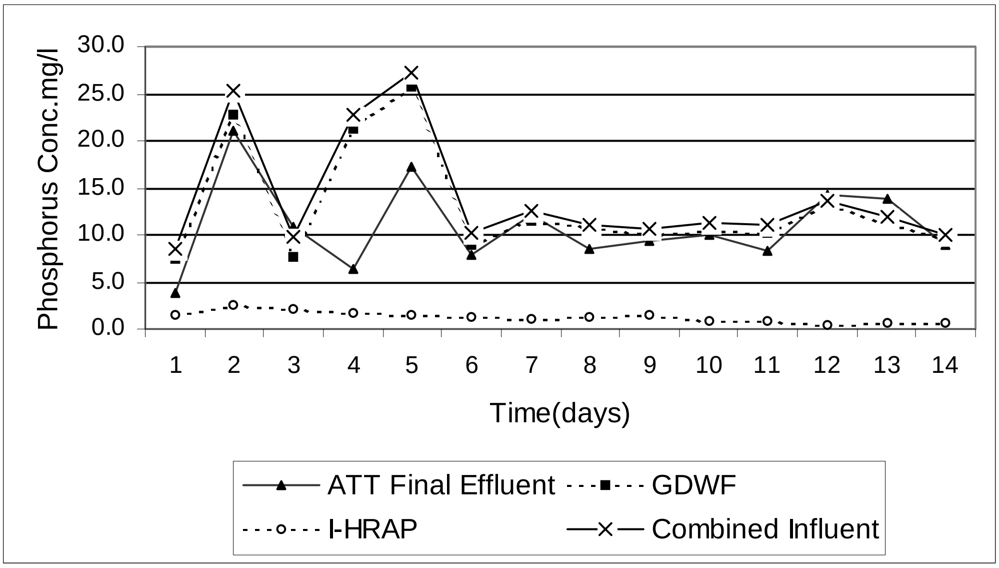

Figure 9.9: Phosphate release in the algal denitrification packed column reactor showing influent phosphate and algal load drawn from the I-HRAP, and set at a hydraulic retention time of 48 hours.

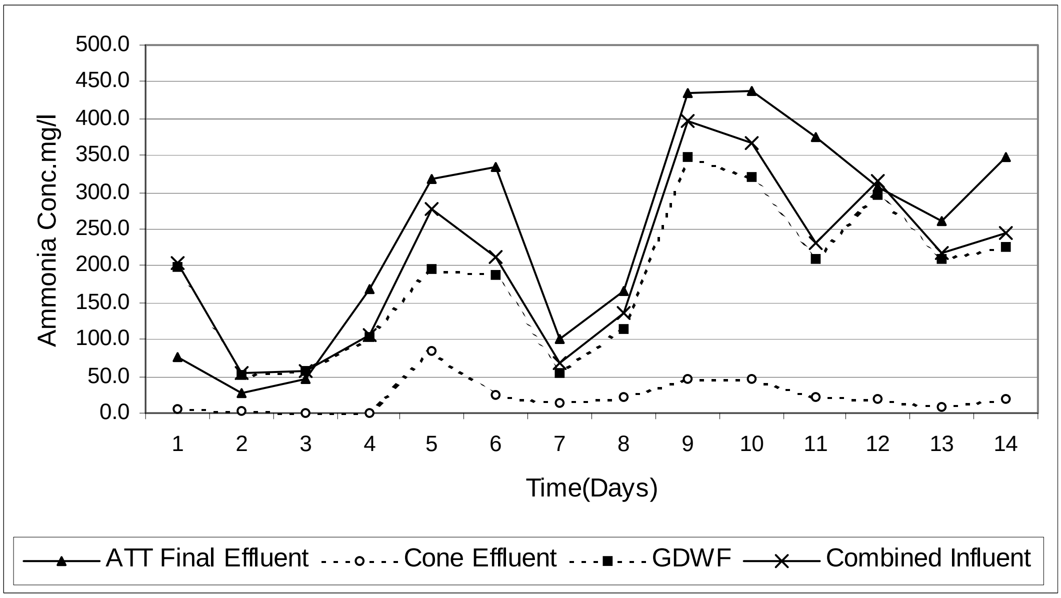

Figure 9.10: Ammonia production in the algal denitrification packed column reactor showing influent ammonia and algal load drawn from the I-HRAP, and set at a hydraulic retention time of 48 hours.

It is evident from this work that while good denitrification may be effected with the use of photosynthate release from algal biomass produced in the I-HRAP, it is counterproductive to allow the biomass to be retained within the system to the point where it begins to degrade. In this regard the packed column reactor used in the scale-up studies did not provide an ideal reactor environment as the algal biomass retention was not under direct control. Nevertheless, it was apparent at the outset that where nitrate removal is desired in an I-HRAP tertiary treatment operation, the use of a dual pond system would be required. The first would be used to generate algal production and initial polishing effects, and the second to complete precipitation of phosphate and ammonia stripping at the elevated pH in the system. A change in reactor design has been investigated and this is the subject of future research investigation by the group.

9.5 CONCLUSIONS

- Harvested algal biomass from the I-HRAP can be used as a reagent for denitrification by releasing polymeric substances, which probably serves as a carbon source for the biological denitrification process.
- Polysaccharide release was possibly due to stress conditions imposed in the system such as the anaerobic, dark environment.
- Phosphate and ammonia released from the algal floc during the denitrification process can be removed subsequently in the I-HRAP

Based on these findings the following process approach (Figure 9.11) was proposed for dealing with high nitrate levels using the I-HRAP system.

Settled algal biomass may be recovered from the primary I-HRAP and ASP and recycled to the denitrification unit located ahead of the primary I-HRAP unit.(see figure 9.11) Following denitrification, the stream passes through the primary to the secondary I-HRAP where phosphate is removed, primarily through calcium phosphate precipitation. Ammonia removal, primarily by alkaline stripping, would also be effected in the secondary I-HRAP and, in addition to that present in the effluent, would include the component released due to protein breakdown in the denitrification unit. During the anoxic phase, carbon dioxide produced in the denitrification unit would dissolve in the alkaline medium and, therefore, be available in the form of bicarbonate for subsequent photosynthetic uptake in the I-HRAP. Algae in the anoxic chamber should not be retained in the denitrification unit and thus on passing to the I-HRAP would still be viable for regeneration and floc formation in the raceway.

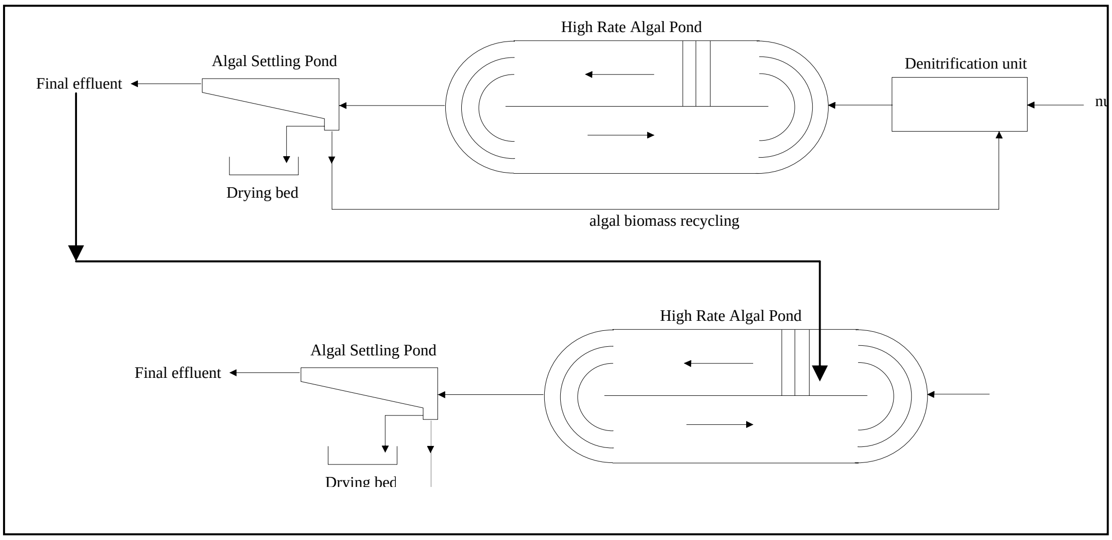

Figure 9.11: I-HRAP tertiary treatment process conceptualisation including denitrification of the influent stream with the use of algal biomass as carbon source.

While Chapters 6 – 8 of this report had detailed findings indicating the use of the I-HRAP as a tertiary treatment operation for disinfection, phosphate and ammonia nitrogen removal, little change in nitrate levels occur in the system due to the low COD loading at the polishing stage and thus entirely aerobic conditions are maintained throughout. The study reported in Chapter 9 indicates that the algal biomass generated in the I-HRAP may be used for the upstream operation of a denitrification unit, and that in this format a complete tertiary treatment of wastewaters may be provided by the free-standing I-HRAP unit operation.

Further work is required to investigate the performance of the I-HRAP in the removal of micro-pollutants such as endocrine inhibitors, and to track the fate of heavy metals in these systems.

10. OVERALL PERFORMANCE EVALUATION OF IAPS OPERATIONS

10.1 INTRODUCTION

The previous chapters have discussed the data pertaining to the performance of the individual unit operations in the IAPS and I-HRAP systems. These various stages are, however, designed to effect different aspects of wastewater treatment, each contributing to the overall performance of the system. This chapter presents an overview of the combined process and includes operational data, from the first commissioning of the plant in 1996 until the present. Also presented is an overview of the IAPS as an alternative technology for domestic wastewater treatment, the I-HRAP as a free-standing tertiary treatment unit operation that might be attached to any wastewater treatment works, and an evaluation of effluent quality achieved in these systems compared with more conventional sewage treatment technology.

10.2 ORGANIC REMOVAL

Organic material entering a watercourse will act as a food source for the microorganisms present in the receiving water and will be metabolised in a series of oxidation reactions (Horan, 1996). The oxygen required for these reactions is obtained from dissolved oxygen in the water, with the consequent de-oxygenation of the water. At the same time, the water is re-oxygenated by transfer of oxygen between the surface of the water and the atmosphere. The rate at which this transfer occurs is dependant on depth, velocity, temperature and turbulence (Horan, 1996). In addition photosynthesising plants and algae may contribute to the dissolved oxygen levels. The difference between the de-aeration and subsequent re-aeration of a watercourse is known as the oxygen sag (Horan, 1996). The greater the organic load discharged, the more severe, in terms of duration and deficit, will be the oxygen sag.

A water body's prevailing oxygen concentration is one of the strongest selection pressures in determining the abundance and distribution of the aquatic community. According to Horan (1996) when water is polluted with an organic effluent, there is usually a fall in the number of species (decrease in diversity), a change in the type of species and a change in the number of individuals of each species. Water discharged with a high oxygen demand can, therefore, have a marked impact on the ecological health of the receiving watercourse. Although nutrient removal and pathogen reduction are becoming more important, the primary goal of wastewater treatment remains the removal and degradation of organic matter (Maier *et al*., 2000).

Figure 10.1 illustrates the CODt removal performance across the various units of the IAPS averaged over the period July 1996 to October 2004.

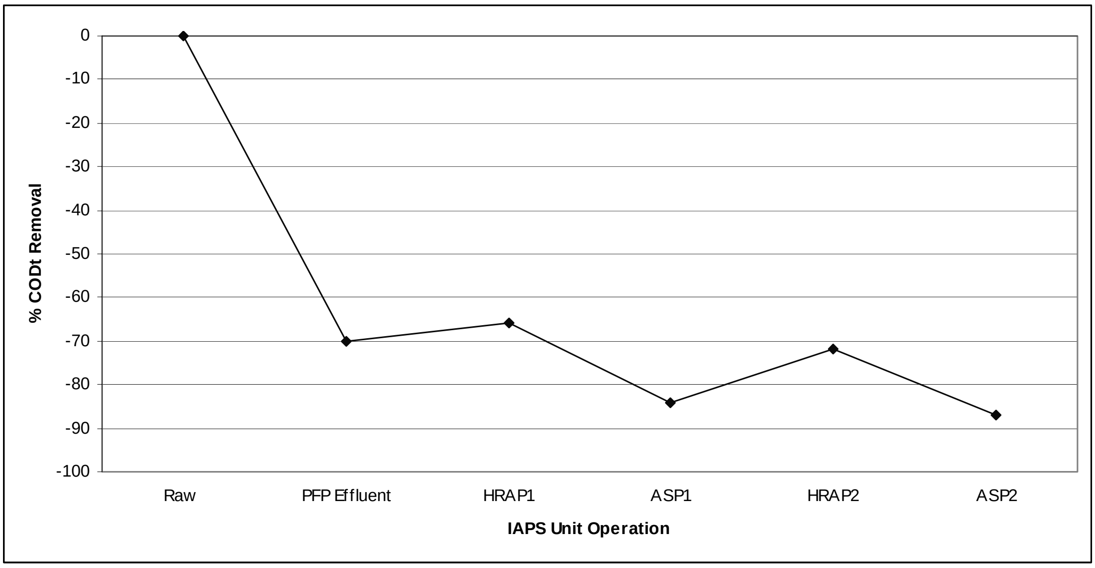

Figure 10.1: Total chemical oxygen demand removal through the integrated algal ponding system. The results depicted for the raw water, PFP effluent, HRAP1 and ASP1 reflect averages for HRAP treatment for the entire period from July 1996 to October 2004. HRAP2 and ASP2 were only brought online as the I-HRAP operation in July 2003.

As shown in Figure 10.1, the mean CODt removal rate through the IAPS over the 9 year operation period was 87%. This is comparable with conventional wastewater treatment processes such as activated sludge and trickle filters (Horan, 1996; Maier *et al*., 2000; Henze *et al*., 2002) as well as WSP (Bryant, 1986; Mara & Pearson, 1986; Soler *et al*., 1995; Racault *et al*., 1995). In a study of a stabilisation pond in Dar es Salaam, Kayombo *et al*. (2002) only found a 71% removal efficiency. Oswald (1991a) reports a slightly better performance of 93% at the AIWPS plant in St Helena, California. The CODt increases in HRAP1 and HRAP2 are due to the increase in algal biomass.

Although the demonstration system displayed effective COD removal, it was unable to consistently meet the South African discharge standard of 75 mg.ℓ^{-1} (DWAF, 2002). However, a large portion of this residual CODt is stabilised algal biomass, which would not contribute to oxygen depletion in the receiving water but, because it is photosynthetic, actually has the potential to increase DO levels (Mara & Pearson, 1986; Oswald, 1991a; Meiring & Oellermann, 1995). It has, in fact, been argued that algae can be beneficial to some receiving waters and in agricultural irrigation (Green *et al*., 1995a). Gloyna and Tischler (1979) maintain that discharge of algae cells in a properly treated effluent may increase productivity at higher trophic levels of aquatic organisms such as fish and certain invertebrate species. Oswald (1991a) also argues that algae may be beneficial to the food chain in the local ecosystem. Because of the advantages of the residual algae, it should not be necessary to remove it from the effluent unless a study of the receiving water reveals a specific reason for this requirement (Gloyna & Tischler, 1979).

In situations where authorities will not accept the low impact nature of the algae, and will not exempt algal pond effluent from strict COD discharge standards, e.g. American Environmental Protection Agency (EPA) (Benemann *et al*., 1980); it is possible to remove the residual algae by other means. Benemann *et al*. (1980)

successfully removed algae using chemical coagulation followed by sedimentation or dissolved air flotation (DAF) followed by rapid sand filtration. Due to their filamentous nature, some species of microalgae lend themselves to removal by vibrating, oscillating or cascade screens (Oswald, 1988b). Poelman *et al*. (1997) recovered up to 95% of microalgae by electrolytic flocculation. Microalgae can also be removed by microfiltration and centrifugation, but in large-scale operations these systems can present problems with rapid clogging and centrifuge size respectively (Oswald, 1988b). The drawback of any of these algal separation techniques is the cost and expertise required to implement such systems, which detracts from the original low-cost, low-tech concept of the IAPS.

10.3 NUTRIENT REMOVAL

Nitrogen control in treatment plants focuses on ensuring that nitrogen appears in the effluent in the desired form and concentration, and allowing for nitrification is often sufficient to alleviate ammonia toxicity and oxygen demand in receiving waters (Barnes & Bliss, 1983). Where more complete nitrogen removal is required, additional or alternative procedures need to be employed. The most commonly used method being the coupling of nitrification to denitrification processes. As these two reactions occur under different regimes, they must either be separated in a multi-stage system or in different zones within the same reactor (Horan, 1996). Plant types used for nitrification include trickling filters, rotating disc filters, activated sludge and two stage activated sludge, while denitrification takes place in systems such as anaerobic filter, anaerobic fluidised bed and combined sludge system with anoxic zones (Barnes & Bliss, 1983). Nitrogen removal may also take place in waste stabilisation ponds (Gloyna & Tischler, 1979; Mara & Pearson, 1986).

Figures 10.2 and 10.3 illustrate the cycling of ammonia and nitrate, respectively, through the IAPS. Due to ammonification and possibly nitrogen fixation, there is an increase in ammonia in the first HRAP. This is then effectively removed in HRAP2 by the probable mechanism of volatilisation and possibly some assimilation into the algal biomass.

The nitrate increase depicted in Figure 10.3 is most likely due to the decomposition and subsequent nitrification of organic nitrogen. The mechanism responsible for the decrease in nitrate in HRAP2 is unclear, as the high levels of oxygen present in this pond make denitrification unlikely. Removal is possibly due to assimilation into the algal biomass (Barnes & Bliss, 1983; Schumacher & Sekoulov, 2003). The mean nitrate in the effluent over the 9-year life of the IAPS is below the 15 $mg.\ell^{-1}$ DWAF discharge standard. The mean performance data does not, however, reflect the widely fluctuating effluent nitrate concentration, with periods of more than double this level recorded. According to Horan (1996), sewage effluent routinely contains nitrates of between 5 and 30 $mg.\ell^{-1}$. The variation in HRAP nitrate levels may be due to differences in algal productivity.

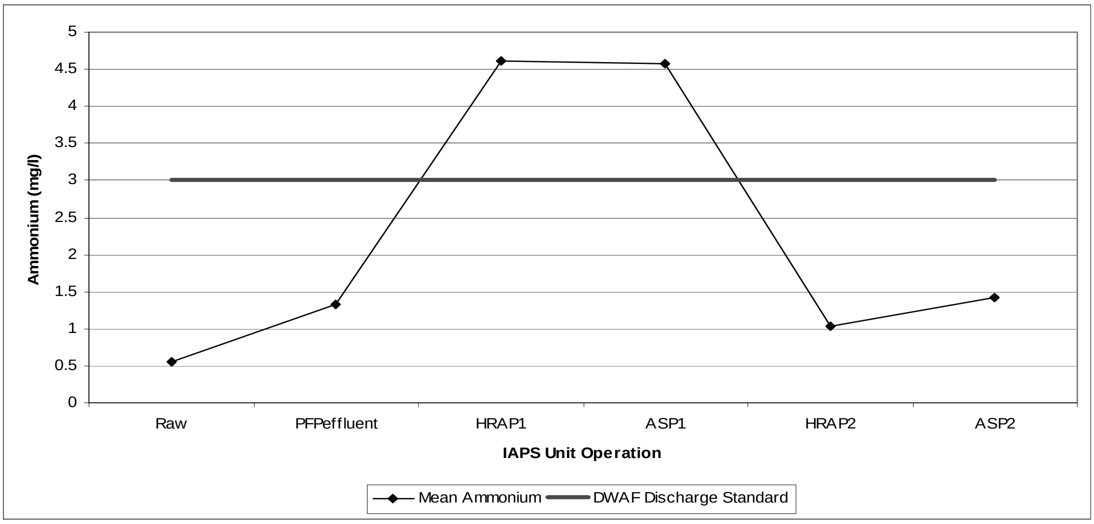

Figure 10.2: Average results for ammonia cycling through the IAPS, monitored over the period 1997-2004 and for the I-HRAP (HRAP2 and ASP2) from 2003-2004.

Maximising algal biomass might not provide the most efficient nutrient removal treatment because of the effects of light attenuation which effectively results in self-shading (Cromar *et al*., 1996). If the treated effluent is to be used for irrigation, as would be recommended in terms of sustainable water usage, the nitrate in the effluent would not be detrimental but would, in fact, be desirable for enhancing crop production. Where treated water is discharged to surface water bodies, further nitrate removal may be necessary via, for instance, a wetland system, which would be consistent with the sustainability concept of pond technology (Tanner & Sukias, 2003). Although nitrate levels in the effluent are not always below the standard, and are considerably higher than those measured by Green *et al*. (1996) in their high rate pond (3 mg.ℓ^{-1}), a mean TKN removal in the system of 55% was observed. Reported TKN removal rates in conventional WSP vary from 35 to 88% (Reed, 1985; Racault *et al*., 1995; Mendes *et al*., 1995; Sukias *et al*., 2003). The 55% removal rate achieved in this study was slightly better than the 46% obtained by Cromar et al. (1996) in a HRAP operated in Scotland although the temperate nature of this location resulted in a wide seasonal variance (0% in winter – 85% in summer).

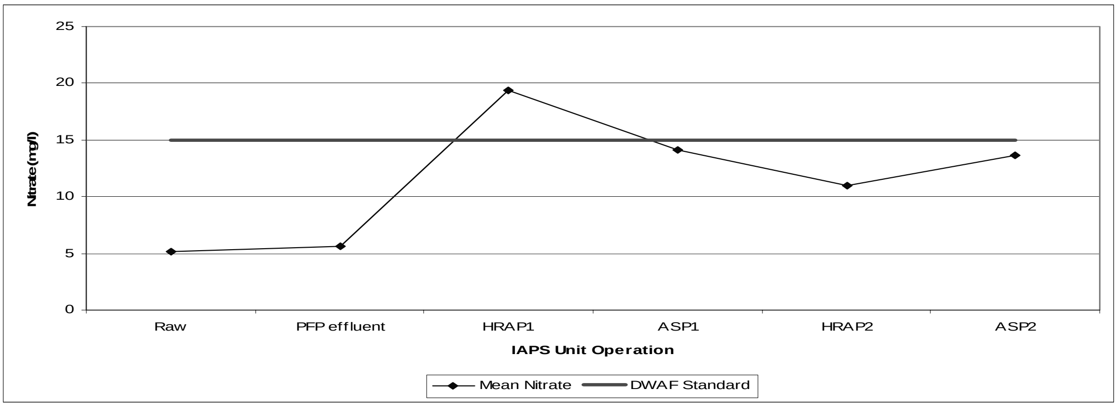

Figure 10.3: Average values for nitrate cycling through the IAPS, monitored over the period 1997-2004 and the I-HRAP (HRAP2 and ASP2) for 2003-2004.

The efficacy of phosphate removal in the IAPS is shown in Figure 10.4. The mean removal rate over the study period was 76%, with >85% removal occurring during 90% of operation. The mean concentration in the treated effluent was 5.4 mg.ℓ$^{-1}$, considerably lower than the South African discharge standard. Studies of WSP in Portugal and France revealed phosphate removal efficiencies of between 50 and 67% (Racault *et al*., 1995; Mendes *et al*., 1995). Constructed wetlands in Brazil and New Zealand, by comparison, reduced phosphate levels by between 5 and 46% (Tanner and Sukias, 2003; Sezerino *et al*., 2003). HRAP in Morocco had mean removal rates from 52 to 61% (El Hamouri *et al.*, 1994; El Hamouri *et al*., 1995). Nurdogan and Oswald (1995) were, however, able to achieve up to 99% phosphate removal with the addition of CaO.

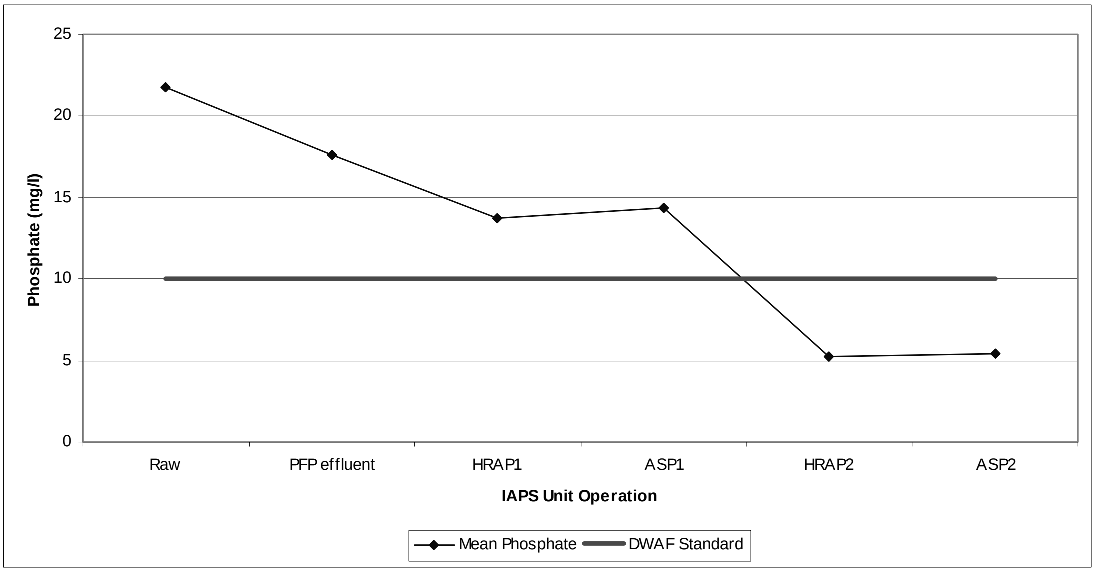

Figure 10.4: Average results for phosphate removal in the IAPS, monitored over the period 1997-2004. I-HRAP (HRAP2 and ASP2) averages for 2003-2004.

10.4 DISINFECTION

The removal of faecal pathogens in wastewater is an important, but often neglected, public health consideration in treatment systems. This is particularly true in developing countries, although there is also an increasing emergence of waterborne parasitic diseases (Stott *et al*., 2003). As has been suggested throughout this report, the IAPS technology has been researched at EBRU for application in smaller, rural communities of Southern Africa. In most cases these are water scarce areas where the reuse of treated effluent is an important consideration for sustainable management of water resources. In order to facilitate this reuse, however, it is imperative that treated water is adequately disinfected. Modern tertiary treatment methods such as chemical flocculation followed by filtration and then chlorine, UV or ozone disinfection are able to produce effluent with faecal coliform levels of <1 cfu.100 mℓ$^{-1}$ (Law, 1996) but are often too costly for smaller municipalities (Fujioka *et al*., 1999). The use of the I-HRAP as a polishing step for pathogen removal was thus investigated as part of this study.

Figure 10.5 illustrates the mean faecal indicator *E. coli* counts in the various ponds in the IAPS sequence. As can be seen from the figure, there is more than a 4 log reduction in *E. coli* in the system, with the final effluent having a count of <1000 cfu.100 $m\ell^{-1}$. These figures, however, represent the mean results over the total eighteen month monitoring period, including winter periods and experimental conditions which allowed insufficient hydraulic retention times. Under conditions of optimal HRT, i.e. 6 days in winter and 3 days in summer, a further 2 log reduction was achieved, with a final mean count of <10 cfu.100 $m\ell^{-1}$ (Figure 10.6). This equates to a 99.999% reduction. Zero *E. coli* were recorded in 78% of the samples tested.

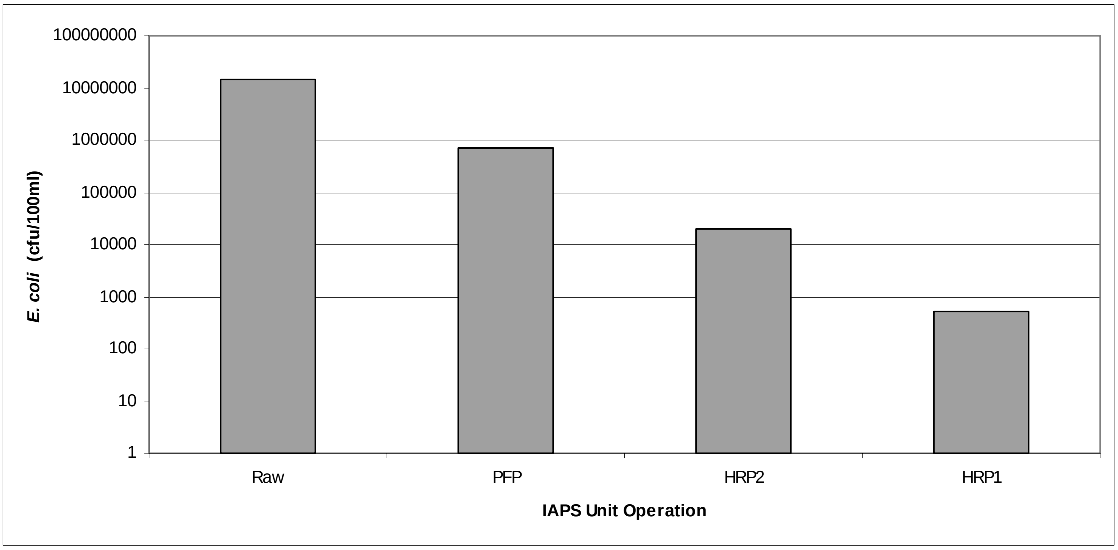

Figure 10.5: *E. coli* counts through the IAPS and I-HRAP sequence. This figure illustrates all data from the 2003-2004 monitoring period, i.e. including results from operation with sub-optimal hydraulic retention times.

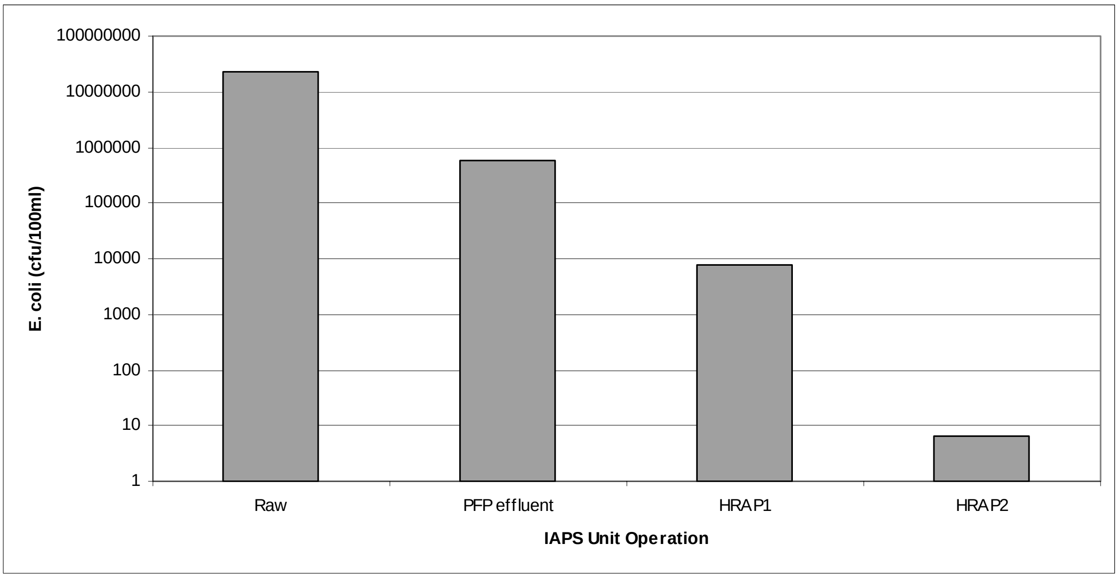

Figure 10.6: *E. coli* counts through the IAPS and I-HRAP sequence, illustrating results only from operation under optimal hydraulic retention times, both during 2003 and 2004.

In most instances, standard WSP are unable to reduce faecal coliforms to below 1000 cfu.100 $m\ell^{-1}$ (El Hamouri *et al*., 1994; Jagels & Lues, 1996; Almasi & Pescod, 1996;

Rangeby *et al*., 1996; Garcia & Bécares, 1997; Bahlaoui *et al*., 1997). Wetland systems have also shown potential for biological pathogen reduction but effluents generally contain faecal coliforms in excess of 1000 cfu.100 mℓ^{-1} (Arias *et al*., 2003; Ansola *et al*., 2003). Davies-Colley *et al*. (2003) achieved similar results to the Grahamstown I-HRAP system, using a HRAP followed by a maturation pond in New Zealand. Sebastian and Nair (1984) also reported total *E. coli* removal with a 2 day contact time at pH 11 in an experimental HRAP system operated in India.

10.5 CONCLUSIONS

- IAPS provides substantial organic removal, with a mean CODt reduction of almost 90% across the system.
- Effective ammonia and phosphate removal was achieved in the system, with final effluent levels of 1-1.5 mg.ℓ^{-1}, below the 3 mg.ℓ^{-1} South African discharge standard.
- While the IAPS was not able to achieve *E.*coli counts below 1000 cfu.100 mℓ^{-1}, with the installation of the I-HRAP operation it was possible to obtain an *E. coli* count of <cfu.100 mℓ^{-1} in the final treated effluent.
- While nitrate removal persists as a weak point of the IAPS and I-HRAP technology, the use of algal biomass as a carbon source in an appended denitrification unit operation may be considered where elevated nitrate levels may require this.

11. CONCLUSIONS: TECHNICAL AND SOCIAL IMPLICATIONS OF THE IAPS AND I-HRAP SYSTEMS DEVELOPMENT

11.1 INTRODUCTION

The current report is part of a widely based WRC programme, which commenced in 1990, to investigate the application of algal biotechnology and algal ponding systems technology in dealing with the problems of salinity and sanitation. These constitute six of the seven priority pollution issues facing the country and the development of locally appropriate treatment technologies to deal with these problems requires urgent attention (DEAT, 2000). Application studies have included tannery, abattoir, winery, hypersaline and acid mine drainage wastewaters.

The IAPS was identified as the appropriate core technology platform for developing the algal biotechnology applications such as low-cost systems appropriate for implementation in meeting environmental sustainability objectives. The AIWPS in the treatment of domestic wastewater was identified as one of the most intensively engineered of IAPS applications and selected as the model for the algal ponding development study to be implemented. In this regard the research and demonstration plant constructed at the EBRU Environmental Biotechnology Experimental Field Station in Grahamstown was based on the AIWPS design.

11.2 OPERATION AND PERFORMANCE OF THE AIWPS PLANT

The objectives of the IAPS research and demonstration plant in Grahamstown was to firstly evaluate the performance of the AIWPS design under South African conditions and then to undertake process development research required to extend the process to the specific problems of salinity and sanitation experienced in this country.

The plant was commissioned in 1996. The performance for the AIWPS configuration of the system was monitored for 5 years and produced the following main results:

Primary Facultative Pond

- Effective removal of the incoming sewage organic content, with CODt removal rates of over 70%.
- A valuable buffering capacity where peaks in CODt load of over 4 000 mg. 1^{-1} were absorbed in the PFP, while CODt in the effluent remained below 400 mg.1^{-1}.
- The very slow build up of sludge in the fermentation pit is a major benefit of the system as the need for frequent sludge handling, with the associated costs, typical of conventional treatment works, is eliminated.
- Good removal of total nitrogen was achieved in the PFP unit, although nitrate and ammonium levels were not affected.
- There was a poor phosphate reduction in the PFP unit operation.

High Rate Algal Ponds

- CODt increased in the HRAP due to the growth of algae biomass, indicated by the similarity in soluble HRP COD and settled effluent COD, from the ASP.
- Algal biomass was effectively removed in the ASP, giving a net CODt reduction during the HRAP/ASP stage of 43%.
- Although residual COD was mainly in the stabilised form of algal material, the full IAPS system (HRAP operated as a single stage unit operation) did not achieve the 75 mg.ℓ^{-1} discharge standard.
- While an average 26% phosphate reduction was observed in the HRAP, this was not sufficient to bring effluent levels to within the 10 mg.1^{-1} required by the discharge standards.
- Despite good ammonia removal, residual levels also at times exceeded the 3 mg.ℓ^{-1} ammonia discharge standard.
- Nitrate removals were somewhat erratic, with the levels of these nutrients increasing at times.
- Algal genera found in the PFP were similar to those recorded in conventional WSP. The operation of the HRAP results in a lower species diversity, dominated by strongly floc-forming green algal forms which provide a good settling characteristic.

11.3 DEVELOPMENT OF THE I-HRAP IN TERTIARY TREATMENT UNIT OPERATION

While the AIWPS design performed well and delivered a final wastewater superior to most ponding systems operated in South Africa, it was nevertheless evident that, as operated in Grahamstown, the system would be unlikely to meet the DWAF discharge standards for nutrient removal with any consistency. With this in mind the I-HRAP development was undertaken in which the use of the HRAP was investigated as a free-standing unit operation in tertiary treatment that could be used as an add-on to the AIWPS or any other sub-optimally performing water treatment works.

The principal results of the I-HRAP study were as follows:

- IAPS provides substantial organic removal, with a mean CODt reduction of almost 90% across the system.
- Effective ammonia and phosphate removal was achieved in the system, with final effluent levels of 1-1.5 mg.ℓ^{-1} - below the 3 mg.ℓ^{-1} South African discharge standard.
- While the IAPS was not able to achieve *E.coli* counts below 1000 cfu.100 mℓ^{-1}, with the installation of the I-HRAP operation it was possible to obtain an *E.coli* count of <1 cfu.100 mℓ^{-1} in the final treated effluent.
- While nitrate removal persists as a weak point of the IAPS and I-HRAP technology, the use of algal biomass as a carbon source in an appended denitrification unit operation may be considered where elevated nitrate levels may require this.

11.4 TECHNOLOGY TRANSFER

Having undertaken a number of technology transfer functions of IAPS technology in tannery, abattoir, hypersaline and acid mine drainage wastewater applications, the question arose of how widespread the potential application of the system, and particularly of the I-HRAP, would be in domestic sewage treatment in the rural areas of the Eastern Cape Province. This study shows that 12% of the 98 WSP and other small plants treating less than 1 Mℓ.day in this province were discharging a water which did not meet the DWAF discharge standards.

It is against this background that the I-HRAP may provide a useful tertiary treatment operation as a low-cost add-on unit operation to enable compliance in these treatment works.

11.5 SOCIAL IMPLICATIONS

The algae that is settled and separated in the algae settling ponds is a beneficial by-product of the HRAP treatment system and has a number of potential uses other than as a potential carbon source for denitrification as investigated in this study. As it is rich in nutrients and plant hormones, the most obvious use would be as a fertilizer (Benemann *et al*., 1980)

Horan and Horan (2004) have undertaken follow-up WRC Project K5/1619, ‘ IAPS Algal Biomass and Treated Effluent Utilisation as a Key Strategy in Sustainable and Low-cost Sanitation’ in order to investigate this potential. For IAPS algal-supplemented trial plantings they have found turnip yields of 1.4 times greater, by mass, compared with crops grown using commercial fertiliser (2:3:2, N:P:K) and 8.7 times those in unfertilised plots. Plots treated with algae and fertiliser yielded turnips with a mean weight 12.6 times that of the control. Similarly, they cultivated Swiss chard at 15.4 $t.ha^{-1}$ in soil enriched with HRAP algae, whilst commercial fertiliser only yielded 10.5 $t.ha^{-1}$ and unfertilised land, 3.2 $t.ha^{-1}$. A combination of algae and fertiliser once again had the greatest yield at 18.5 $t.ha^{-1}$ (Horan & Horan, 2004).

Another potential use of the algae is as a dietary protein feed supplement for animal nutrition including pigs, poultry and cattle (McGarry & Tongkasame, 1970). In Thailand the production of *Tilapia mosambique* was proved feasible with the use of algae-containing pond effluent (McGarry & Tongkasame, 1970). Nutritional analyses of the HRAP algal biomass revealed an approximate composition (protein 41.5%, lipid 4.8%, carbohydrate 35.1%) similar to that of other feed supplements such as soya oil cake meal and sorghum gluten meal (Potts, 1998). Potts (1998) was able to include this algae in formulated diets at protein levels of up to 20% to productively grow ornamental fish (family: Poeciliidae) in an experimental system. A further potential use of wastewater grown algae is in energy generation via their fermentation to methane (Oswald, 1988c).

The potential of linking water treatment and social activity, including job creation, through the recovery and re-use of treated waters has been the subject of WRC Project K5/1456/Part 4 “The Biotechnology of Saline and Sewage Wastewater Co-treatment”

(Rose *et al.*, 2005). This study investigated the application of treated acid mine drainage wastewaters in urban agricultural programmes.

The following model is proposed for the application of the IAPS and particularly I-HRAP technology in linking water treatment and job creation initiatives which are dependent on the ability of the system to produce a water quality that at least meets DWAF irrigation water discharge standards. The development of this model is dealt with in greater detail in WRC Project K5/1619 "IAPS Algal Biomass and Treated Effluent Utilisation as a Key Strategy in Sustainable and Low Cost Sanitation."

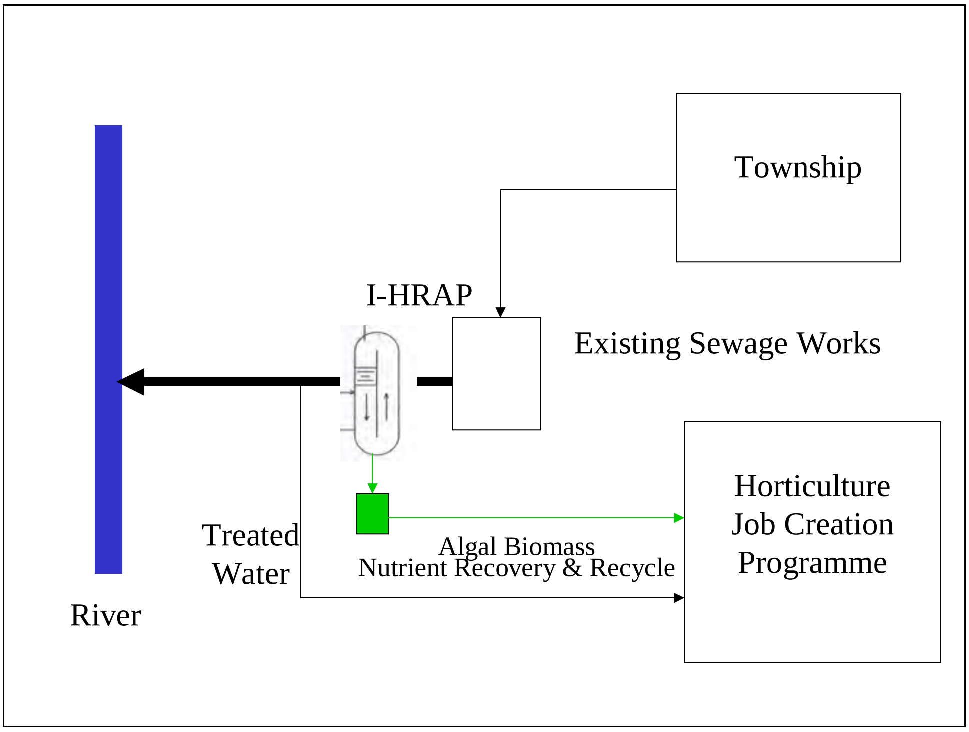

Figure 11.1: A schematic outline of the application of the I-HRAP as a retrofit to an existing poorly performing sewage works. Tertiary treatment, including disinfection, would enable the recovery and re-use of the water resource, and algal biomass as fertiliser, in community gardening or urban agriculture job creation projects.

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APPENDICES

APPENDIX 1

WRC STUDY ‘SALINITY SANITATION AND SUSTAINABILITY’ - PROJECT REPORTS

The WRC study which has been summarised here developed out of a number of closely interrelated studies, undertaken for the WRC by the Rhodes University Environmental Biotechnology Group, over a 10 year period. The detailed findings associated with this work will be published separately as individual project reports. The following lists the WRC reports which cover the various investigations dealt with in the programme. The individual WRC projects under which the various studies were undertaken are listed separately below:

Report 1

Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.
Volume 1. Overview

Report 2

Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.
Volume 2. Integrated Algal Ponding Systems and the Treatment of Saline Wastewaters.
Part1: Meso-saline Wastewaters - The *Spirulina* Model.

(Project K5/495: A Biotechnological approach to the removal of organics from saline effluents - Part 1.)

Report 3

Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.
Volume 2. Integrated Algal Ponding Systems and the Treatment of Saline Organic Wastewaters.
Part 2: Hyper-saline Wastewaters - The *Dunaliella* Model.

(Project K5/495: A biotechnological approach to the removal of organics from saline effluents - Part 2.)

Report 4

Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.
Volume 3. Integrated Algal Ponding Systems and the Treatment of Domestic and Industrial Wastewaters. Part1: The AIWPS Model.

(Project K5/651: Appropriate low-cost sewage treatment using the integrated algal high rate oxidation ponding process.)

Report 5

Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.
Volume 3. Integrated Algal Ponding Systems and the Treatment of Domestic and Industrial Wastewaters. Part 2: Abattoir Wastewaters.

(Project K5/658: Algal high rate oxidation ponding for the treatment of abattoir effluents.)

Report 6

Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.
Volume 3. Integrated Algal Ponding Systems and the Treatment of Domestic and Industrial Wastewaters.
Part 3: Mine Drainage Wastewaters - The ASPAM Model.

(Project K5/656: Appropriate low-cost treatment of sewage reticulated in saline water using the algal high rate oxidation ponding system.)

Report 7

Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.
Volume 3. Integrated Algal Ponding Systems and the Treatment of Domestic and Industrial Wastewaters.
Part 4: System Performance and Tertiary Treatment Operations.

(Project K5/799: Development and monitoring of integrated algal high rate oxidation pond technology for low-cost treatment of sewage and industrial effluents;
Project K5/1073: Extension of applications and optimisation of operational performance of algal integrated ponding systems technology in appropriate low-cost treatment of industrial and domestic wastewaters.
Project K5/1362: Development and technology transfer of IAPS applications in upgrading water quality for small wastewater and drinking water treatment systems.)

Report 8

Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.
Volume 3. Integrated Algal Ponding Systems and the Treatment of Domestic and Industrial Wastewaters.
Part 5: Winery and Distillery Wastewaters.

(Project K5/1073: Extension of applications and optimisation of operational performance of algal integrated ponding systems technology in appropriate low-cost treatment of industrial and domestic wastewaters.)

Report 9

Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.
Volume 4. The Rhodes BioSURE Process®.
Part 1: Biodesalination of Mine Drainage Wastewaters.

(Project K5/869: Biological sulphate desalination and heavy metal precipitation in industrial and mining effluents using the IAPS.)

Report 10

Salinity, Sanitation and Sustainability: A Study in Environmental

Biotechnology and Integrated Wastewater Beneficiation in South Africa.

Volume 4. The Rhodes BioSURE Process®.
Part 2: Enhanced Hydrolysis of Organic Carbon Substrates - Development of the Recycling Sludge Bed Reactor.

(Project K5/972: Process development and system optimisation of the integrated algal trench reactor process for sulphate biodesalination and heavy metal precipitation in mining and industrial effluents.)

Report 11

Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.
Volume 4. The Rhodes BioSURE Process®.
Part 3: Sulphur Production and Metal Removal Unit Operations.

(Project K5/1078: Development and piloting of the integrated biodesalination process for sulphate and heavy metal removal from mine drainage water incorporating co-disposal of industrial and domestic effluents;
Project K5/1336: Scale-UP development of the Rhodes BioSURE Process® for sewage sludge solubilisation and disposal.)

Report 12

Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.

Volume 4. The Rhodes BioSURE Process®.
Part 4: Treatment and Disposal of Sewage Sludges:
(Project K5/1169: Intermediate scale-up evaluation of the Rhodes Process for hydrolysis and solubilisation of sewage sludges in a sulphate reducing bacterial system.)

PROJECTS

The following lists the WRC Projects the findings of which have been detailed in the reports as outlined above:

Project K5/410

A Biotechnological approach to the removal of organics from saline effluents.

Report: 1. Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.
Volume 1. Overview.

Project K5/495

A Biotechnological approach to the removal of organics from saline effluents.

Report: 2. Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.
Volume 2. Integrated Algal Ponding Systems and the Treatment of Saline Wastewaters. Part1: Meso-saline Wastewaters - The *Spirulina* Model.

Report: 3. Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.
Volume 2. Integrated Algal Ponding Systems and the Treatment of Saline Organic Wastewaters. Part 2: Hyper-saline Wastewaters - The *Dunaliella* Model.

Project K5/651

Appropriate low-cost sewage treatment using the integrated algal high rate oxidation ponding process.

Report 4: Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.
Volume 3. Integrated Algal Ponding Systems and the Treatment of Domestic and Industrial Wastewaters. Part1: The AIWPS Model.

Project K5/656

Appropriate low-cost treatment of sewage reticulated in saline water using the algal high rate oxidation ponding system**.**

Report 6: Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.
Volume 3. Integrated Algal Ponding Systems and the Treatment of Domestic and Industrial Wastewaters. Part 3: Mine Drainage Wastewaters - The ASPAM Model.

Project K5/658

Algal high rate oxidation ponding for the treatment of abattoir effluents.

Report 5: Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.
Volume 3. Integrated Algal Ponding Systems and the Treatment of Domestic and Industrial Wastewaters. Part 2: Abattoir Wastewaters.

Project K5/799

Development and monitoring of integrated algal high rate oxidation pond technology for low-cost treatment of sewage and industrial effluents

Report 7: Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.
Volume 3. Integrated Algal Ponding Systems and the Treatment of Domestic and Industrial Wastewaters. Part 4: System Performance and Tertiary Treatment Operations.

Project K5/869

Biological sulphate desalination and heavy metal precipitation in industrial and mining effluents using the IAPS.

Report 9: Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.
Volume 4. The Rhodes BioSURE Process®. Part 1: Biodesalination of Mine Drainage Wastewaters.

Project K5/972

Process development and system optimisation of the integrated algal trench reactor process for sulphate biodesalination and heavy metal precipitation in mining and industrial effluents.

Report 10: Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.
Volume 4. The Rhodes BioSURE Process®. Part 2: Enhanced Hydrolysis of Organic Carbon Substrates - Development of the Recycling Sludge Bed Reactor.

Project K5/1073

Extension of applications and optimisation of operational performance of algal integrated ponding systems technology in appropriate low-cost treatment of industrial and domestic wastewaters.

Report 7: Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.
Volume 3. Integrated Algal Ponding Systems and the Treatment of Domestic and Industrial Wastewaters. Part 4: System Performance and Tertiary Treatment Operations.

Report 8: Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.
Volume 3. Integrated Algal Ponding Systems and the Treatment of Domestic and Industrial Wastewaters. Part 5: Winery and Distillery Wastewaters

Project K5/1078

Development and piloting of the integrated biodesalination process for sulphate and heavy metal removal from mine drainage water incorporating co-disposal of industrial and domestic effluents.

Report 11: Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.
Volume 4. The Rhodes BioSURE Process®. Part 3: Sulphur Production and Metal Removal Unit Operations.

Project K5/1169

Intermediate scale-up evaluation of the Rhodes Process for hydrolysis and solubilisation of sewage sludges in a sulphate reducing bacterial system.

Report 12: Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.
Volume 4. The Rhodes BioSURE Process®. Part 4: Treatment and Disposal of Sewage Sludges.

Project K5/1336

Scale-up development of the Rhodes BioSURE Process® for sewage sludge solubilisation and disposal.

Report 11: Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.
Volume 4. The Rhodes BioSURE Process®. Part 3: Sulphur Production and Metal Removal Unit Operations.

Project K5/1362

Development and technology transfer of IAPS applications in upgrading water quality for small wastewater and drinking water treatment systems.

Report 7: Salinity, Sanitation and Sustainability: A Study in Environmental Biotechnology and Integrated Wastewater Beneficiation in South Africa.
Volume 3. Integrated Algal Ponding Systems and the Treatment of Domestic and Industrial Wastewaters. Part 4: System Performance and Tertiary Treatment Operations.

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APPENDIX 2.

RESEARCH PRODUCTS

2.1. STUDENTS TRAINED

2.1.1 Post-Doctoral Fellows

Dr. O. Shipin 1992 – 1996. Integrated Algal Ponding Systems.

2.1.2 PhD Students

L. Dekker (2003) Integrated algal ponding systems and the treatment of wine processing wastewaters.

2.1.3 MSc Students

S. Clark (2002) The independent high rate algal pond as a unit operation in tertiary wastewater treatment.

A. Neba (2004) The Independent High Rate Algal Pond (HRAP) integrating Biological Nitrogen Removal as a unit operation in Tertiary Wastewater Treatment

C. Wells (2005) Tertiary treatment in Integrated Algal Ponding Systems.

2.2 PUBLICATIONS

2.2.1 Papers

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APPENDIX 3

TECHNOLOGY TRANSFER ACTIONS

3.1 OFFICIAL OPENING OF THE AIWPS PLANT

The AIWPS Plant and the IAPS demonstration and research facility constructed at the Rhodes University Environmental Biotechnology Experimental Field Station, Grahamstown, were officially opened by the Minister of Water Affairs and Forestry, Prof Kader Asmal, on 18 April 1997. The event was attended by some 300 local people, engineers, scientists and senior government officials. See Fig A1

Figure A1: Hon. Minister of Water Affairs and Forestry, Prof Kader Asmal handing over the keys of the plant to the Mayor of Grahamstown Cnlr. Mpahlwa. Background from left to right: Dr D Woods, Vice Chancellor Rhodes University; Dr O Hart and Prof P Rose, Rhodes University EBG.

3.2 WISA TECHNICAL TOUR

A Technical Tour to the Environmental Biotechnology Field Station and the IAPS Plant took place during the Port Elizabeth WISA Conference in 1996. Approximately 150 visitors attended including engineers, scientists, local government and DWAF officials. See Figure A2

3.3 SITE VISITS

The IAPS Plant has been visited by over 5000 people since its opening, including scientists, engineers, municipal officials, students, scholars and the general public. It has attracted particular attention at the Grahamstown Scifest, and is regularly used for teaching students in environmental biotechnology, industrial microbiology, applied biochemistry and environmental economics.

Figure A2: Technical visit to the Environmental Biotechnology Field Station during the Mine Water Conference, January 2000.

3.4 WRC TECHNICAL TOUR

A technical tour of inspection of developments in the WRC study relating to IAPS Technology in South Africa was undertaken 24 – 26 January, 1996 (Figure A3). Members of the Technical Committee included: Prof C. T Johnson (Chairman WRC); Mr P.E. Odendaal (Executive Director WRC); Mr D.S. van der Merwe (Deputy Executive Director WRC); Mr Z. Ngcakani (Research Manager WRC); Mr J.R. Muller (Abakor); Dr A. Jarvis (Sasol Ltd); Dr O.O. Hart (Rhodes University); Prof P.D. Rose (Rhodes University).

Figure A3: Technical tour party which undertook the inspection of WRC AIPS project installations. From left to right Prof P.D. Rose (Rhodes University); Mr D. S. van der Merwe (Deputy Executive Director WRC); Mr Z. Ngcakani (Research Manager WRC); Prof C. T. Johnson (Chairman WRC); Mr J. R. Muller (Abakor); (Pilot); Dr O. O. Hart (Rhodes University); Dr A. Jarvis (Sasol Co.); Mr P. E. Odendaal (Executive Director WRC).

Sites visited included:

1. SASOL β-carotene production technical scale plant in Upington.
2. WRC/Mossop Western Leathers commercial scale IAPS plant in Wellington.
3. WRC demonstration plant IAPS sewage treatment in Grahamstown
4. WRC/Abakor demonstration IAPS plant at Cato Ridge Abattoir.
5. De Beers pilot plant for treatment of diamond wastes at De Beers Diamond Research Laboratory, Johannesburg.

APPENDIX 4

APPLICATIONS OF INTEGRATED ALGAL PONDING SYSTEMS TECHNOLOGY

A primary goal of the AIWPS technology transfer exercise was that in addition to demonstrating the technology in sewage treatment in South Africa, inputs would be made in the development of IAPS as a 'core technology' in an integrated beneficiation approach to saline and sanitation wastewater treatment. Applications of the technology were studied in the treatment of a number of industrial wastewater types. These technology development studies were undertaken in WRC projects noted below.

4.1 IAPS IN TANNERY WASTEWATER TREATMENT

Following research on the performance and operation of the tannery WSP in Wellington, and piloting of IAPS process development at Mossop Western Leathers Co., Wellington, the full-scale implementation of the IAPS process was undertaken. This involved construction of the full-scale 2 500 m^2 *Spirulina*-HRAP (Figure A4), and incorporated the retrofitting of ponding units in the established WSP system. These studies are detailed in WRC Report 'Integrated Algal Ponding Systems and the Treatment of Saline Wastewaters. Part 1: Meso-saline Wastewaters – The *Spirulina* Model'.

The industrial-scale IAPS plant treating tannery wastewaters was officially opened by the Hon. Minister of Water Affairs and Forestry, Prof Kader Asmal, on 28 November, 1997. The event was attended by about 250 local people, engineers, scientists and senior government officials.

Figure A4: Paddle wheel of the High Rate Algal Pond at Mossop-Western Leathers Co., in Wellington

4.2 THE *DUNALIELLA*-HRAP AND β-CAROTENE PRODUCTION

The treatment of hyper-saline wastewaters, utilising the halophilic micro-alga *Dunaliella salina*, and linkage to β-carotene recovery as a value-added by-product of treatment, was investigated in the hyper-saline compartments of the tannery IAPS. Studies on the optimisation of β-carotene production by *D.salina* led to the development and patenting of the Dual Stage Process. Research was partly funded by Sasol Ltd., who also undertook the industrial scale-up development of the process at Sastech in Sasolburg, and technical-scale production studies in Upington. Full-scale commercialisation has been developed in Upington.

Development of a *Dunaliella*-based HRAP (D-HRAP) for the treatment of hyper-saline wastewaters was scaled-up and evaluated in the treatment of organic contamination in saline carbonate brines at the Botswana Ash Co. soda ash production facility at Sua Pan, Botswana (Figure A 5)

Figure A5: The *Dunaliella*-HRAP pilot plant treating saline carbonate brines at Botswana Ash Co. Sua Pan, Botswana

4.3 TREATMENT OF ABATTOIR WASTEWATERS

A trend in South Africa away from centralised controlled slaughtering to small rural abattoirs has required a low-cost response to treatment of these wastewaters to deal with the diffusion of the water pollution problems. The abattoir also presents a case study for the application of IAPS as an upgradeable ‘core technology’, in the sustainable development context. Here the initial investment by a community in sewage treatment technology should be upgradeable as its economic development unfolds. In generating a high-strength agro-industrial wastewater the abattoir provides a practical example to evaluate the flexibility of the 'core technology' investment.

Following laboratory studies on the IAPS application in abattoir wastewater treatment, a pilot plant was construction on-site at the Cato Ridge Abattoir in Kwa Zulu-Natal (Figure A 6). The results of this study are the subject of WRC Report 'Integrated Algal Ponding Systems and the Treatment of Domestic and Industrial Wastewaters. Part 2: Abattoir Wastewaters'.

Figure A6: The HRAP unit operation of the IAPS plant constructed at the Cato Ridge Abattoir

4.4 TREATMENT OF WINERY AND DISTILLERY WASTEWATERS

The investigation of high organic load saline wastewater treatment using IAPS technology was extended in studies on wine lees and distillery wastewaters. The saline IAPS was evaluated as an alternative to the existing practice of disposal to land irrigation. Final disposal via evaporation ponds, and associated micro-algal production, was investigated as a basis for both environmental and social sustainability in this agro-industrial application in the rural economy. In addition to a further evaluation of the upgradeability of IAPS as a 'core technology', a specific focus of this programme involved an evaluation of the beneficiation potential in transforming these agriculturally-derived wastewaters into a resource, with downstream production of algal bioproducts providing the basis for an 'integrated wastewater resource management' approach to the problem.

These studies commenced in the EBG laboratories, and involved the use of the anaerobic baffle reactor as an initial unit pre-treatment operation to reduce the organic load fed to the IAPS. Findings were then subjected to scale-up pilot study at the Brennokem (Pty) Ltd wine lees plant in Worcester, South Africa (Figure A7).

The results of this study are detailed in WRC report 'Integrated Algal Ponding Systems and the Treatment of Domestic and Industrial Wastewaters. Part 5: Winery and Distillery Wastewaters'.

Figure A7: Pilot plant at Brennokem Co., in Worcester, South Africa, evaluating the IAPS in the treatment of wine lees and distillery wastewaters.

4.5 THE ASPAM PROCESS

The investigation of enhanced hydrolysis of complex organic substrates present in tannery effluents, and their use as carbon and electron donor sources, supporting high rates of sulphate reduction, provided an indication that ponding systems might themselves be used as bioreactors for the biological treatment of large-volume AMD flows. While WSP technology has been developed over the past 40 years for a wide range of wastewater treatment applications little attention, if any, has focussed on the use of these systems for AMD remediation.

This application of IAPS was investigated in WRC Project K5/869: 'Biological sulphate desalination and heavy metal precipitation in industrial and mining effluents using the IAPS', and the use of tannery effluent and sewage sludges as effective electron donors in sulphate-salinity reduction applications was demonstrated. These studies resulted in the conceptual development of the Algal Sulphate Reducing Ponding Process for Acid Metal

Wastewater Treatment (ASPAM) and are detailed in WRC report 'Integrated Algal Ponding Systems and the Treatment of Domestic and Industrial Wastewaters. Part 3: Mine Drainage Wastewaters – The ASPAM Model'.

4.6 THE RHODES BIOSURE PROCESS®

Fundamental studies were undertaken to explain the enhanced hydrolysis of organic particulate solids and sludges in the sulphate reducing compartments of the IAPS treating high-sulphate wastewaters. Application of these findings in the treatment of AMD as optimised reactions outside the IAPS environment, and utilising sewage sludges as the carbon source, resulted in the development of the Recycling Sludge Bed Reactor (RSBR) and the Rhodes BioSURE Process®. The linkage of saline and sanitation wastewater treatment would provide a sustainable management for the AMD problem for the long periods of time over which the decanting mine waters are expected to flow. The I-HRAP was used for the final treatment and polishing of the AMD process wastewaters.

The process was scaled up and evaluated together with ERWAT in a pilot plant at Grootvlei Mine near Springs (Figure A8). These studies are detailed in WRC report 'the Rhodes BioSURE Process®. Part 1: Biodesalination of Mine Drainage Wastewaters'.

Figure A8: Rhodes BioSURE Process® pilot plant constructed at the ERWAT Ancor Works, Springs