Tertiary Treatment in

Integrated Algal Ponding Systems

THESIS

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Charles Digby Wells

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Abstract

Inadequate sanitation is one of the leading causes of water pollution and consequently illness in many underdeveloped countries, including South Africa and, specifically, the Eastern Cape Province, where cholera has become endemic. As modern wastewater treatment processes are often energy intensive and expensive, they are not suitable for use in these areas. There is thus a need to develop more sustainable wastewater treatment technologies for application in smaller communities. The integrated algal ponding system (IAPS) was identified as a possible solution to this wastewater management problem and was investigated for adaptation to local conditions, at the Rhodes University Environmental Experimental Field Station in Grahamstown, South Africa.

The system was monitored over a period of nine years, with various configuration adjustments of the high rate algal pond (HRAP) unit operation investigated. Under standard operating conditions, the system was able to achieve levels of nutrient and organic removal comparable with conventional wastewater treatment works.

The mean nitrate level achieved in the effluent was below the $15mg.l^{-1}$ South African discharge standard, however, nitrate removal in the IAPS was found to be inconsistent. Although the system was unable to sustain chemical oxygen demand (COD) removal to below the $75mg.l^{-1}$ South African discharge standard, a removal rate of 87% was recorded, with the residual COD remaining in the form of algal biomass.

Previous studies in the Eastern Cape Province have shown that few small wastewater treatment works produce effluent that meets the microbial count specification. Therefore, in addition to the collation of IAPS data from the entire nine year monitoring period, this study also investigated the use of the HRAP as an independent unit operation for disinfection of effluent from small sewage plants. It was demonstrated that the independent high rate algal pond (IHRAP) as a free standing unit operation could consistently produce water with *Escherichia coli* counts of $0cfu.100ml^{-1}$. The observed

effect was related to a number of possible conditions prevailing in the system, including elevated pH, sunlight and dissolved oxygen.

It was also found that the IHRAP greatly enhanced the nutrient removal capabilities of the conventional IAPS, making it possible to reliably and consistently maintain phosphate and ammonium levels in the final effluent to below $5mg.l^{-1}$ and $2mg.l^{-1}$ respectively (South African discharge standards are $10mg.l^{-1}$ and $3mg.l^{-1}$ in each case).

The quality of the final effluent produced by the optimisation of the IAPS would allow it to be used for irrigation, thereby providing an alternative water source in water stressed areas. The system also proved to be exceptionally robust and data collected during periods of intensive and low management regimes were broadly comparable.

Results of the 9 year study have demonstrated reliable performance of the IAPS and its use an appropriate, sustainable wastewater treatment option for small communities.

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List of Abbreviations

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
BOD	Biological Oxygen Demand
BODult	Ultimate Biological Oxygen Deamnd
cfu	Colony Forming Units
COD	Chemical Oxygen Demand
CODs	Soluble Chemical Oxygen Demand
CODt	Total Chemical Oxygen Demand
DAF	Dissolved Air Flotation
DO	Dissolved Oxygen
DWAF	Department of Water Affairs and Forestry
EBG	Environmental Biotechnology Group
EBPR	Enhanced Biological Phosphate Removal
EBRU	Environmental Biotechnology Research Unit (Rhodes University)
EPA	United States Environmental Protection Agency
F/M	Food to Microorganism Ratio
GDW	Grahamstown Disposal Works
HRAP	High Rate Algal Pond
HRT	Hydraulic Retention Time
IAPS	Integrated Algal Ponding System
IHRAP	Independent High Rate Algal Pond
IWRM	Integrated Water Resource Management
PAO	Phosphorous Accumulating Organisms
PE	Person Equivalents
PFP	Primary Facultative Pond
RO	Reverse Osmosis

RSDN	Rural Services Development Network
TDS	Total Dissolved Solids
TKN	Total Kjeldal Nitrogen
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

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CHAPTER ONE

Domestic Wastewater Treatment

1.1 The Impacts of Domestic Wastewater

Covered sewers appear to have been built in the cities of the Indus Valley some 5 000 years ago and by 2 000 years ago, sewers were widely used by the Romans for human waste collection (Cunningham & Saigo, 1995). Most European cities only began using sewers in the middle of the 19^{th} century (Asano & Levine, 1996; Matos & Bertrand-Krajewski, 2004). These early sewers provided sanitation for households and, while dramatically improving living conditions, only removed the problem from the cities to the nearest lake, river or sea, where the wastewater was discharged. Whilst communities were still relatively small, the dilution in these water bodies provided satisfactory routes of disposal. However, as populations grew and industries began to contribute to the wastewater, the load on these rivers began to cause serious degradation of the water quality. Oxygen depletion and toxic pollutants threatened wildlife and made receiving rivers uninhabitable for fish and other aquatic biota. However, domestic wastewater control and treatment only really began once it was realised that these wastes were also linked to diseases such as cholera and typhoid (Oswald, 1994). Modern treatment processes, therefore, were only introduced on a large scale towards the end of the 19^{th} century (Maier *et al.,* 2000).

In evaluating the environmental impacts of wastewater, Metcalf and Eddy (2003) describe the principal constituents of concern. These include suspended solids and grit, which contribute to both organic and inorganic pollution. The organic portion may lead to the development of sludge deposits and anaerobic conditions. Both will increase turbidity and thereby reduce the amount of sunlight reaching aquatic plants. Domestic wastewater will also contain biodegradable organic wastes of plant and animal origin, composed mainly of carbohydrates, proteins and fats. The biological stabilisation of these substances uses up natural oxygen resources, resulting in septic conditions. Pathogens are often present in very large numbers and can be transmitted to humans and animals through water contact, spreading diseases. Nutrients such as nitrogen and phosphorous can stimulate excessive, often unwanted, aquatic plant growth. This may exclude other life forms as well as affecting aesthetics. Inorganic or organic toxic substances are classified as such due to their mutagenicity, carcinogenicity, teratogenicity or high acute toxicity and are also found in wastewater. Refractory organics such as surfactants, phenols and agricultural pesticides can also be present in wastewater streams and often resist conventional treatment methods. These may be toxic to humans, birds, fish and aquatic flora. There is increasing pressure on commercial and/or industrial water users to treat their waste on site before discharging any effluent. This, however, is not always the case and these activities may be a source of heavy metals, which can also cause acute and chronic health problems. Dissolved inorganics such as calcium, sodium and sulphate often concentrate in wastewater and may need to be removed if the water is to be reused. Even some industrial reuse of water is very sensitive to high dissolved solids. Wastewater is sometimes warmer than the receiving aquatic environment. This may change the species composition and diversity as well as reducing the capacity of water to absorb oxygen.

Despite the potential for domestic wastewater to contaminate the receiving environment, approximately 90% of sewage in cities in developing countries is discharged untreated into water bodies (Senzia *et al.,* 2003). According to Mara and Feachem (2001) deficiencies in water supply, sanitation and hygiene are, globally, second only to malnutrition as the principal cause of death and disability-adjusted years lost. They go on to claim that in 2000, 18% of the world population was without sufficient water supplies and about 40% was without adequate sanitation. Whilst figures indicate that an impressive number of people are receiving improved water supply and sanitation, due to population growth, the overall statistics are actually deteriorating. The situation in Africa is even worse than indicated by these global figures where morbidity and mortality, due to lack of adequate safe water supplies and sanitation, are still very high. Water related diseases kill an estimated 3 million Africans annually (Rose, 2002). At the turn of the 20th century it was estimated that around 21 million South Africans did not have adequate sanitation and approximately 45% of the rural population did not have a satisfactory water supply (Rose, 2002). The South African Department of Water Affairs and Forestry (DWAF) recognise this in their management policy, where they state that improved access to water is necessary to "increase levels of health and general well being" (DWAF, 2002). They emphasise in their water quality management goal that while water resources need to be scientifically managed and conserved, they also need to meet the social and economic requirements of the country (DWAF, 2002). The South African Water Act of 1998 supports this goal by stipulating that future water resource developments should be environmentally sustainable and that a component of the natural flow of rivers should be reserved to ensure some level of ecological functioning (Hughes & Hannart, 2003). This challenge is exacerbated by the arid climate in South Africa. South Africa receives only half the world's average rainfall and, due to high evaporation rates, only 8% of this rainfall is carried in the rivers, compared to the world mean of 31% (van Zyl, 2003). The combined effect of climate, rapid population growth and inefficient water infrastructure is increasing the pressure on the river ecosystems across the whole of Southern Africa.

This threat to natural water supplies is also manifest in the Eastern Cape, where the State of South Africa Population Report (2000) notes that only about 34% of households have access to sewage treatment facilities. The provision of sewage treatment facilities does not in itself ensure satisfactory effluent water quality. In a study conducted by Mohale (2003), it was found that of the 190 treatment works listed in the Eastern Cape, only 98 (51.6%) were monitored by DWAF between 2002 and 2003. Of those that were monitored only 12% were meeting all the set discharge limits. What is of particular concern, are the high levels of indicator organisms measured in some of the effluent. Some of the 'treated' effluents in this area showed faecal coliform counts of over $10^4.100ml^{-1}$, some 100 times the discharge limit (Antrobus, 2003). Since around 1982 cholera has been spreading through South Africa from its origin in Kwazulu-Natal and has recently become endemic in the Eastern Cape (Rural Development Services Network, RDSN, 2003). The RDSN (2003) attributes this largely to the discharge of untreated or partially treated sewage.

1.2 Wastewater Treatment

Conventional domestic wastewater treatment is considered to address four principal unit operations known as primary, secondary, tertiary and quaternary treatment (Maier *et al*., 2000; Madigan *et al*., 1997). Other more detailed descriptions sometimes break these down further into additional stages such as preliminary treatment and advanced treatment (Metcalf & Eddy, 2003; Horan, 1996). Depending on the technology used, the actual pollutants may be removed in more than one stage.

1.2.1 Primary Treatment

The first aspect of primary treatment, sometimes known as preliminary treatment, is a physical process involving screens or grids and grit channels. Large debris such as plastic bags, branches and rags are removed by the screens or grids, sized accordingly. Grit channels or chambers are designed so that the velocity of the sewage (normally about $0.3 m.s^{-1}$ (Horan, 1996)) is such that the grit will be deposited but the suspended organic material passes through. The sediment is then removed by suction pumps, bucket dredges or manually.

Primary treatment also involves the separation of suspended organic matter from the waste stream by sedimentation in a primary settling tank or clarifier. The resulting settled material is known as primary sludge. An effective settling tank will remove up to 40% of the biological oxygen demand (BOD) simply by sedimentation (Horan, 1996). Primary treatment may be enhanced by chemical addition or filtration (Metcalf & Eddy, 2003). According to Horan (1996) there are four important facilities to be considered when designing a sedimentation tank. The first of these is the inlet zone which facilitates dissipation of the incoming sewage flow velocity as rapidly as possible. If excessive flow velocities are experienced at the tank inlet, there will be a reduction in settlement efficiency due to turbulence and short circuiting. The second consideration is an outlet weir that collects settled sewage prior to further treatment. This generally takes the form of a V-notched weir around the tank periphery, allowing liquid to be withdrawn as a thin layer. A scum board is normally required to prevent floating material, oil and grease from passing over the outlet weir. The largest area of the sedimentation tank is the settlement zone, where the settleable solids are actually removed. The tank capacity is determined by the size of the settlement zone, which should be free of turbulence and short circuiting. The fourth and final zone of the sedimentation tank described by Horan (1996) is the zone catering for the storage, collection and withdrawal of the solids (sludge). This may be either manual or mechanical but, to avoid the necessity for storing sludge between desludging, it is preferable to have mechanical, automated removal of sludge. The are three main types of sedimentation tanks i.e. horizontal flow, radial flow and upward flow tanks, which as their names suggest, utilise different flow patterns to achieve solid settling (Horan, 1996).

1.2.2 Secondary Treatment

Secondary treatment consists of the biological degradation of soluble and suspended organic matter. Some nutrient removal may also be included as part of the secondary treatment. While numerous technologies are available, the most commonly used in this stage are biological trickle filters or activated sludge. These processes are comprehensively reviewed by Horan (1996) and Metcalf & Eddy (2003).

Trickle filters consist of a bed of media over which the wastewater is trickled. The media may be rock, gravel, slag, ceramic material or plastic. As the water flows over the media, microorganisms decompose the organic material aerobically, converting it to microbial biomass. This forms a biofilm on the filter medium surfaces, known as the zoogleal film, composed of bacteria, fungi, algae and protozoa. Maier *et al*. (2000) define a biofilm as a layer of organic matter and microorganisms formed by the attachment and proliferation of bacteria on the surface of an object. The growth of the biofilm microorganisms is facilitated by the availability of organic compounds, mineral salts and oxygen (Degrėmont, 1991). These requirements are, in most cases, met in a moist or submersed surface environment such as that provided by a trickle filter. Biofilms are often characterised by a slimy appearance. This slime layer is created by extracellular polysaccharides secreted by the bacteria and provides a matrix for the attachment of the microorganisms (Maier *et al*., 2000). As the film develops, stratification occurs, with an aerobic outer layer, where oxygen diffusion takes place, over a deeper, anaerobic layer. Aerobic degradation products, and the organic material released by the lysis of bacteria cells, moves from the outer layer into the deeper, anaerobic layers, where they are further broken down by fermentation (Degrėmont, 1991). Alcohols and fatty acids produced in the fermentation process are then dissolved back into the water stream or utilised by the aerobic bacteria. This stratification is illustrated in Figure 1.1. Although organic removal is effective in trickle filters, between 60 and 85% of BOD (Degrėmont, 1991), the removal of enteric pathogens has been found to be low and erratic (Maier *et al*., 2000).

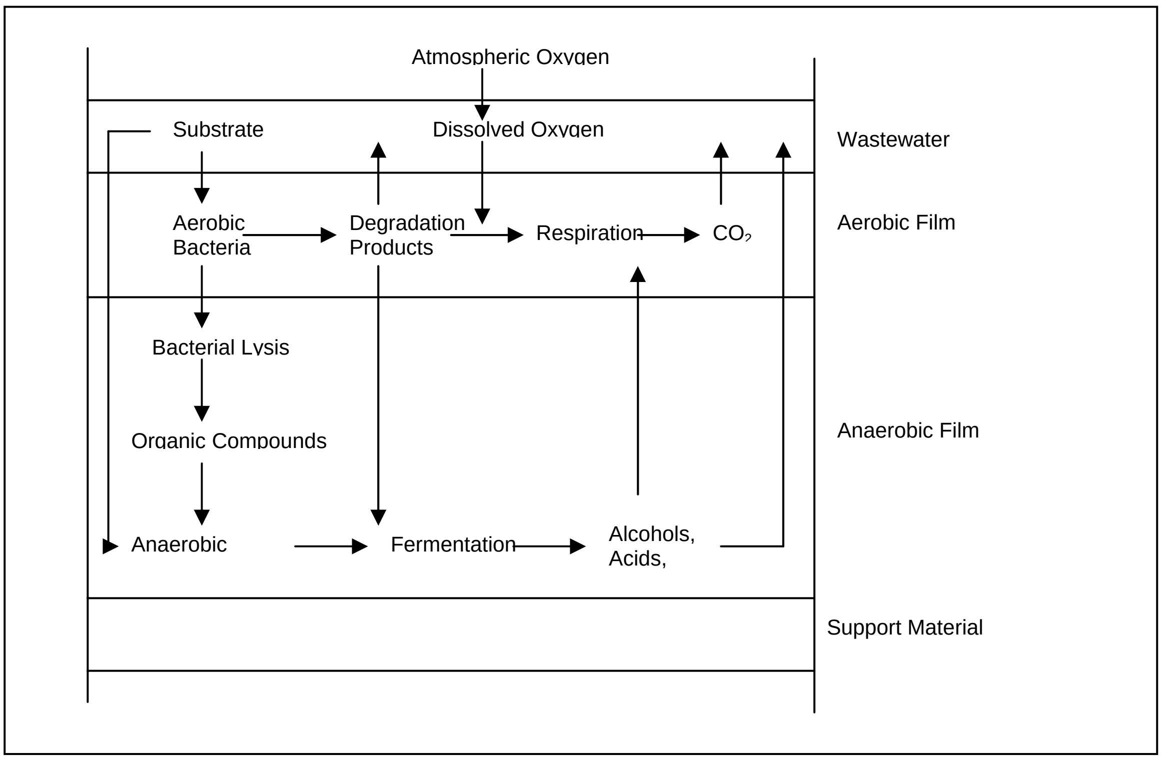

Figure 1.1: Section of a biofilm, typical of trickle filters. After Degrémont (1991).

In activated sludge treatment, primary effluent is mixed with a bacteria-rich slurry and vigorously aerated. This results in oxidation of the organic material and the formation of bacterial floc. The slurry then goes to a secondary settling tank, where the treated effluent is siphoned off the top and the secondary sludge settles at the bottom. Some of the sludge is then recycled and used as inoculum for the incoming primary effluent, while the remainder is removed. Due to the large number of microorganisms being made available by this recycling, oxidation of organic matter occurs in a relatively short time. Apart from the availability of microorganisms, it is important to control the food-to-microorganism ratio (F/M) in activated sludge plants, expressed as BOD per kilogram per day (Maier *et al*., 2000). A high F/M ratio indicates an excess of food relative to microorganism numbers, while a low F/M ratio indicates nutrient limitation and thus more efficient wastewater treatment (Maier *et al*., 2000). Up to 90% of BOD can be removed in a well operated, conventional activated sludge plant (Horan, 1996).

1.2.3 Tertiary and Quaternary Treatment

The distinction between tertiary and quaternary treatment varies depending on the information source (Horan, 1996; Mujeriego & Asano, 1999; Weber & Leboeuf, 1999; Metcalf & Eddy, 2003). Tertiary treatment, however, generally involves the removal of residual organics, nutrients and pathogens, while quaternary treatment removes metals, salts, pesticides (Malato *et al*., 1999) and often includes further disinfection. The amount of final treatment required depends on the ultimate destination of the water.

Conventional tertiary treatment usually consists of physicochemical processes such as coagulation, filtration, activated carbon and chlorination (Mujeriego & Asano, 1999). Mujeriego and Asano (1999) describe filtration as a solid/liquid separation process, where particles, larger than 3μm, are removed by impaction, interception and physical straining as wastewater moves through a column of granular medium. Filtration is often required to reduce solid loading on more advanced treatment stages such as reverse osmosis (RO). Cikurel *et al*. (1999) found that filtration is improved by adding a coagulant such as alum and they achieved reductions in turbidity of 65-70%, particulate matter of 70-80% and phosphates up to 90%. Due to their low water solubility, hydrophobic organic compounds such as chlorinated organic solvents can be removed by activated carbon adsorption, however, larger or water soluble compounds are better removed by oxidation or ultrafiltration (Mujeriego & Asano, 1999).

Quaternary treatment is sometimes referred to as advanced treatment and is usually only necessary where a high level of treatment is required for irrigation or recreation reuse. Quaternary treatment makes use of more advanced technology such as membrane processes and ion exchange. Ion exchange is effective in softening and removal of cations such as calcium, magnesium and iron as well as anions such as nitrate (Mujeriego & Asano, 1999). In membrane processes, membrane pore size ranges from 200nm in microfiltration through ultrafiltration (2-50nm), nanofiltration (<2nm), down to less than 1nm in reverse osmosis (Weber & Leboeuf, 1999). Various contaminants are targeted, depending on the membrane size used and include suspended solids, *Giardia* and bacteria (Weber & Leboeuf, 1999), viruses and proteins (Herath *et al*., 1999), organics, pesticides, metals, dissolved solids, nitrates and radionuclides (Weber & Leboeuf, 1999).

Miserez *et al*. (1999) have described advanced biological treatment methods under development. Genetic engineering is one of the options available to make biological treatment suitable for advanced treatment. In an attempt to construct more efficient degradative strains and thereby ensure more complete mineralization of contaminants, essential genes are selected. According to Miserez *et al*. (1999), this should result in superfluous genetic material, coding for unproductive or counterproductive enzymes, being eliminated.

1.2.4 Sludge Management

A major portion of operational time and costs at conventional treatment works is associated with solids (sludge) processing. Although sludge volume is less than 1% of the total effluent, sludge handling may account for as much as 60% of the total capital operating costs (Horan, 1996). Sludge is removed from primary, secondary and advanced treatment stages and needs to be thickened, stabilised, dewatered and disposed of (Maier *et al*., 2000). Stabilisation can be achieved by aerobic and anaerobic digestion, composting, chemical addition and heat treatment. Most modern plants use anaerobic digestion (Ross *et al*., 1992). Ross *et al*. (1992) describe some of the advantages of anaerobic digestion. These include: the volume of the sludge is significantly reduced by the conversion of organic matter into gaseous end-products; the obnoxious odour of the sludge is removed; fats and grease are broken down; pathogens are significantly reduced; the change in chemistry, after digestion, makes the sludge potentially suitable for agricultural use and the biogas produced can be used as a fuel source. There are, however some disadvantages such as: apart from anaerobic ponds, relatively high costs are involved; the slow growth rate of anaerobic bacteria requires long periods for start-up and limits flexibility to adjust to changes in feed loads and environmental conditions and close monitoring and control is required to prevent process failure.

Equipment such as filter presses and centrifuges effectively dewater sludge but most facilities in developing countries lack the money necessary for this type of technology and simply use sand drying beds for dewatering. This applies to treatment works in Southern Africa as well (Ross *et al*., 1992). Dried sludge still needs to be disposed of by methods such as incineration and land application (Dean & Lund, 1981). A third disposal route is co-disposal in a solid waste landfill. Sludge may be desirable in landfills for inoculating the refuse with methanogenic bacteria and providing moisture that promotes and accelerates the microbiological, physical and chemical attenuation mechanisms responsible for decomposition (Ross *et al*., 1992). Other less common means of disposal include pellitisation, active sludge pasteurisation, use as an energy source such as co-combustion in coal fired power stations, composting and clay brick making (Slim, 1991; Hosetti & Frost, 1995; Lotter & Pitman, 1997).

Land application of wastewater sludge as a soil conditioner is beneficial in that it breaks up clayey soils and improves the moisture retaining ability of sandy soils (Ross *et al*. 1992). Ross *et al*. (1992) also argue that although the nitrogen, phosphorus and potassium content of sludge is not as high as inorganic fertilizers, there is a slower release of these elements, which helps with the nutrient enrichment of soils. Sludges also contribute humic matter to soils, which is important in South Africa, where soil organic content is generally low (Korentajer, 1991). Despite the benefits of sludge application to land, the use of sludge in agriculture is restricted in South Africa by strict guidelines pertaining mainly to nitrogen loading rates and heavy metal content (van der Waals & Snyman, 2004). A survey by Snyman *et al*. (2004) revealed that out of 72 wastewater treatment plants in South Africa, 60% exceeded the metal content outlined in the Permissible Utilisation and Disposal of Sewage Sludge. van der Waals and Snyman (2004) maintain that because of these restrictions and the high cost of landfill disposal, the easiest and cheapest option is sacrificial land disposal. To prevent contamination of groundwater, liming of these sites to near neutral pH is recommended. Liming, however, would then have cost and manpower implications.

1.3 Water Recycling

Up until the 1980s, the trend in water resource management was supply-driven management, characterised by managers and policy makers focusing on maximising the amount of water available for direct use (Al Radif, 1999). This approach has not solved world wide water shortages and, as with other natural resources, the shift is, therefore, towards more sustainable water use (Jewitt, 2002). Integrated Water Resource Management (IWRM) is thus advocated as a better strategy for maintaining water resources (Al Radif, 1999). Definitions vary but Jewitt (2002) explains that IWRM should meet human requirements for the use of freshwater and provision of goods and services, whilst maintaining hydrological and biological processes as well as biodiversity which are considered essential for the functioning of ecosystems. Thomas and Durham (2003) expand on this definition by recognising that IWRM is multidimensional i.e. it has time, space, discipline and stakeholder aspects. Time means that actions now should bear in mind the interests of future generations. Space dimension considers the management of water at the level of river basin or watershed and the necessity to look globally before acting locally. There are a number of different disciplines that have to be incorporated into the decision making process, including science, technology, legislation, politics, socio-economic impacts, historical and cultural issues. All stakeholders need to be involved as early as possible in order to resolve any conflict of interests. One of the most important areas of IWRM is the development of alternative water resources (Thomas & Durham, 2003) and it’s here that wastewater recycling fits into this management plan.

A major benefit of adequately treating wastewater is that it may be reclaimed and form a valuable source of raw water for certain applications. Many industries have closed loop systems, where the water is treated on site and fed directly back into the process e.g. Eraring Power Station, New South Wales, Australia (Thomas & Durham, 2003) and Iscor, Saldanha Steel, South Africa. However, most domestic wastewater (treated and untreated) is discharged into the nearest water course and is not available for reuse, especially if the receiving water body is the sea. Shelef (1991) suggests that recycling, reuse, reclamation, recovery and renovation should form the basis for the solution to environmental pollution and that this is particularly true of fresh water, which is the commodity most threatened by pollution and, on a world wide scale, is the most scarce. The most obvious areas for wastewater reuse would be low rainfall areas of the world (such as Southern Africa), where meagre water resources are under threat of pollution due to lack of dilution and dispersion and where there is a demand for water for irrigation purposes (Shelef, 1991). Wastewater reuse has been practised for centuries in many parts of the world, beginning as far back as the Minoan civilisation, 5 000 years ago, but problems in developing adequate water resources have led to recent interest in wastewater recycling (Asano & Levine, 1996).

Although the main application for wastewater reuse has been agriculture, there are different levels of quality required for various types of agriculture. There are also many other potential users of treated wastewater. Technologies exist today to treat water to just about any specified quality (Metcalf & Eddy, 2003); however, high levels of treatment may not always be required or cost effective. Population increase and growing water demand have led to a global deterioration in surface water quality and, in water stressed areas, more and more emphasis will be placed on the reclamation and reuse of wastewater, especially for requirements such as irrigation and, in some cases, even dual water supply for domestic non-potable use e.g. toilet flushing (Law, 1996; Shepherd, 2003)

Wastewater reuse can be grouped into a number of categories. Mujeriego and Asano (1999) divide these categories as follows: irrigation of urban areas - either unrestricted access such as parks, school fields and golf courses or restricted, infrequent access such as cemeteries, freeway verges and greenbelts. Urban reuse might also include fire protection or dual systems for toilet flushing. Irrigation with wastewater effluent is common; depending on quality, this may be of food crops, consumed directly by humans or non-food and animal food crops e.g. cotton and lucerne. Treated effluent may also have recreation uses – either unrestricted e.g. lakes and dams used for swimming or restricted e.g. fishing, boating and other non-contact recreational activities. There is potential for environmental use e.g. artificial wetlands and to sustain stream flows. Groundwater recharge to control salt water intrusion and subsidence as well as maintain aquifers as a water source is another option. Industrial reuse of water, include cooling-system make up water, boiler feed water and dust suppression. Potable water supply from wastewater is possible but requires extremely strict and advanced treatment to achieve suitable quality.

With any reuse of wastewater, human health risks remain the major concern and the removal of pathogens, including bacteria, helminths, protozoa and enteric viruses is, therefore, critical in wastewater treatment and there is increasing importance being placed on this aspect at treatment facilities (George *et al*., 2002).

1.4 Wastewater Disinfection

It is generally accepted that the majority of microbiological health hazards associated with water consumption originate from faecal contamination (Dean & Lund, 1981; George *et al*., 2002; Amahmid *et al*., 2002). *E. coli, Shigella* sp., *Salmonella* sp. and *Vibrio cholerae* causing respectively diarrhoea, dysentery, typhoid fever and cholera are some of the pathogenic bacteria listed by Dean and Lund (1981) that may occur in faecal contaminated water. Viruses, causing diseases such as meningitis and hepatitis, as well as parasitic protozoans and helminths, are also usually present in domestic wastewater. According to Dean & Lund (1981) primary treatment can eliminate between 40 and 70% of the bacteria, whilst biological processes such as trickle filters and activated sludge are effective at removing up to 99% of the pathogenic microorganisms. However, in order to make wastewater safe for reuse, further disinfection is necessary (Maynard *et al*., 1999), a coliform count of <1000cfu/100ml is required by the South African DWAF (2001) for irrigation with reclaimed water (crops that are eaten raw, require water with a maximum of 1cfu/100ml).

Some pathogens are present in very low numbers in wastewater and, because of this, or because effective isolation techniques have not yet been developed, are difficult to detect. Microorganisms that are more numerous and more easily tested are, therefore, commonly used as indicators of faecal contamination. The digestive system contains a large population of rod-shaped bacteria known collectively as coliform bacteria and each individual may discharge between 100 and 400 billion coliform bacteria per day (Metcalf & Eddy, 2003). Thus, since their first isolation from faeces towards the end of the 19th century (Rompre *et al*., 2002), the presence of the coliform group in water has been taken as an indication that pathogenic organisms associated with faeces may also be present. Definitions of coliform may differ slightly depending on the regulatory body responsible for the monitoring regulations but most are essentially based on common biochemical characteristics (Rompre *et al*., 2002). The following descriptions are taken from the Standard Methods for the Examination of Water and Wastewater (1998):

1. All aerobic and facultative anaerobic, Gram negative, non spore-forming, rod-shaped bacteria that ferment lactose with gas and acid fermentation within 48 hours at 35ºC (multiple-tube fermentation technique: section 3.1) or;
2. All aerobic and many facultative anaerobic, Gram-negative, non spore-forming, rod-shaped bacteria that develop a red colony with a metallic sheen within 24 hours at 35ºC on an Endo-type medium containing lactose (membrane filter technique: section 3.2).

Coliform bacteria may, however, originate from a variety of sources and can for instance grow in soil (Metcalf & Eddy, 2003). Their presence does not necessarily, therefore, mean contamination with faecal waste. Tests have thus been developed that distinguish faecal coliform and, specifically *Escherichia coli*, which is the most common coliform among the intestinal flora of warm blooded animals and therefore more indicative of faecal contamination (Rompre *et al*., 2003). Other organisms such as faecal streptococci, enterococci and *Clostridium perfringens* have also been proposed for use as indicators (Metcalf & Eddy, 2003) but coliform, faecal coliform and *E. coli* remain the most commonly reported organisms in the literature and legislation. Some of the bacteriophages are also sometimes useful indicators as they signify the presence of important human viral pathogens (Davies-Colley, 1999).

Disinfection refers to the partial destruction of pathogens to acceptable limits which differs from sterilisation, where all living organisms are destroyed or removed. Although sterilisation is possible with small quantities of water in a laboratory environment, for all practical purposes it is impossible when dealing with large flows in a wastewater treatment plant. The following description of the four most conventional disinfection methods is as described by Metcalf & Eddy (2003) and Dean and Lund (1981).

1.4.1 Chlorine

Since its introduction at the end of the 19th century, chlorine has remained the principal disinfectant for water (Dean & Lund, 1981; Lazarova *et al*., 1999). It is a strong oxidant and halogenating agent, both of which contribute to its use as a disinfectant. Some of the advantages of chlorine are that its application is a well established technology; it has a residual effect, therefore continues to work for a relatively long period after dosing and, as an oxidant, chlorine destroys odours such as hydrogen sulphide, mercaptans and other products of anaerobic decay. There are, however, a number of disadvantages to the use of chlorine. These include it being a hazardous chemical to work with, potentially toxic to the biota in the receiving environment, increasing the total dissolved solids (TDS) of the effluent and it may be consumed by oxidising inorganic compounds such as iron and magnesium (Metcalf & Eddy, 2003). In a study on two rivers in Kwazulu-Natal, South Africa, Williams *et al*. (2004) found that chlorination of sewage effluent led to large scale localised destruction of riverine macroinvertebrates. Chlorine also has the potential to form carcinogenic substances known as trihalomethanes e.g. chloroform, bromoform, by its action on a variety of oxygenated organic compounds such as acetone (Lazarova *et al*., 1999). Due to these negative effects dechlorination plants are often necessary, which add a significant cost increase to water treatment.

1.4.2 Chlorine Dioxide

Chlorine dioxide is also an effective disinfectant with a good residual value and capability to oxidise odorous sulphides (Metcalf & Eddy, 2003). Apart from also affecting receiving water quality by increasing TDS and forming toxic substances such as chlorite when reacting with organic matter, the main disadvantage is that it is unstable and must be produced on site, greatly increasing operating cost (Dean & Lund, 1981).

1.4.3 Ozone

Ozone is another effective option (van Leeuwen, 1996; Ernst & Jekel, 1999; Liberti & Notarnicola, 1999). It has the added benefits of oxidising sulphides and contributing dissolved oxygen. However, it carries with it many safety concerns as it is both highly corrosive and toxic (Metcalf & Eddy, 2003). It is also energy intensive and expensive, with high maintenance and operational requirements.

1.4.4 Ultraviolet

According to Bergmann *et al*. (2002), ultraviolet light (UV) is becoming more widely used as a disinfectant and is now well documented in the literature (Almasi & Pescod, 1996; Fallowfield *et al*., 1996; Moreno *et al*., 1997; Sommer *et al*., 1998; Davies-Colley *et al*., 1999; Liberti & Notarnicola, 1999; Andreadakis *et al*., 1999; George *et al*., 2002; Craggs *et al*., 2004). The main advantage of UV over the previously discussed techniques is that it is safer to work with and does not have any residual toxicity or negative effect on the effluent quality. It is however energy intensive and relatively expensive. Another concern with UV disinfection is that while it is very effective in reducing the culturable faecal coliform count it does not necessarily eliminate the micro-organisms. Studies by George *et al*. (2002) found that β-D- glucuronidase activity was not reduced by UV disinfection suggesting that the faecal coliform continued to exist in a viable but non culturable state.

1.4.5 Viable but Non-Culturable Cells

It is suspected that indicator organisms such as *E. coli* often exist as viable but non-culturable cells (VBNC), meaning that the organism is in a state of metabolic shut down that prevents its growth on a culture medium, but is not actually dead (Edwards, 1999; George *et al*., 2002). There are two conflicting schools of thought interpreting the VBNC state: one is that it is a survival strategy and, as such, cells should be able to reverse the process when conditions become favourable. Conversely, it may be a moribund condition in which cells become progressively debilitated until death finally occurs (McDougald *et al*., 1998). Bloomfield *et al*. (1998) give a possible explanation for the inability to culture cells in such a state. Bacteria undergo both biochemical and physiological adaptations that enable them to survive environmental stress such as nutrient limitation or UV exposure (Edwards, 1999). An important consequence of this is a reduction of growth rate to near zero. When such organisms are transferred to rich culture media, the metabolic pathways are rapidly switched on and flooded. The oxidation of substrates leads to overproduction of superoxide and free radicals, resulting in cell death (Bloomfield *et al*., 1998). The inability to detect VBNC by traditional culture methods, usually following membrane filtration, has led to the development of alternative analysis techniques such as the detection of the enzymes, β-D- glucuronidase and β-D-galactosidase, produced by *E. coli* (Apte *et al*., 1995; Rompré *et al*., 2002) and the use of viability dyes (Edwards, 1999). Although the existence of VBNC is widely accepted, the extent to which inactivated pathogens remain virulent is still under investigation (McDougald *et al*., 1998; Rompré *et al*., 2002; George *et al*., 2002)

1.4.6 Passive Disinfection

There are a number of other disinfection techniques that have been tested with various degrees of success e.g. using hydrogen peroxide (Drogui *et al*., 2001), high energy electrolysis (Miyata *et al*., 1990; Bergmann *et al*., 2002), titanium dioxide as a photocatalyst (Herrera Melian *et al*., 2000) and ultrasound (Mason *et al*., 2003). These methods are, however, not yet widely used and remain, for the most part, expensive, requiring high energy and expertise input.

Wastewater disinfection may also be affected in more passive treatment operations. Waste stabilisation ponds (WSP) have been fairly intensively investigated for their disinfection efficacy (Fernandez *et al*., 1992; Pearson *et al*., 1995; Almasi & Pescod, 1996; Rangeby *et al*., 1996; Jagals & Lues, 1996; Garcia & Becares, 1997; Sarikaya & Saatci, 1988; Maynard *et al*., 1999; Amahmid *et al*., 2002; Kadlec, 2003). The long retention times in WSP enables factors such as sedimentation, predation, competition and sunlight to all contribute to the die-off of pathogens. While these ponds may not be able to achieve the same level of disinfection as the methods discussed above, they have no anthropogenic energy input requirement and are very low operation and maintenance systems. Wetland systems work on the similar low energy, low cost concept and in addition to the disinfection mechanisms active in ponds also use filtration to eliminate pathogens. Work done on wetlands indicate a similar level of disinfection as waste stabilisation ponds (Ansola *et al*., 2003; Kadlec, 2003; Fujioka *et al*., 1999; Arias *et al*., 2003; Gearhart, 1999; Garcia & Becares, 1997). High rate ponds provide potentially the most effective disinfection within the constraints of sustainability requirements (El Hamouri *et al*., 1994; Oswald, 1994; Fallowfield *et al*., 1996; Bahlaoui *et al*., 1997; Davie-Colley *et al*., 2003). The disinfection mechanisms active in these ponds are high pH, sunlight, high oxygen production as well as some of the other factors, such as predation, present in conventional ponds. The disinfection capability of high rate ponds is the focus of a portion of this thesis and will be discussed in greater detail in chapter 4, 5 and 6.

1.5 Waste Stabilisation Ponds

Conventional systems, such as trickle filters and activated sludge, speed up biological oxidation of organic material by mechanical aeration. In the case of the latter, this requires a relatively large amount of energy and, often, fairly intensive maintenance (Horan, 1996). In addition, these systems require some sort of tertiary treatment in order to disinfect discharged effluent. Many smaller communities and parts of the developing world lack access to the technical expertise and funding requirements necessary to operate such systems. Waste stabilisation ponds (WSP), or lagoons, therefore provide a relatively low-cost, simple alternative for domestic wastewater treatment in these areas (Nelson *et al*., 2004). Although the earlier ponds were very rudimentary compared with some of the advances that have been made in ponding technology, they are one of the oldest systems used to treat domestic wastewater (Maier *et al*., 2000).

Mara and Pearson (1986) define WSP as shallow basins into which wastewater continuously flows and from which treated effluent is discharged. Horan (1996) explains that the degree of treatment is a function of the number of ponds in series and the retention time of the wastewater in each pond. Although the number of ponds and retention time have a major effect on the quality of the effluent, it is possible to manipulate each individual pond to achieve a desired function (Hosetti and Frost, 1998). A wide range of pond types therefore exists, allowing configuration flexibility to suit different conditions and discharge standards (Mara & Pearson, 1986; Pearson, 1996). According to Pearson (1996) correctly designed ponds can match the effluent quality of other wastewater treatment technologies.

Detailed descriptions of the different kinds of WSP can be found in the literature e.g. Horan (1996), Mara and Pearson (1986), Metcalf and Eddy (2003) and Maier *et al*. (2000). Below is a description of the various pond types taken from these texts:

1.5.1 Anaerobic Ponds

Anaerobic ponds are designed to be totally devoid of dissolved oxygen. This is achieved by making the ponds deeper than other pond types, typically 2 – 5m and maintaining a high organic load, BOD_5 > 300mg/l (Mara & Pearson, 1986).

1.5.1.1 Organic Removal Mechanisms Operating in Anaerobic Ponds

Carbon in Earth's atmosphere is kept in balance globally by cycling between abiotic forms such as carbon dioxide and biotic forms such as sugar and other carbohydrates (Maier *et al*., 2000). Figure 1.2 below illustrates both the aerobic and anaerobic contributions to this cycle.

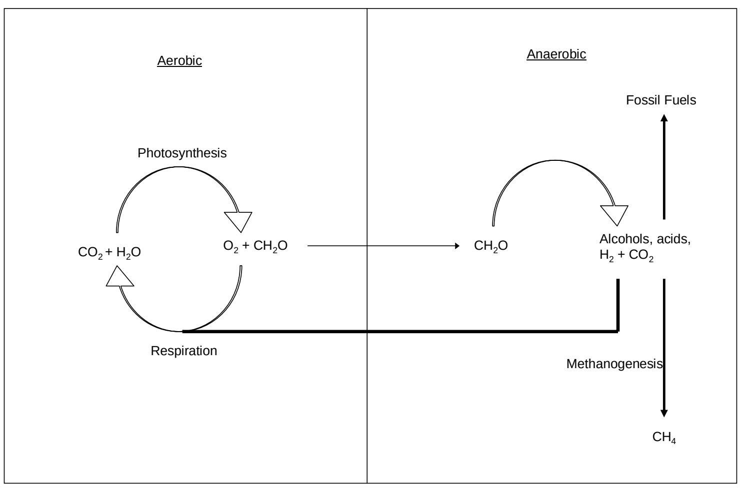

Figure1.2: Aerobic and anaerobic contributions to the carbon cycle. Taken from Maier *et al*. (2000)

Some of the cellular components are less degradable than others under anaerobic conditions and have accumulated in sediments over geological time giving rise to the present-day fossil fuel reserves (Maier *et al*., 2000). The other carbon compound produced under anaerobic conditions is methane. This anaerobic methane production or methanogenesis is exploited in the degradation of organic material present in the fermentation pit.

According to Maier *et al*. (2000) the organisms responsible for methanogenesis are a group of obligately anaerobic archaebacteria known as the methanogens. Autotrophic methanogens produce methane, utilising CO_2 as the terminal electron acceptor and H_2 as the electron donor. The exothermic reaction given by Maier *et al*. (2000) and Horan (1996) is:

$$4H_2 + CO_2 \rightarrow CH_4 + 2H_2O \quad \Delta G = -130\text{kJ/mol} \tag{1}$$

Methanogens can also produce methane during heterotrophic growth on a limited number of substrates such as acetate and methanol:

$$CH_3COOH \rightarrow CH_4 + CO_2 \tag{2}$$

As methanogens are dependent on the production of these substrates by other microorganisms, they generally exist in a symbiotic relationship with these acid-forming bacteria (Henze *et al*., 2002). A large proportion of the oxygen demand in influent wastewater is in the form of suspended solids, which are removed in anaerobic ponds by sedimentation (Horan, 1996). This sedimentation results in the development of a sludge blanket, where an anaerobic microbial population becomes established. The settled organic substrate, containing proteins, fats and carbohydrates, is initially degraded by extracellular enzymes in a process known as hydrolysis (Henze *et al*., 2002). The resultant constituent monomers such as glycerol, sugars and amino-acids are then further broken down by acidogenesis, producing volatile fatty acids (VFA) and, in particular, acetate (Horan, 1996). The bacteria catalysing this fermentation stage are known as the acid-formers and typical genera include *Clostridium, Propionibacterium* and *Bacteroides* (Horan, 1996). This VFA formation is known as putrefaction and some of the end products such as butyric acid are extremely malodorous. Fortunately, however, methanogens then utilise these acids in the methanogenesis. The resulting gaseous end products are odourless and when they escape to the atmosphere, they assist in the removal of oxygen demand (Horan, 1996). The mineralization of polymers through to gaseous end products is summarised in Figure 1.3. Anaerobic metabolism is very inefficient compared with respiration and consequently has a low cell yield. According to Horan (1996), as much as 70% of the BOD removed in an anaerobic pond will be in the form of methane gas.

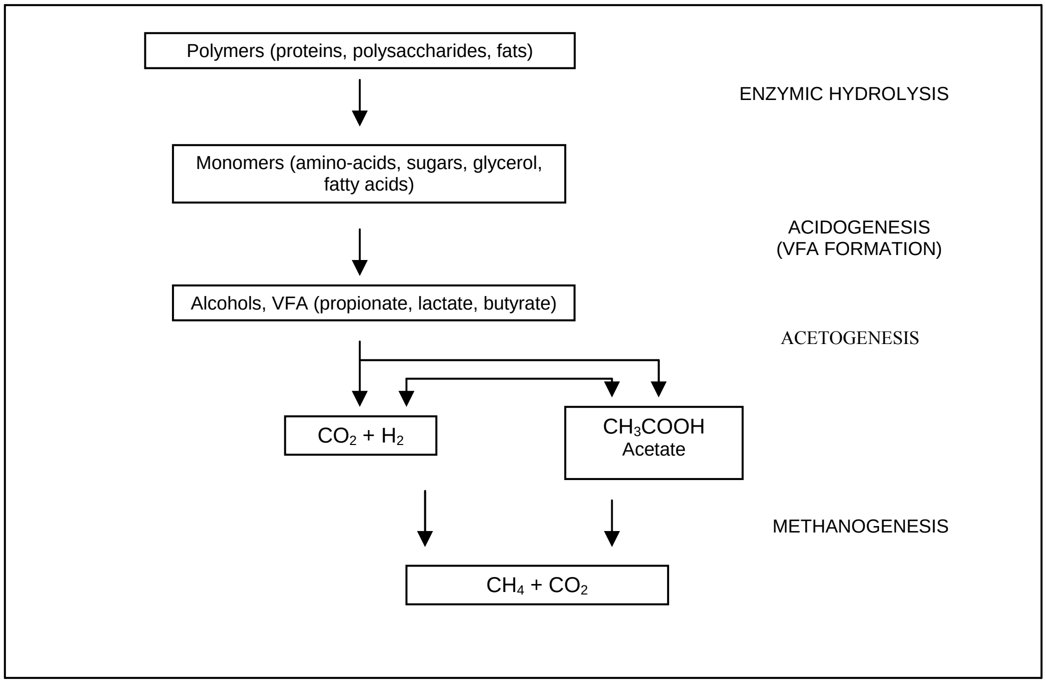

Figure 1.3: Degradation of organic material in the fermentation pit. Adapted from Horan (1996), Madigan et al. (1997) and Degrémont (1991).

Methanogens are, however, sensitive to their environment and if an excess of VFA are produced, with a reduction in pH, methanogenesis will be inhibited and ultimately stop. The growth rate of the methanogenic bacteria is thus the rate limiting step and determines the organic load to an anaerobic pond (Horan, 1996).

Sedimentation also plays a role in removing organic pollutants from the water in these ponds. Most of the suspended solids settle to the bottom, where they also undergo anaerobic digestion.

1.5.2 Facultative Ponds

Facultative ponds are typically shallower than anaerobic ponds and have a lower anoxic zone and an upper aerobic zone, where most of the oxygen is provided by algal photosynthesis. The combination of aerobic and anaerobic metabolism within the same pond results in very effective nutrient recycling, although the main purpose of facultative ponds is BOD reduction. Anaerobic degradation of sediment sludge allows minimal desludging. Many of the anaerobic gases get oxidised as they pass through the aerobic upper layers, reducing odours emitted from the facultative pond. Dissolved organic material is also removed in this oxygen rich upper layer; either by oxidation to CO_2 or assimilation into biomass (Henze *et al*., 2002). On an organism level, these functions occur predominantly via glycolysis and the Krebs Cycle (Campbell, 1993):

$$\text{Organic Compounds} + \text{Oxygen} \rightarrow \text{Carbon Dioxide} + \text{Water} + \text{Energy} \qquad (3)$$

Facultative ponds may serve as primary facultative ponds fed with raw wastewater or secondary facultative ponds, receiving settled wastewater.

1.5.3 Maturation Ponds

The principal function of maturation ponds is pathogen removal. Tertiary treatment operations, such as additional BOD and nutrient removal, may also occur in these ponds. In order to achieve satisfactory pathogen reduction, maturation ponds are generally designed to have a long retention time. Due to their relatively large size, helminth eggs and protozoal cysts are thought to be removed by sedimentation. A retention time of longer than 11 days is thought to be sufficient (Horan, 1996), although some studies have found that as much as 37 days is required for effective removal (Wiandt *et al*., 1995). Whilst these parasites are removed from the water, they may still remain active in the sludge layer on the bottom of the pond, which is an important consideration should it become necessary to empty or desludge the pond.

1.5.4 Specialised Pond Types

While the above three pond types are the most common waste stabilisation ponds in use, there are, however, specialised ponds that are sometimes used for wastewater treatment. These include high rate algal ponds (HRAP) and macrophyte ponds. A HRAP is a shallow, paddle wheel mixed pond, which is designed to enhance exposure of the algae to sunlight and avoid thermal stratification, thereby maximising growth, photosynthesis and productivity (Oswald, 1988a; Oswald, 1988b; Horan, 1996). This results in a surplus of dissolved oxygen, high pH as well as a high rate of carbon assimilation and nutrient uptake (Oswald, 1988a; Rose *et al*., 2002a). All of these contribute to the efficacy of HRAP as a combined secondary and tertiary treatment operation. The function and performance of HRAP will be examined in more detail throughout this study. Macrophyte ponds are ponds where aquatic plants are grown, either on the pond surface (floating macrophyte) such as water hyacinth or attached to the bottom of the pond (rooted macrophyte). Macrophyte ponds remove suspended algae and thus further reduce BOD in WSP effluent. Faecal coliform removal is, however, negligible (Horan, 1996).

1.5.5 Application of Waste Stabilisation Ponds

According to Gloyna and Tischler (1979), WSP typically contain abiotic substances, producer organisms, consumer organisms and decomposers. Microbial ecology of ponds involves a complex interaction of these various groups. One of the most important being the mutualistic relationship between algae and bacteria (Mara & Pearson, 1986), although protozoa, rotifers and ciliates all have a role in the functioning of a pond system. Algae produce the oxygen required for the oxidation of malodorous gases, produced by anaerobic digestion, as well as for the aerobic functioning of the ponds. CO_2 and released mineral elements, resulting from these activities, are then utilised by the algae in photosynthesis.

WSP have many advantages over more conventional technologies for wastewater treatment. The most widely recognised of these advantages is their low operational and maintenance requirements, from the aspects of both cost and skills input (Stott *et al*., 2003; Nelson *et al*., 2004; Mara & Pearson, 1986; Horan, 1996; Gloyna & Tischler, 1979; Oswald, 1995; Wood *et al*., 1995; Ceballos *et al*., 1995; Shelef & Kanarek, 1995). Because of this, WSP are suited for rural, more isolated areas in developed countries, and as an appropriate technology in developing countries (Meiring & Oellermann, 1995, Wood *et al*., 1995; Emparanza-Knörr & Torella, 1995; Racault *et al*., 1995; Stott *et al*., 2003). As has been mentioned previously, ponds are also effective in reducing pathogens, which is particularly pertinent in cases where the water is to be reused in agriculture or aquaculture (Mara & Pearson, 1986). Mara and Pearson (1986) also list the ability to absorb hydraulic and organic shock loads and tolerance to high heavy metal concentrations as advantages of WSP. The last significant advantage of ponds is the capacity to produce beneficial by-products such as algal biomass, which can be used for human and animal consumption, and methane for energy production (Pearson, 1996; Hosetti & Frost, 1998; Green *et al*., 1995a).

WSP do, however, have some distinct disadvantages. Because these systems rely on biological activity, they are generally characterised by a low specific reaction rate, which means that a long retention time and thus large areas are necessary to achieve sufficient effluent quality (Hobus & Hegemann, 2003; Racault *et al*., 1995). Xian-wen (1995) believes the cost of land is the main restriction in the implementation of pond systems. WSP are therefore not ideal in heavily built up areas, where land is expensive, or for treating the wastewater from very large cities, where the size of the treatment facility would be unfeasible, although examples of ponds over 500ha, receiving wastewater from over a million people, do exist in New Zealand (Archer & Mara, 2003), Melbourne Water, Australia (ref) and the sewage from Nairobi, Kenya is treated in a 96ha WSP system (Pearson *et al*., 1996). With artificial aeration, degradation rates are increased and ponds size can therefore be decreased but a supplementary energy supply is necessary (Hobus & Hegemann, 2003). However, the necessity for additional energy input detracts from the main appeal of WSP. Because WSP are dependent on a long hydraulic retention time (HRT), they are liable to malfunction if short circuiting occurs. Although this is usually more of a design flaw than a technology failing, it may be seen as a disadvantage in using WSP. There are, however, a number of ways to avoid this problem. Correct positioning of inlets and outlets (Shilton & Harrison, 2003), having a number of ponds in series (Horan, 1996) and using baffles (Pearson *et al*., 1995; Archer & Mara, 2003) are some of the options available to prevent short circuiting. WSP generally have a similar problem to activated sludge or trickle filters and that is the accumulation of secondary sludge that has to be dealt with. Methane fermentation is minimal and carbon rich organic material is integrated by bacteria and microalgae that grow and settle (Green *et al*., 1995b). According to Green *et al*. (1995a) and Oswald (1990), this leads to sludge accumulation, decreasing pond volume and treatment capacity, causing the ponds to age prematurely and requiring frequent desludging. Oswald (1990) also mentions the potential of ponds for odour nuisance generation as another drawback. Craggs *et al*. (2003) found that ponds treating dairy farm wastewater often had elevated nutrient levels, BOD and faecal indicators, whilst algal growth was limited by high ammonia and sulphide levels as well as restricted light penetration.

1.6 Advanced Integrated Wastewater Ponding Systems

Oswald (1990) argues that it is possible to maintain the advantages of waste stabilisation ponds whilst mitigating their shortcomings. He has achieved this with the development of the Advanced Integrated Wastewater Ponding system (AIWPS). These systems are capable of enhancing and optimising the natural treatment processes that occur in conventional WSP, thereby improving effluent quality (Craggs *et al*., 2003). Wastewater treatment by AIWPS is based on the same model employed in conventional systems i.e. primary sedimentation, flotation, fermentation, aeration, secondary sedimentation, nutrient removal, storage and final disposal.

Oswald (1987) describes his conception of the AIWPS design as follows:
“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?" Figure 1.4 illustrates the pond sequence, without the maturation pond.

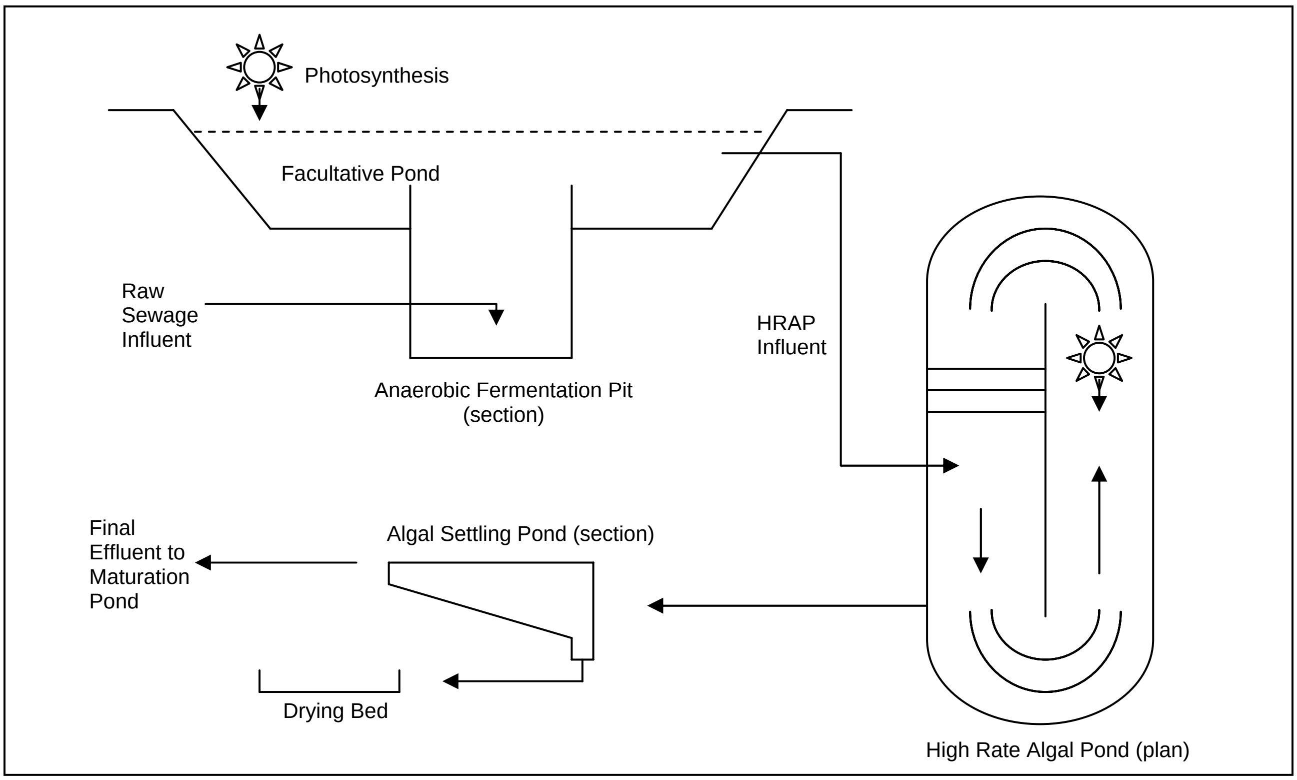

Figure 1.4: Schematic diagram of the principal unit operations associated with the IAPS plant design constructed at the Rhodes University Environmental Biotechnology Experimental Field Station, Grahamstown Disposal Works. After Rose *et al*. (2002).

Through research and application, the above thinking led eventually to the design of the AIWPS by Prof. Oswald (Rose *et al*., 2002a). This design, as described by Oswald

(1988a, 1990, 1991a, 1994), consists of four ponds in series. The first is a facultative pond, with an anaerobic internal fermentation pit. Next is a paddle wheel mixed, shallow high rate pond followed by an algal settling pond and, lastly, a maturation pond.

1.6.1 Primary Facultative Pond and Fermentation Pit

Raw sewage enters the primary facultative pond (PFP) near the bottom of the fermentation pit, which is constructed in the base of the pond. These pits are deeper than the pond floor and have a surrounding berm or wall to prevent the intrusion of oxygenated water. This is an advancement over conventional facultative ponds, where wind can cause inversions and mixing, resulting in oxygenated water intruding into the anaerobic zone, inhibiting anaerobic digestion. The overflow velocity in the pit is low enough (less than 1.5m/day) to allow the solids to settle. Most helminth ova and parasite cysts also settle with the solids and are retained in this unit. Some solids are lifted by bubbles of biogas but as these bubbles rise, they expand and break leaving the solids to resettle and accumulate, creating an anaerobic sludge blanket, through which all wastewater must flow. Such a system results in a very long sludge age, allowing almost complete digestion, hence sludge removal is seldom, if ever, required. At the prototype AIWPS plant in St. Helena, California, desludging was not necessary over a 25 year operation period (Oswald, 1990; Oswald, 1988c). Algal growth in the upper layers of the PFP is supported by carbon dioxide in the biogas and the consequent photosynthetic oxygen production provides, in part, for the aerobic function of this compartment. The high oxygen content of this layer enables the entrapment and oxidation of odour causing compounds. Depth in the PFP is regulated by the level of the outlet pipe. The inlet of this pipe should be about 1m below the surface to avoid transfer of floatables into the secondary pond.

1.6.2 High Rate Algal Pond

The second unit operation of the AIWPS is the high rate algal pond (HRAP), which is a shallow, paddle-wheel mixed raceway, generating profuse algal growth and hence a high rate of photosynthesis. This tends to raise the pH of the water to above pH 9 and produces a surplus of dissolved oxygen. The high productivity in the HRAP means that a relatively short retention time is needed, generally not more than 4 days (Oswald, 1994). This is one of the advantages of the HRAP over a secondary facultative pond. Oswald (1990) recommends using some of this effluent to overlay the PFP with oxygen rich water.

1.6.3 Algal Settling and Maturation Ponds

The HRAP should be followed by an algal settling pond. A mixed pond usually selects for algae that settle when not mixed. The settling pond is, therefore, designed to be quiescent, allowing the algae to settle and the treated effluent to overflow. As the algal sludge is rich in nutrients and plant hormones, there is potential for its use as a fertiliser or animal feed (Oswald, 1991b; Maart, 1992; Potts, 1998)

If the effluent is intended for reuse, a maturation pond is generally added as the fourth pond in the series to ensure adequate pathogen removal and further BOD reduction, by allowing more time for algae to settle. A maturation pond also serves as a storage facility for irrigation and can provide a habitat for aquatic life and waterfowl.

1.7 Development of Integrated Algal Ponding Systems in South Africa

The advantages of the AIWPS for sewage treatment make it ideally suited for low cost, community development projects as is the case in South Africa and particularly the Eastern Cape. This led to a visit by Prof. Peter Rose (Rhodes University) and then by Water Research Commission (WRC) manager, Dr Oliver Hart, to the University of California, Berkley, USA, where Prof Oswald had developed the AIWPS process (Rose *et al*., 2002a). Following a visit to South Africa by Prof Oswald, in which the development of the technology in South Africa was investigated, Rhodes University Environmental Biotechnology Group (EBG) was awarded the WRC Project K5/651 "Appropriate low cost sewage treatment using the Integrated Algal High Rate Oxidation Ponding Process." As stated by Rose *et al*. (2002a), the principle objective of this initiative was the construction of a demonstration and research facility at the Rhodes University Environmental Biotechnology Experimental Field Station in Grahamstown and involving a technology transfer function with the implementation of the AIWPS process design in South Africa.

The AIWPS is a registered trade mark relating to the Oswald design and since numerous applications of the technology were to be undertaken, the more generic term, Integrated Algal Ponding System (IAPS) referring to various combinations of ponding system units, involving an algal component in their operation (Rose *et al*., 2002a), was used by the group to describe the overall research and development system.

Construction of the pilot plant commenced on 25 October, 1994 at the Grahamstown Disposal Works (GDW) and was commissioned in February, 1996 with the first raw sewage pumped into the fermentation pit on 27 February, 1996. The facility was then run by the EBG until the operation and monitoring was taken over by the Environmental Biotechnology Research Unit (EBRU) which was established at Rhodes University in 2003.

In addition to the IAPS pilot plant in Grahamstown, various other applications of this system have been investigated by Rhodes University (Rose, 2002). These include treatment of tannery (Rose *et al*., 2002b), abattoir (Rose *et al*., 2002c) as well as winery and distillery wastewaters (Rose, 2002; Dekker, 2002).

A study of the microbial ecology of the saline WSP at Mossop Western Leathers Co. in Wellington, South Africa led to the development of the *Spirulina*-HRAP, with a full scale industrial unit being built in Wellington (Rose, 2002). This unit not only produced a high quality effluent but also manufactured feed-grade *Spirulina* biomass as a by-product of the treatment process. A *Dunaliella* HRAP was used to remove organic contamination from saline carbonate brines at the Botswana Ash Co. Sua Pan, Botswana. The *Dunaliella salina* produced in these ponds is also a valuable source of β-carotene. Dekker (2002) demonstrated effective nutrient and COD removal, using an IAPS system to treat winery and wine lees wastewaters. Aspects of the IAPS system have also been incorporated in the development of the Algal Sulphate Reducing Ponding Process for Acidic and Metal Wastewater Treatment (ASPAM) and the Rhodes BioSURE Process. Both of these have shown successful reduction of sulphate and heavy metals in wastewater (Rose *et al*., 2002b; Whittington-Jones *et al*., 2002).

1.8 Research Programme

1.8.1 Background

After the initial commissioning period, Rose *et al*. (2002a) made the recommendation that the pilot plant should be operated for an appropriate period as a demonstration plant in order to facilitate the adoption of the technology into routine professional engineering practice in South Africa. Various researchers at the Rhodes University EBG have, therefore, investigated the treatment efficacy of the IAPS, over a period of nine years, since commissioning in 1996. However, until this study, no critical analysis and interpretation of the complete data set had yet been undertaken

From these previous investigations and collation of existing data, a number of questions emerged:

- Clark (2001) found that an Independent High Rate Pond (IHRAP) provided an effective "polishing" treatment for effluent from a trickle filter plant, which raised the question if a second HRAP in series would provide a superior quality effluent compared to having a single HRAP in the IAPS sequence, in which the secondary and tertiary operations were averaged?
- The IAPS with an averaged HRAP operation provided satisfactory organic and nutrient removal (Dekker, 2002) but could this function be optimised and improved with different configurations?
- What was the overall treatment performance of the IAPS, compared with standard treatment facilities and could it consistently meet South African discharge standards?

- How did the system perform under local conditions?
- As the intended use of the IAPS technology in South Africa will be in small communities, human and/or livestock contact with the effluent is highly likely, especially in cases where there is a need for recovery and reuse such as irrigation water. Assessing the disinfection capabilities of the HRAP was, therefore, critical in considering its role as a unit of the IAPS.

These queries regarding the averaged versus separated unit operations of the IAPS ultimately led to the formulation of the research hypothesis on which the findings in this report have been based.

1.8.2 Research Hypothesis

The separation of processes within the averaged IAPS and specifically the HRAP unit operations will hold precise advantages in optimising tertiary treatment processes such as nutrient removal and disinfection.

1.8.3 Research Objectives

i. Evaluate long-term performance of the IAPS under local conditions.
ii. Investigate and describe the disinfection function of the IAPS.
iii. Devise process design changes required to implement more effective disinfection operation, particularly with rural and low management applications in mind.
iv. Investigate single and series operation of the HRAP unit operations to determine optimal system configuration for highest quality effluent.
v. Formulate process design and operational criteria based on the long term study of the IAPS in Grahamstown.

CHAPTER 2

Methodology

2.1 Introduction

The IAPS, at the Rhodes University Environmental Biotechnology Experimental Field Station (Figure 2.1) was used as the experimental system for the study to be reported here. Although the Grahamstown IAPS was intended as a demonstration unit, it was sized to provide credible performance data, suitable for engineering scale-up requirements. Design rationale and calculations were provided by Prof William Oswald and Dr. Bailey Green, consulting as Oswald Green and were used as the basis for the conceptual plan for the construction of the plant (Rose *et al.*, 2002a).

Figure 2.1: The integrated algal ponding system plant at the Experimental Field Station, showing the primary facultative pond (top) and high rate algal ponds, algal settling ponds and drying beds below.

2.2 The Integrated Algal Ponding Systems Pilot Plant Process Design

The system was built to have the capacity to treat the liquid wastes of 500 person equivalents (PE). An average water consumption and disposal per capita of approximately 150l.day^{-1} was assumed. Accordingly the design flow was calculated at 75m^3.day^{-1}. With an ultimate Biochemical Oxygen Demand (BOD_{ult}) assumed to be 80g BOD_{ult} per person per day, the organic loading to the system is 40kg.day^{-1}.

Apart from preventing possible intrusion of oxygen into the anaerobic layer of the PFP, the idea of a fermentation pit in the base of the PFP, is 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 0.45m^3 per capita, which is 15 times the standard per capita (0.03m^3) value used in conventional sewage sludge digesters. The volume of the fermentation pit is therefore 225m^3, giving a hydraulic retention time (HRT) of 3 days. A perimeter berm or wall was also built around the pit to further preclude wind-induced mixing and therefore oxygen from the fermentation zone. The depth of the pit is 4.5m, 3m below the bottom of the PFP and 1.5m 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 2m per day. A daily flow of 75m^3 over the pit area of 50m^2 results in an upflow velocity of 1.5m^3.day^{-1}. This adds a 25% margin of safety over the recommended 2m.day^{-1}.

The primary facultative pond (PFP), depicted in Figure 2.2, surrounds the fermentation pit and has a volume of 1500m^3, to give a maximum HRT of 20 days. The surface area of this pond is 840m^2, which, assuming a conservative BOD removal in the fermentation pit of 50%, will give an organic loading of 0.024kgBODult/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 1m below the surface to avoid depletion of the surface algae layer and carry over of floating solids.

Figure 2.2: The primary facultative pond

Using an assumed conservative BOD loading to the HRAP from the PFP, the depth in the two HRAPs was set at 30cm. The total volume of these ponds is, therefore 150m^3, with a surface area of 500m^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 75m^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 30cm.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 Rhodes University 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. 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 against the outside wall moving faster than against the dividing wall. This zone of reduced flow velocity results in settlement of algae floc on the bottom of the raceway. 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 2.3.

Figure 2.3: High rate algal ponds, depicting channel dividers.

The algae settling ponds, shown in Figure 2.4, were designed to provide half a 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 between 50 and 80% of the algae settling and a slurry concentration of 3-5% solids. Taking a 3% slurry concentration, the maximum daily volume would be 0.25m^3. At a depth of 5cm on a sand bed surface, the daily area required for drying the slurry would be 5m^2. If a 7 day drying period is allowed for, the drying beds need to be 35m^2. Another estimation used for sizing drying beds is 10% of the total HRAP area, which would give a sand bed area of 50m^2. To compensate for these different size calculations, four 10m^2 drying beds (Figure 2.5) were built to give a total area of 40m^2.

Figure 2.4: Algal settling pond

Figure 2.5: The four algal drying beds between the two algal settling ponds.

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 demonstration plant was operated with a constant inflow to the PFP. This was achieved by pumping raw sewage from behind the Grahamstown Disposal Works' (GDW) head of works. Figure 2.6 gives the orientation of the IAPS and GDW. As the pump delivers approximately $165m^3.day^{-1}$, the excess wastewater, i.e. over and above the $75m^3$ 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 the flow to the PFP.

Figure 2.6: The integrated algal ponding system at left, in relation to the Grahamstown Disposal Works at right.

2.3 Operation of the Integrated Algal Ponding System

Operation of the system varied during the course of the study period depending on the specific research objectives under investigation. The hydraulic loading to the fermentation pit and PFP remained constant at $75m^3.day^{-1}$. 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 the HRAPs. During commissioning and the first 4 years of operation these two ponds were operated in parallel, each unit taking half of the PFP effluent i.e. $37.5m^3.day^{-1}$ (Figure 2.7, A). This equates to a 4 day hydraulic retention time (HRT).

In February 2000, the system was reconfigured and one HRAP was retrofitted to investigate the efficacy of the process as an independent unit operation, used as a tertiary treatment stage, polishing final effluent from a conventional sewage treatment facility. This application became known as the Independent High Rate Pond (IHRAP), with the final treated water being sourced from the GDW (Clark, 2001). A HRT of 5 days was used for this study (Figure 2.7, B). During this time the PFP effluent was split so that HRAP1 continued to operate as it did during the parallel, averaged configuration, i.e. receiving its design load. The excess flow was wasted to drain. The ponds were operated in this manner until June 2003, when the IHRAP treating GDW final effluent was discontinued.

In June 2003 HRAP2 was reincorporated into the IAPS cascade but configured to operate in series with HRAP1 (Figure 2.7, C) rather than in parallel as in the earlier phases of the project. The performance of the second HRAP as a polishing unit, receiving effluent from the first, after settling algae in the ASP, was thus evaluated. Retention times during this last period were varied between 3 and 6 days.

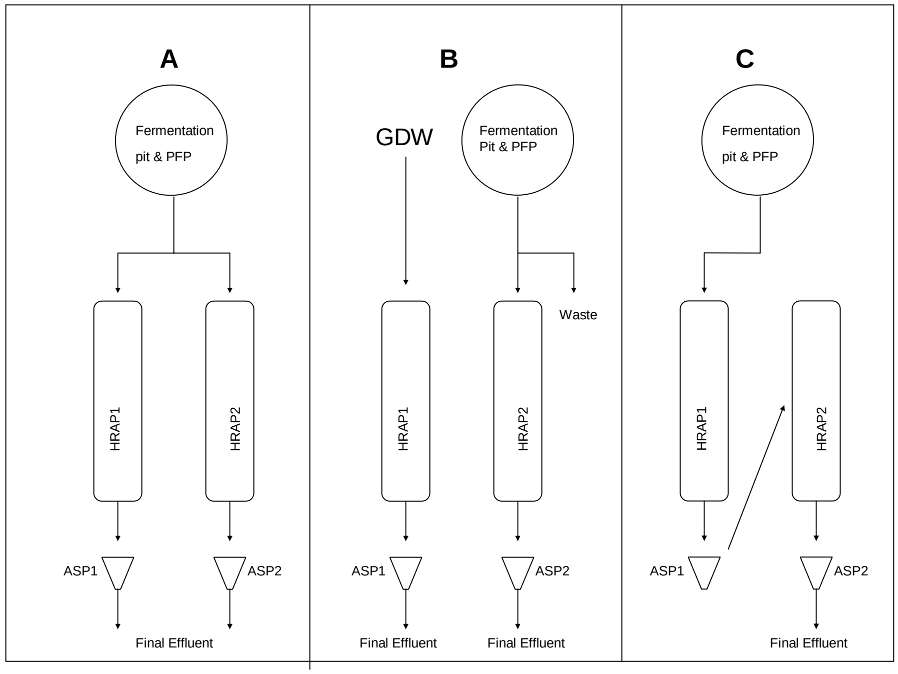

Figure 2.7: Different configurations of the HRAP unit operation discussed in this study.

2.4 Analytical Methods

COD, ammonium as NH_3-N, nitrate as NO_3-N and orthophosphate as PO_4-P were analysed using a Merck Spectroquant SQ118 and Merck analysis kits. Samples were filtered through Whatman GF/C microglass filters. The pH was taken 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.

2.5 Microbiological Analysis

Bacteriological analysis was carried out using the filtration method described in Standard Methods (A.P.H.A. 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 distinction of faecal coliform colonies. Using Chromocult agar (Merck), the enumeration of *Escherichia coli* and total coliform colonies was, however, successful and this method was, therefore, adopted for the analysis of indicator bacteria.

2.6 Sampling Protocols

Samples were taken from the incoming, raw sewage and then 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 24h.

2.7 Data Processing

The data to be described in the following chapters has been presented from several angles, depending on which aspect of the system was under investigation. Chapter three deals with the fermentation pit and PFP. 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.

Chapter four details the performance of the HRAP, during the period when this unit was run as a single stage treatment with the performance averaged between the two high rate ponds. This chapter also gives an account of the performance of the IHRAP, treating GDW final effluent.

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 chapter five.

Chapters three to five 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 six in such a way as to describe the performance of the IAPS as an alternative, wastewater management technology.

CHAPTER 3

Fermentation Pit and Primary Facultative Pond

3.1 Introduction

The PFP in the IAPS, is designed to promote three distinct microbial consortia i.e. a deep anaerobic stratum, overlain by facultative strata and finally by an aerobic surface layer. There are many contaminants of concern in domestic wastewater, however, the organic load, and therefore oxygen demand, is a distinct characteristic of sewage that requires attention in any treatment plant. According to Green *et al*. (1995b), methane fermentation is one of the most efficient ways to manage this organic load and 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 to decrease pathogens and nutrients in the PFP, the primary function of this unit operation remains COD removal.

Although different researchers have investigated various aspects of the IAPS over the study period, the operation of these units remained consistent. The results presented in this chapter are, therefore, a collation of all the data recorded by both the EBG and EBRU during the nine year monitoring programme.

3.2 Material and Methods

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

3.3 Results and Discussion

3.3.1 Chemical Oxygen Demand Removal

Due to a large unfiltered COD (CODt) variation in the sewage feed (raw), between 1997 and 1999, the results since commissioning in 1996 have been presented as two separate data sets: from commissioning until October 1996 in Figure 3.2 and 1999 until October 2004 in Figure 3.3. The results for the earlier period indicate an average Raw CODt of 2 340mg.l^{-1} while the average since the beginning of 1999 is 1 013mg.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.

The performance of the PFP during commissioning is discussed in more detail by Rose *et al*. (2002a) 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 3.1 shows the reduction of VFAs and subsequent consistency of these levels after about week 40. Despite the variable performance of the fermentation pit, a COD_t removal of between 60 and 85% was consistently achieved in the PFP during the first 50 weeks of operation (Rose *et al*., 2002a).

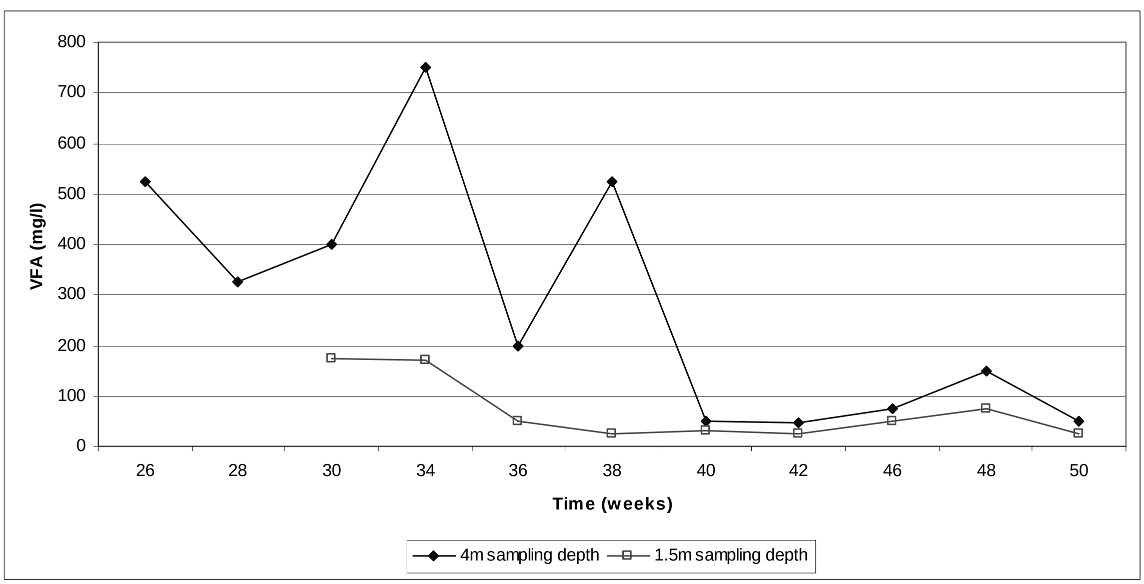

Figure 3.1 Stabilisation of volatile fatty acid levels in the fermentation pit, during commissioning, after reseeding with anaerobic sludge from the Grahamstown Disposal Works.

From Table 3.1, it is evident that, at times, the PFP effluent had high COD_t levels. The average COD_t removal during this time was, however, 68.4%. As was mentioned previously, 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 been correlated with incidents of high rainfall, during which storm-water enters the Grahamstown sewage system (Dekker, 2002). The dilution effect of rainwater on sewage has been identified in other studies e.g. Schetrite and Racault (1995).

Table 3.1: Primary facultative pond unfiltered chemical oxygen demand removal from February 1996 to October 1997.

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

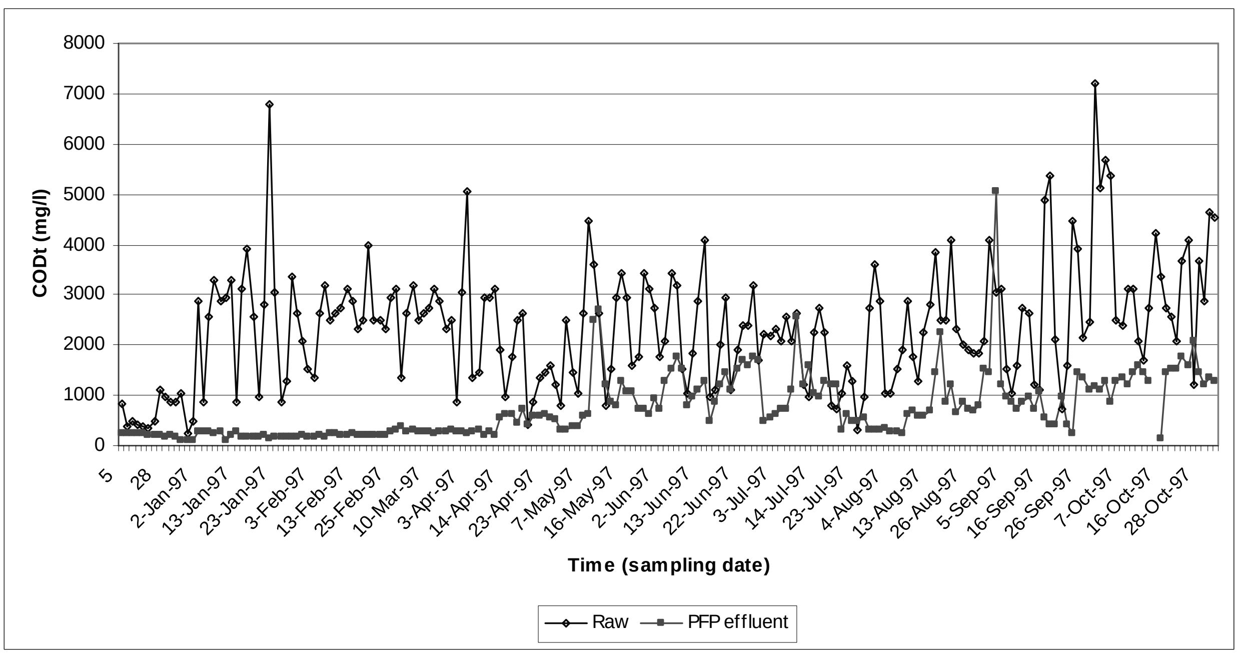

Figure 3.2: Unfiltered chemical oxygen demand removal in the primary facultative pond from commissioning until October 1997.

Although the high COD_t levels of 1997 are not repeated in the later data set, summarised in Table 3.2, there is still a wide COD_t variation in the sewage feed, with values fluctuating between 312mg.l^{-1} and 6 535mg.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 works from collection sumps in Grahamstown. The mean COD_t load of 1 162mg.l^{-1} in the feed is comparable with levels measured in other studies in Africa (Horan, 1996) but is about double that found in areas of higher water consumption such as 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 500mg.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 1600mg.l^{-1}.

The mean percentage COD_t removal 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 3.2: 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 3.3; from the end of June 2004, high levels of up to 6 500mg.l^{-1} COD_t were measured in the sewage feed. These high levels did not appear to significantly affect the COD_t in the PFP effluent, which averaged 326mg.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 are also not reflected in the PFP effluent.

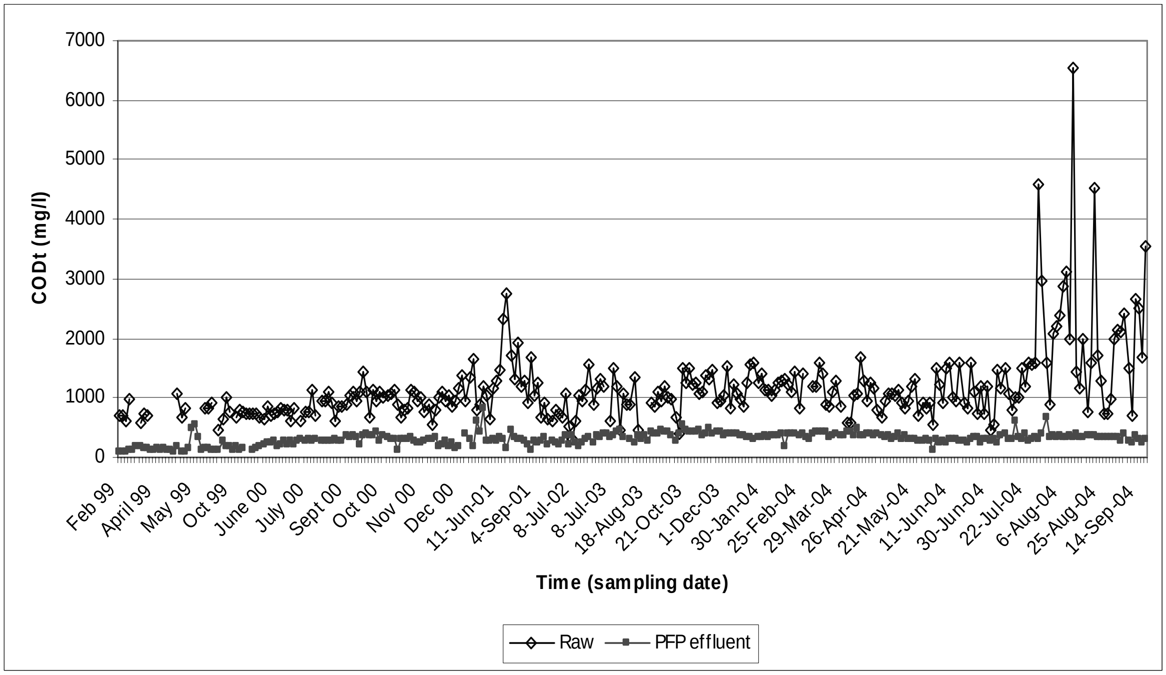

Figure 3.3: Unfiltered chemical oxygen demand removal in the primary facultative pond between February 1999 and October 2004

Removal figures for filtered or soluble COD (COD_s) (Figure 3.4) are not as impressive as those for total COD. During commissioning and the monitoring period up to October 1997, the ponds were not analysed for filtered COD. Data, therefore, only exists 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 total COD being in the form of suspended solids, which are settled out in the fermentation pit and thus removed. This fraction would, however, be excluded by the 45µm filters used in the measurement of soluble COD and the removal rate is, therefore, not replicated in the COD_s figures.

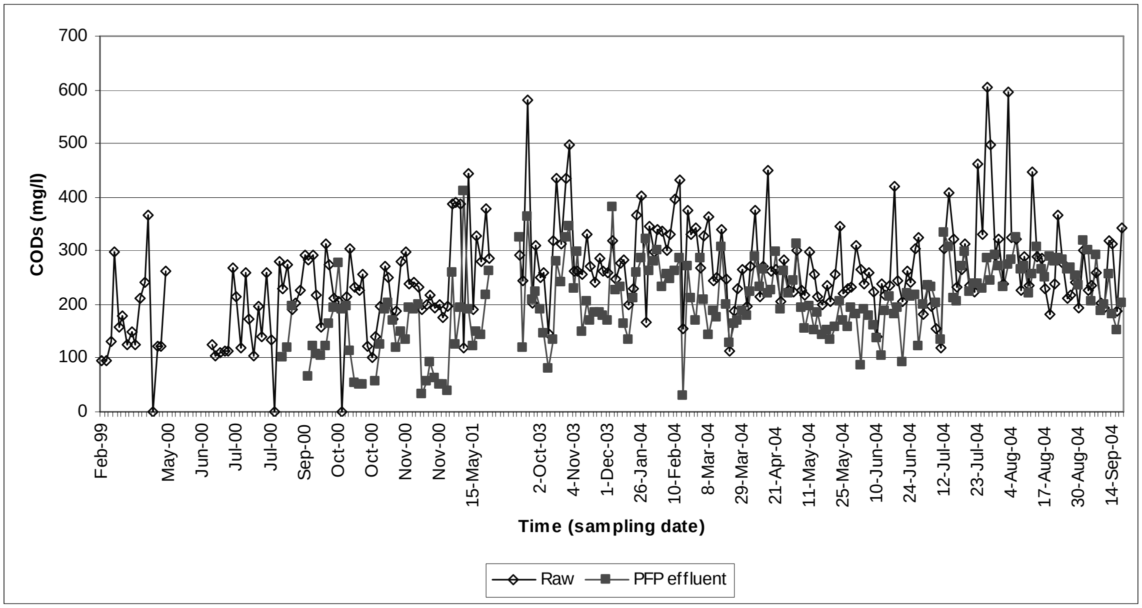

Figure 3.4: Filtered (soluble) chemical oxygen demand removal in the primary facultative pond for the period February 1999 to October 2004

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 necessity. After nine years of operation, the Grahamstown demonstration plant has a non-degradeable sludge build up of an average 300mm in the fermentation pit.

Methanogens are very sensitive to environmental conditions, in particular to changes in pH. They are only tolerant 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, releasing objectionable odours (Oswald, 1994). Apart from a brief drop in pH during commissioning, the pH of the pit and the PFP remained stable, within the optimal methanogen range. The average pH of the PFP effluent over the monitoring period was pH 7.7. Figure 3.5 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.

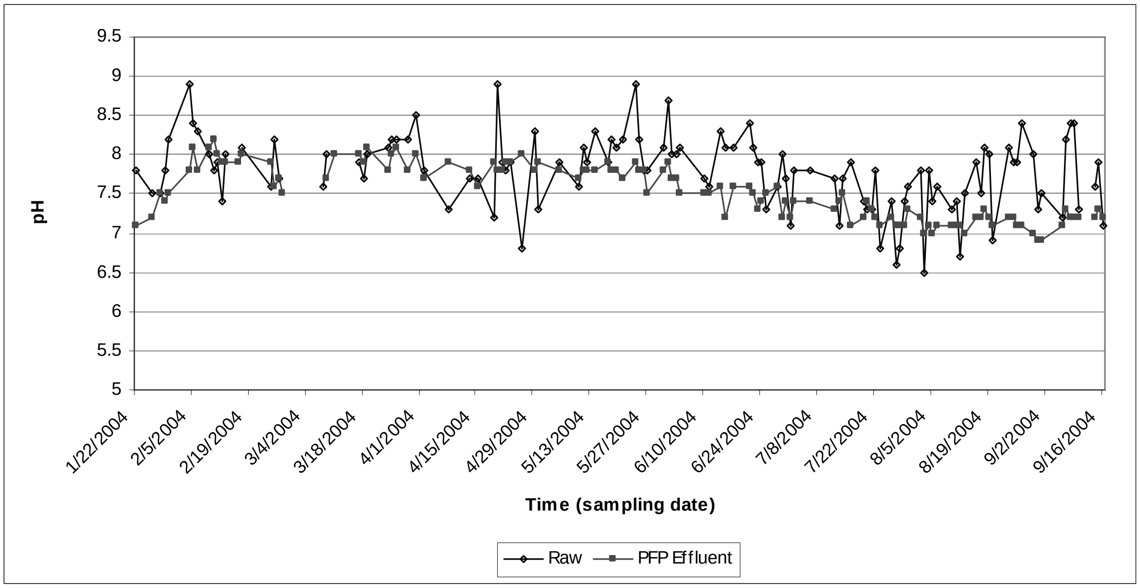

Figure 3.5: Comparison of raw and primary facultative pond effluent pH over the sampling period January to September 2004

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 in biomass (Henze *et al*., 2002). This is similar to any other aerobic treatment except the oxygen is supplied by algae photosynthesis and not by mechanical means (Horan, 1996).

3.3.2 Nutrient Removal

The principle function of the PFP is 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 3.3 and figures 3.6 to 3.9 present a comparison of the nutrients present in the sewage feed (raw) and PFP effluent.

Table 3.3: 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

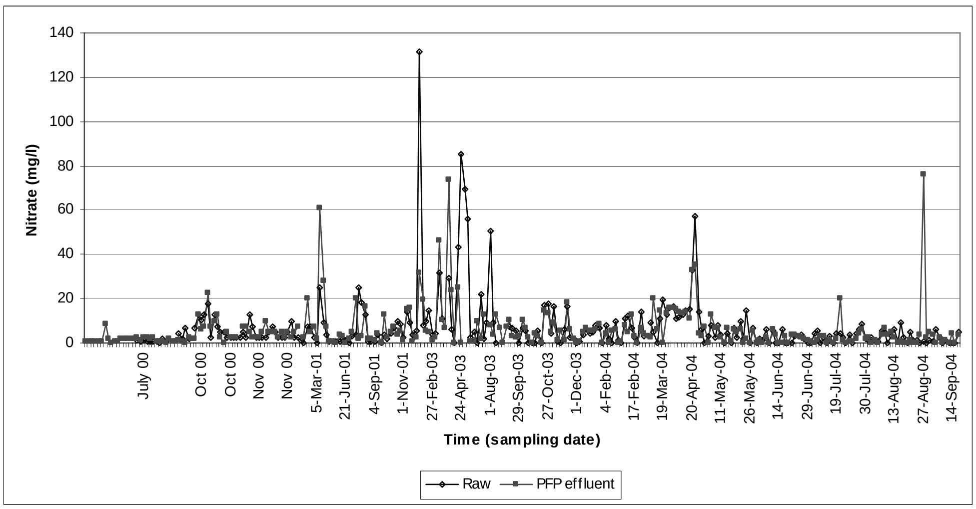

Figure 3.6: Comparison of nitrogen as $mg.l^{-1}$ NO_3^{-} in the sewage feed (raw) and primary facultative pond effluent

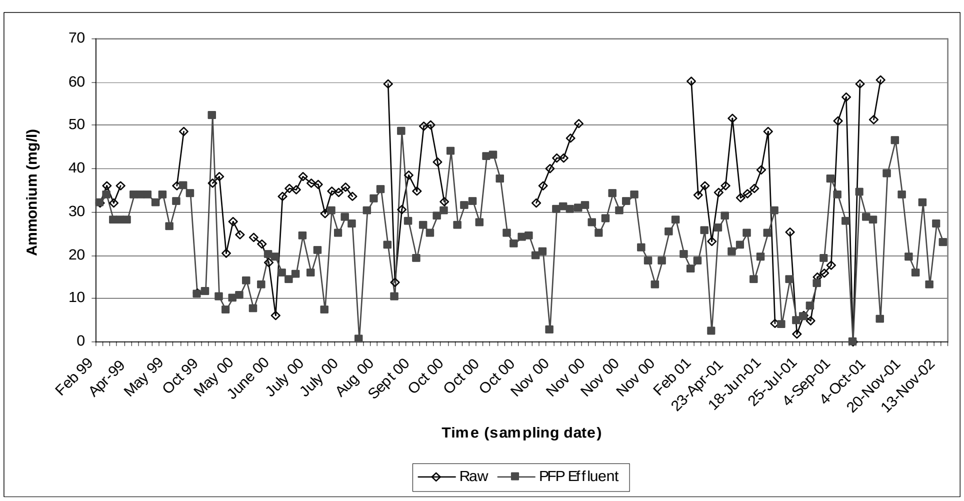

Figure 3.7: Comparison of nitrogen as $mg.l^{-1}$ NH_4^{+} in the sewage feed (raw) and primary facultative pond effluent during the period February 1999 to November 2002.

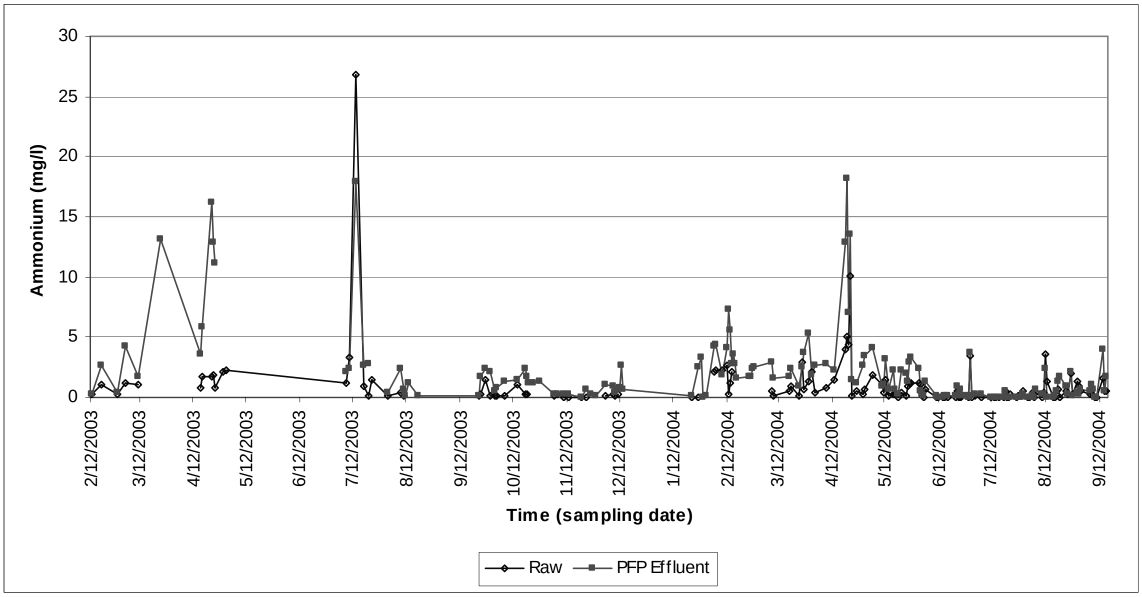

Figure 3.8: Comparison of nitrogen as mg.l^{-1} NH_4^+ in the sewage feed (raw) and primary facultative pond effluent during the period February 2003 to September 2004.

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 ammonium 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 3.8), 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.

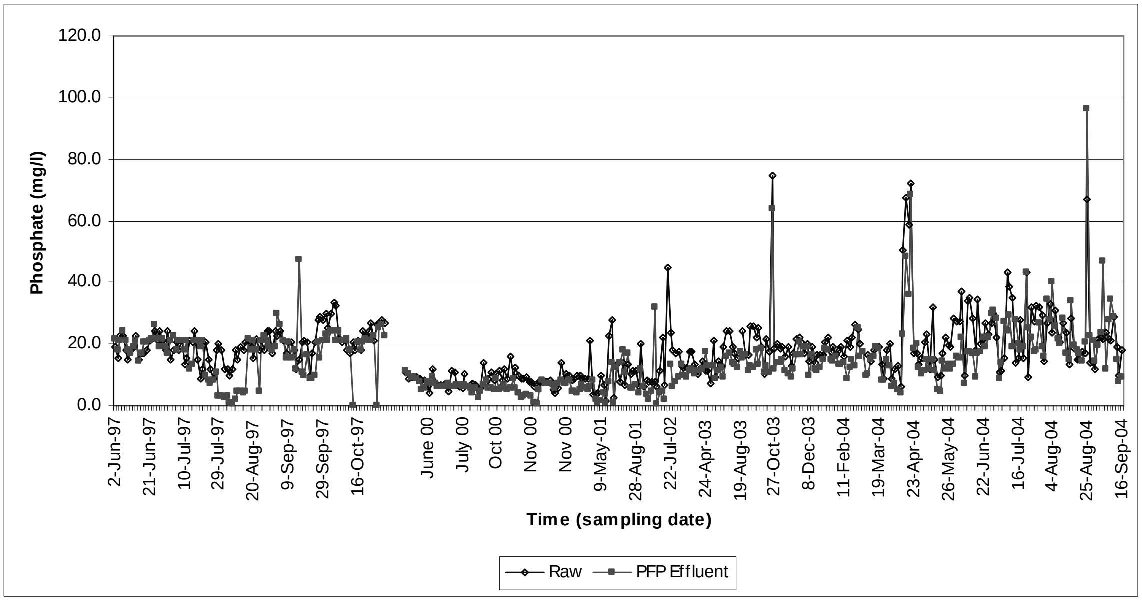

Figure 3.9: Comparison of phosphorous as mg.l^{-1} PO_4^{3-} in the sewage feed (raw) and primary facultative pond effluent.

According to the mean figures, represented in Table 3.3, an 18% decrease in phosphate across the PFP was observed. From Figure 3.9, however, it can be seen that this decrease was 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 any significant elimination taking place.

3.3.2.1 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 ammonium. This was probably due to volatilisation of the ammonia as Zimmo *et al*. (2003), have reported higher ammonia volatilisation rates when influent ammonium was raised. Maynard *et al*. (1999) have claimed that a 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 ammonium was possibly also removed by algae and other microbe assimilation in the PFP (Horan, 1996). The period where there was a slight increase in ammonium 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 bacteria 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 treatment stage.

3.4 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 000mg.l^{-1} were absorbed in the PFP, while CODt in the effluent remained below 400mg.l^{-1}.
- A 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.

CHAPTER 4

High Rate Algal Ponds

Over the period of the IAPS studies in Grahamstown, a number of configuration changes were made to the HRAP unit operation. These different configurations are illustrated in Figure 2.6 and include A) the averaged, parallel operation of the two HRAP, B) the operation of the IHRAP treating GDW final water and C) the operation of the a second HRAP in series, effectively functioning as an IHRAP, after a conventional IAPS.

The HRAP process had been an element of the IAPS since the project was initiated in 1996, however, no long term performance evaluation of this unit operation had been reported. The collation and interpretation of the data collected over the entire monitoring period therefore formed a component of this study. Data presented in this chapter represents the results of the HRAP unit operation since commissioning of the plant in 1996 to May 2003, including both the averaged, parallel operation of the two HRAP as well as the IHRAP treating GDW final effluent, but excludes the data from the two stage HRAP process, which is reported in chapter 5. Although there was a configuration change into a series operation of the two HRAP from June 2003 until December 2004, data from HRAP1 during this period has been included here. This is because, unlike HRAP2, its position and function in the IAPS effectively remained unchanged in either configuration.

4.1 Introduction

The component processes occurring in the HRAP are summarised in Figure 4.1. 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, respectively.

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, 1988c). CO_2 produced by the 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 (Rose *et al*., 2002a).

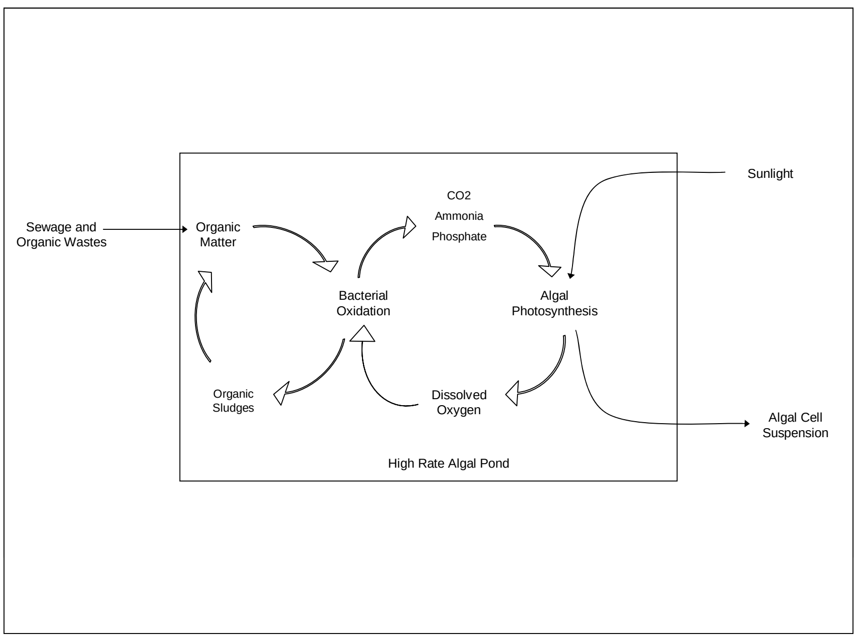

Figure 4.1: Cycle for photosynthetic oxygenation of wastewater. After Oswald (1988c)

4.1.1 Nutrient Removal Mechanisms in the High Rate Algal Pond

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

$$NH_4^+ + O_2 + 2H^+ \rightarrow NH_2OH + H_2O \rightarrow NO- + 5H^+ \tag{4}$$

$$NO_2^- + 2O_2 \rightarrow 2NO_3^- \tag{5}$$

Many biological systems remove nitrate by exploiting denitrification, where nitrate is microbially reduced, through various inorganic gaseous forms to N_2 that is then released to the atmosphere (Maier *et al*., 2000). Cromar *et al*. (1996), however, claim that nitrate bacteria are susceptible to high irradiance and high temperatures, both typical of a HRAP, and 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 (Fogg, 1989) 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 that effluent from algae, left in settling ponds, showed an increase in orthophosphate, possibly due to the release from the dead portion of the algae culture.

The other likely removal mechanism is precipitation at high pH in calcium rich waters (Moutin *et al*., 1992; Mesplé *et al*., 1995). This is known to occur at pH greater than pH8.5 (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 coprecipitation of phosphate on calcium carbonate (calcite) (House, 1990) according to the equation:

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

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:

$$5Ca^{2+} + 3PO4^{3-} + OH^- = Ca_5(PO_4)_3OH \text{ (solid)} \quad (7)$$

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

4.2 Materials and Methods

The materials and methods are as described in Chapter 2.

4.3 Results and Discussion

4.3.1 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 3. Due to this variation, the data has been separated into two periods: January to October 1997 (Figure 4.2) and February 1999 to September 2004 (Figure 4.3).

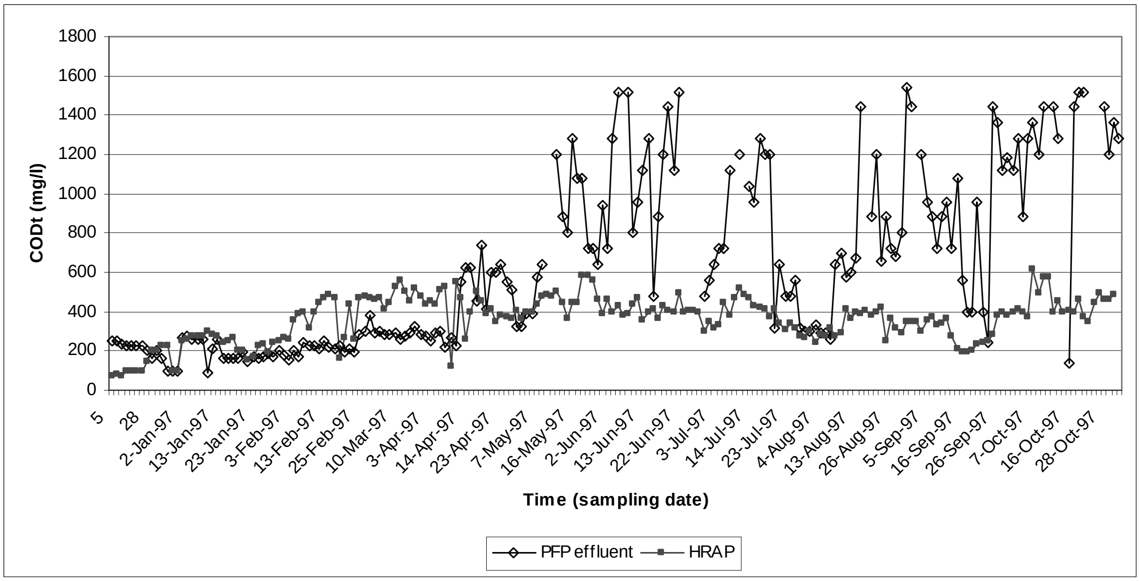

Figure 4.2: Unfiltered chemical oxygen demand removal in the first high rate algal pond, during 1997.

From Figure 4.2, 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 appreciated. 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 500mg.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 4.1 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 308mg.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 4.1: 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 4.3 depicts 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 on 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.

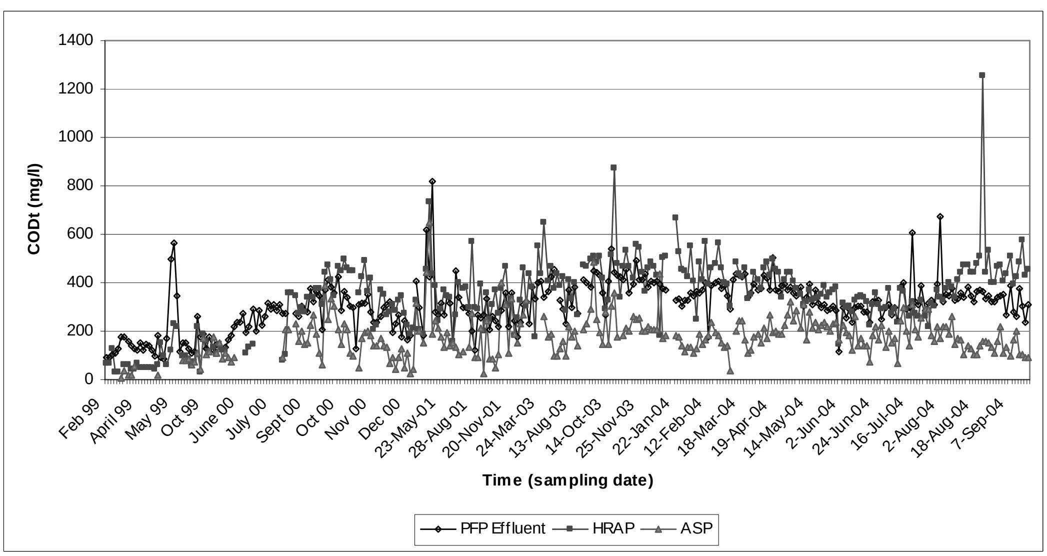

Figure 4.3: Unfiltered chemical oxygen demand removal in high rate algal pond 1 and algal settling pond 1 for the period February 1999 to September 2004.

Analysis of the soluble COD, 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 on the figure).

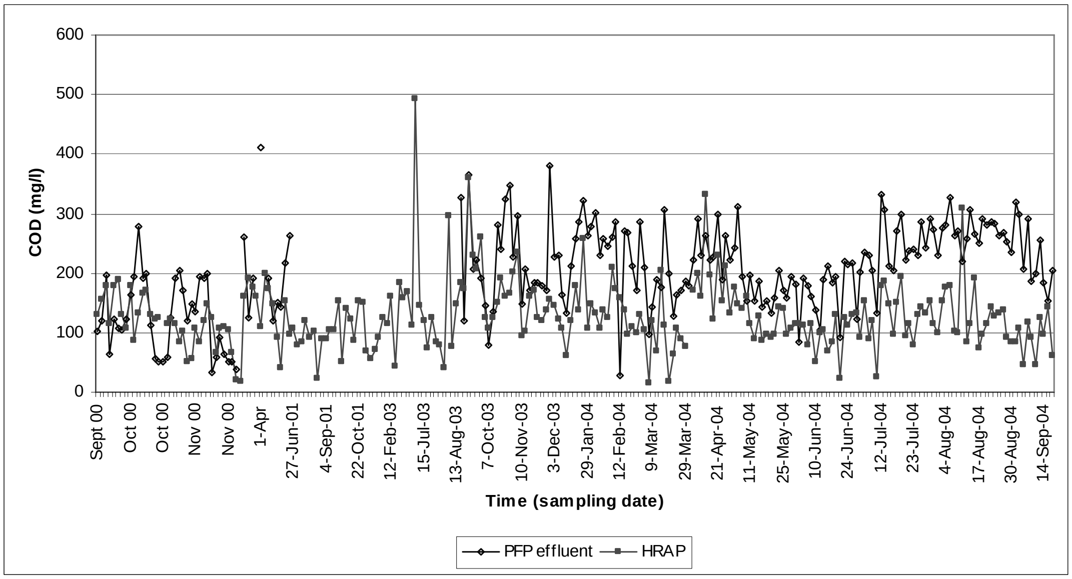

Figure 4.4: Soluble chemical oxygen demand removal in the high rate algal pond for the period September 2000 to September 2004.

Because there were a number of different researchers investigating various aspects of the IAPS, there were periods between specific studies when operation 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. With the commencement of this project, the system underwent vigorous 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 has both positive and negative implications; firstly, it indicates the system has limitations in terms of COD removal and, although the 75mg.l^{-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 made 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).

The slight increase in COD in the HRAP can be attributed to algal growth. This insoluble organic content is then effectively removed by the settling out of algal biomass in the ASP, hence the similarity between the total COD in the ASP effluent and the soluble COD in the HRAP.

4.3.2 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 nitrogen removal was less consistent. The following figures illustrate the nutrient removal performance of the HRAP.

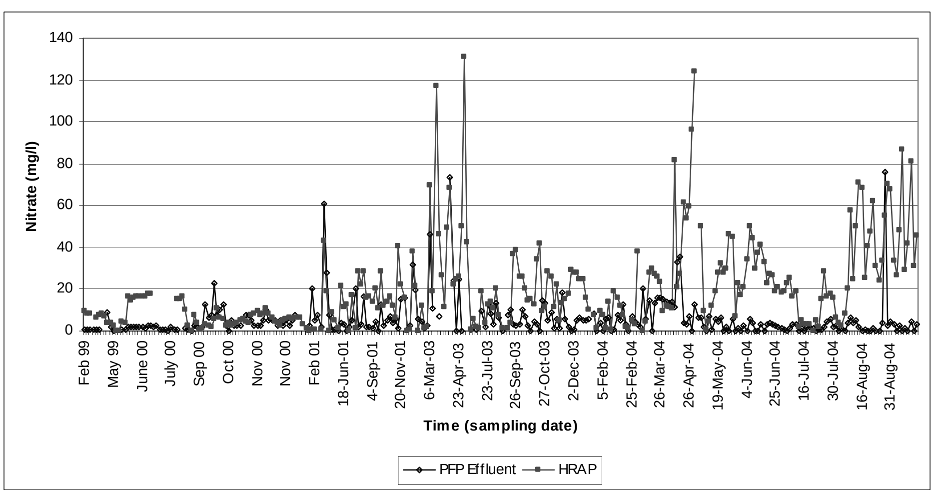

Figure 4.5: Nitrate in the primary facultative pond effluent and high rate algal pond for the period February 1999 to August 2004

During most of the period depicted in Figure 4.5, it appears as though there was an increase in nitrates in the HRAP. 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 Grahamstown plant, 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, there were relatively high concentrations of ammonium in the PFP effluent. This was effectively removed in the HRAP (Figure 4.6). During 2003 and 2004, these ammonium levels dropped, corresponding to periods of high ammonium in the HRAP (Figure 4.7).

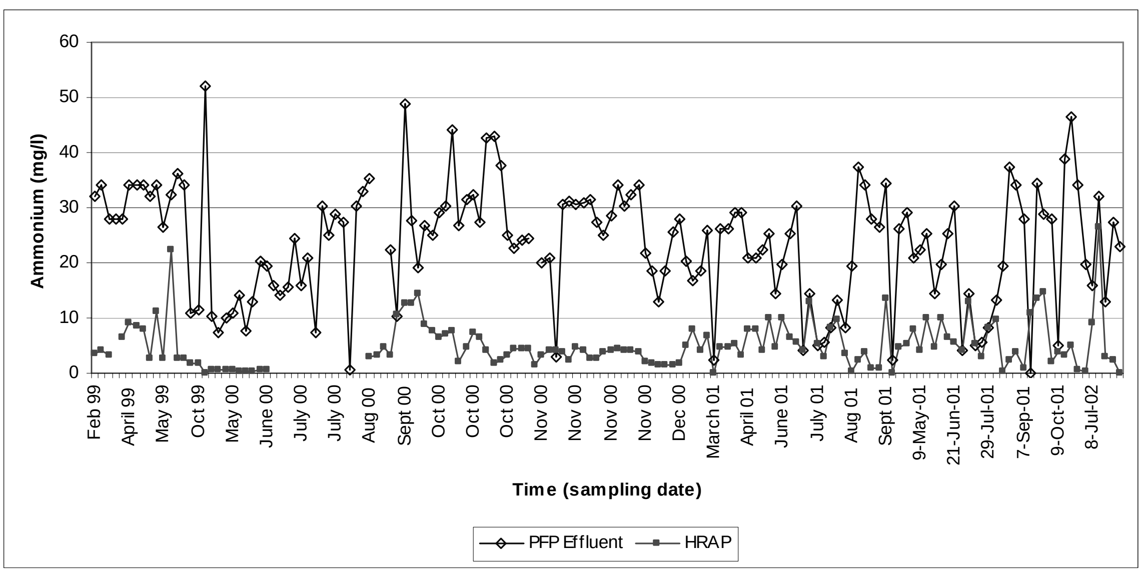

Figure 4.6: Ammonia removal in the high rate algal pond for the period February 1999 to July 2002**.**

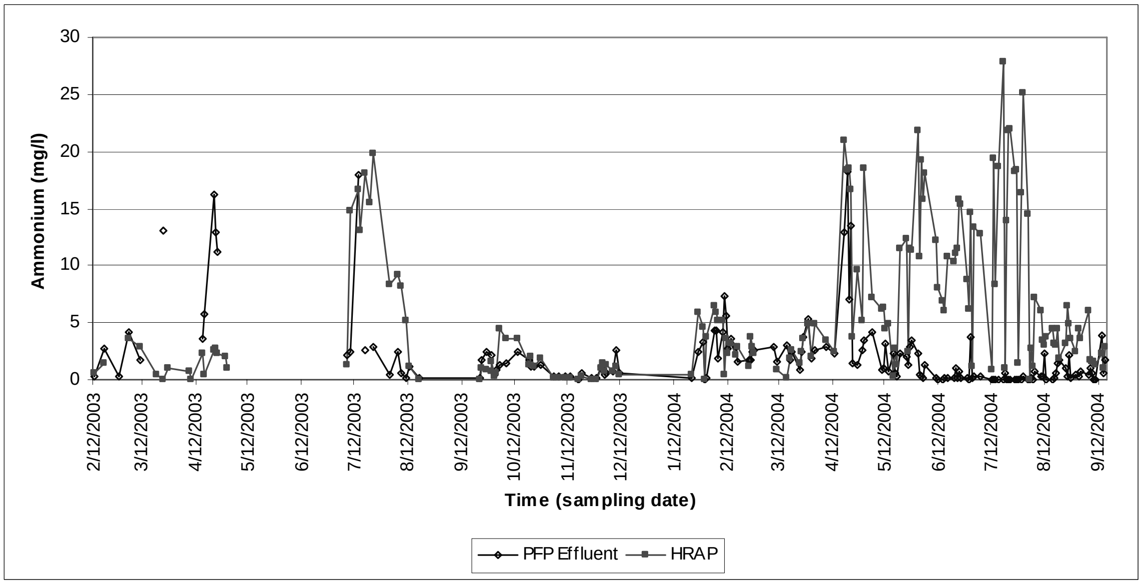

Figure 4.7: Increase in ammonium in high rate algal pond 1 during 2003 and 2004.

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 possibly explanation.

Organic nitrogen in the PFP effluent is broken down, releasing ammonium, evidenced by the increase shown in Figure 4.7. Some of this ammonium 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 run on the HRAP, where the organic load to the HRAP was increased for a ten day period. During this time the nitrate levels in the effluent decreased considerably (Figure 4.8). 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.

$$NO_3^- + 5H_2 + 2H^+ \rightarrow N_2 + 6H_2O \qquad (8)$$

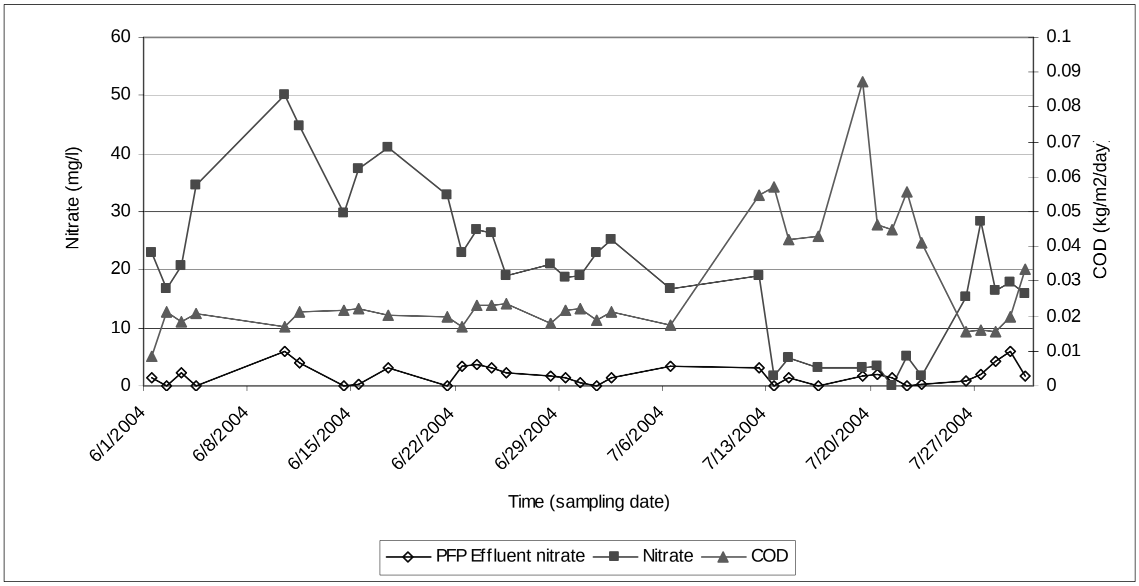

Figure 4.8: Nitrate in the high rate algal pond plotted against chemical oxygen demand load to the high rate algal pond, illustrating a considerable drop in nitrate levels at high organic loading.

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 ammonium 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 4.9 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.* (2002a), 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.

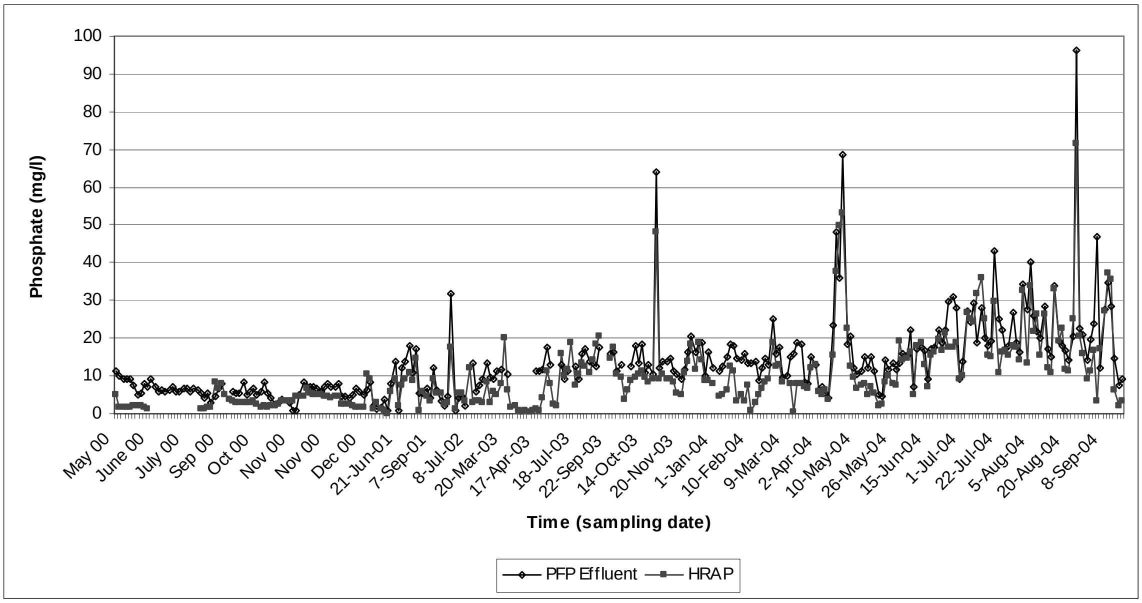

Figure 4.9: Phosphate in the primary facultative pond effluent and high rate algal pond.

4.3.3 Independent High Rate Algal Pond

As part of the EBG's investigations, a study was conducted on the efficacy of the IHRAP in treating final effluent from a conventional wastewater treatment works, in this case the GDW trickle filters. The main focus of this research was nutrient removal in the IHRAP. In reviewing these studies, some of the findings, briefly discussed here, became apparent. The results are, however, more comprehensively reported by Clark (2001).

The IHRAP was monitored over a period of 45 days to determine the orthophosphate reduction. The GDW final effluent contained phosphate levels up to 16mg.l^{-1}. This was effectively removed by the IHRAP, with a mean reduction of approximately 80%, from 8.3 ± 2.9 mg.l^{-1} to 2.0 ± 1.3 mg.l^{-1} PO_4-P (Clark, 2001). Ammonium was also successfully removed from the GDW final water with a 79% mean reduction rate, from 6.5 ± 4.1 mg.l^{-1} to 1.35 ± 1.1 mg.l^{-1} NH_4-N. Clark (2001) found that the IHRAP performed poorly with regard to nitrate removal. Effluent nitrate levels were often still as high as 25mg.l^{-1} NO_3-N (Clark, 2001).

Based on the IHRAP findings, the need for a subsequent denitrification step in line with the IHRAP was identified and a laboratory scale investigation was undertaken by Clark (2001). Clark (2001) used two types of denitrification reactors, an Upflow Anaerobic Sludge Blanket (UASB) and another type designed as a Drowned Anaerobic Trickle Filter (ANTRIC). Drawing on the experience of the successful use of UASB-type reactors in wastewater treatment, a novel design in which algal biomass would be used as the sludge packing material in a submerged environment was proposed (Clark, 2001). Clark (2001) reasoned that the algae produced at a high rate in the IHRAP can then be used as a sludge packing material through which wastewater can be passed to remove organic nutrients and other unwanted material.

A build up of degraded biomass in the UASB-type reactor caused inhibition of denitrification due to substance toxicity or a decrease in available carbon (Clark, 2001). On the other hand the stress manipulation of metabolically active algal cells supplied sufficient energy to drive denitrification in the ANTRIC reactor (Clark, 2001).

Because the ANTRIC process was observed to perform better than the UASB-type, this reactor was selected for scale up to test in a pilot plant, given the term Algal Tertiary Treatment (ATT) bioreactor (Neba, 2003). The ATT was operated with algal biomass supplied by the IHRAP and wastewater sourced from GDW final effluent. The concept behind the drowned trickle filter was firstly that, unlike conventional trickle filters, oxygen is excluded and the ATT columns were, therefore, allowed to go anaerobic. The wastewater was then pumped in at the head of the reactor and percolated through the algae packing material under pressure, resulting in the development of an algal-bacteria biofilm, which should enhance denitrification (Clark, 2001).

The pilot ATT showed good nitrate removal with an average removal of 80.56% and peaks of up to 90mg.l^{-1} reduced to less than 5mg.l^{-1} (Neba, 2003). The best performance was achieved approximately 60 days after start up, presumably because that was when all the oxygen was utilized and the denitrifying bacteria switched to nitrate as a final electron acceptor for cellular respiration. Although the nitrates were successfully removed, there was unfortunately a large amount of ammonia generated, with up to a 92.9% increase and peak levels in the effluent of 600mg.l^{-1} (Neba, 2003). This was attributed to the degradation of algal biomass by the bacteria consortium. Neba (2003) also reported an 85.7% increase in phosphates, thought to be caused by a VFA induced drop in pH, resulting in dissociation of the phosphate precipitate. Neba (2003) suggested directing the ATT effluent into the IHRAP to remove the generated phosphate and ammonium. However, this could again result in nitrate production from the nitrification of ammonium in the aerobic environment of the IHRAP.

Also of interest in the study conducted by Neba (2003), it was found that the CODs in the ATT effluent averaged 63.3mg.l^{-1}, which is better than the CODs levels recorded in the effluent of both the IHRAP and IAPS HRAP.

4.3.4 Integrated Algal Ponding System Algae

Alga is the Latin word for seaweed; the term algae is, however, not a formal taxonomic grouping but refers to a vast array of primitive, 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 70m (Chapman & Gellenbeck, 1989). Algal forms important in wastewater ponds are mostly from the classes Chlorophyta and Euglenophyta, which includes non-motile and flagellate green algae (Gloyna & Tischler, 1979; Mara & Pearson, 1986). Other, less dominant, phytoplankton include genera from the Cyanophyta (blue-green algae/cyanobacteria) and Chrysophyta (Mara & Pearson, 1986). Although the Cyanophyta are prokaryotic, which allies them with bacteria, while all the other algae are eukaryotic; 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 algae types flourish in conditions of varying water quality, knowledge of the algae 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. (Figure 4.12) together with *Actinastrum* sp. (Figure 4.13) were prevalent during winter, while in summer, *Pediastrum* sp. (Figure 4.10 and 4.11) 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 (Figure 4.14) 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).

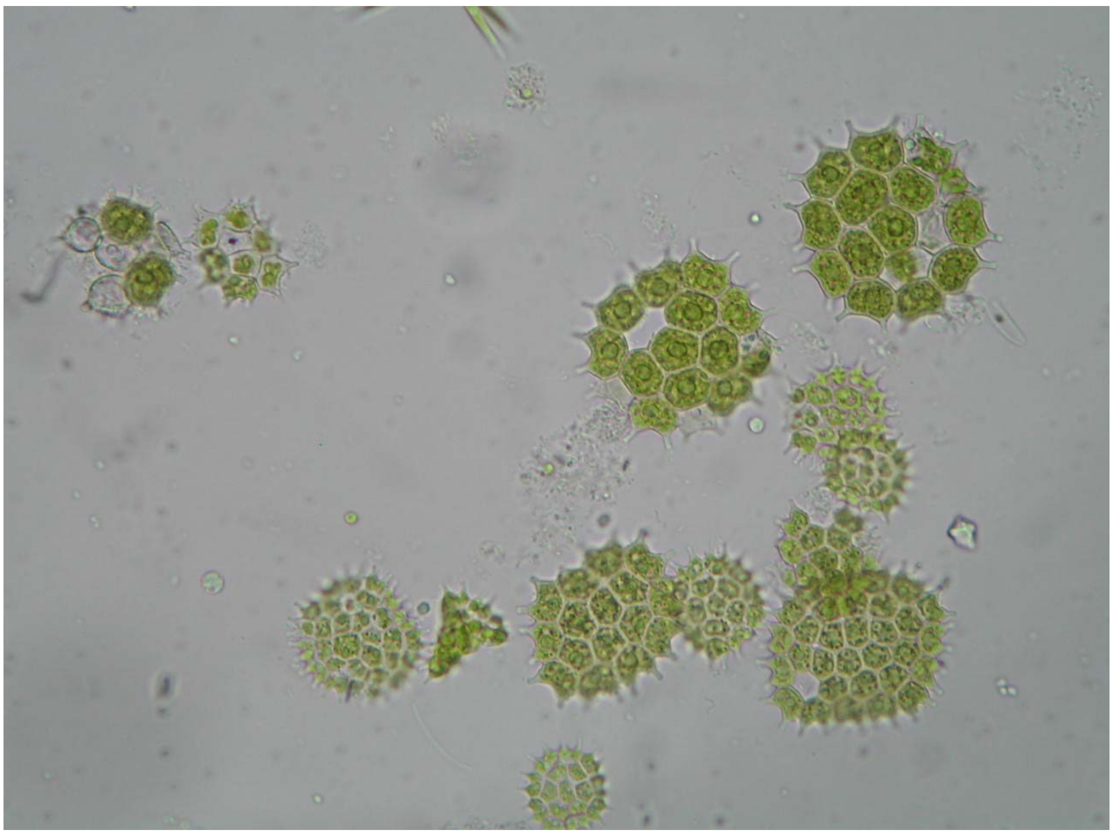

Figure 4.10: *Pediastrum* sp. was observed to be dominant in the second high rate algal pond during summer operating conditions.

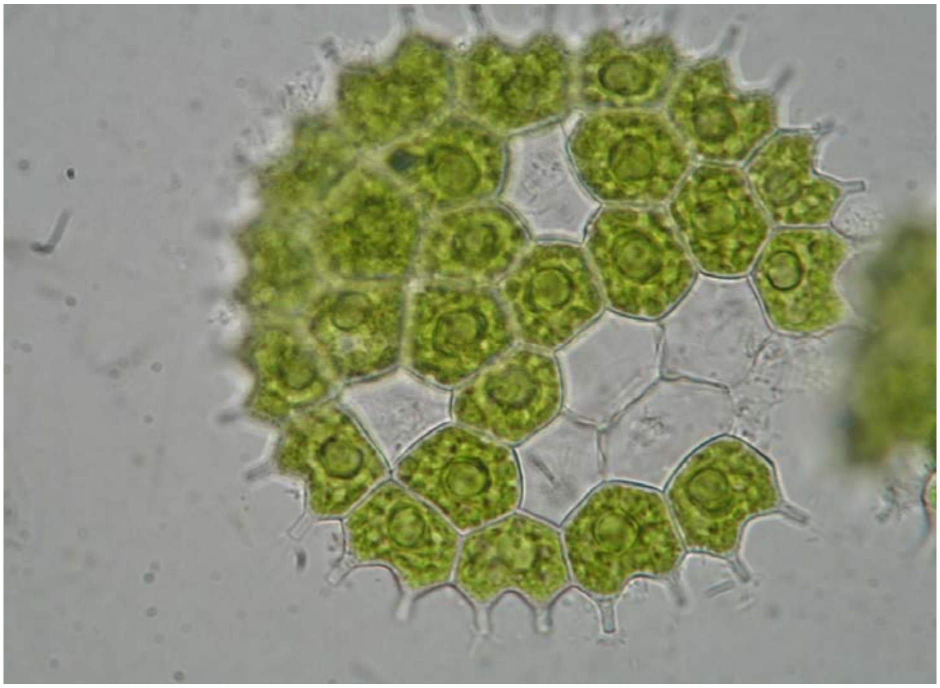

Figure 4.11: Colonial *Pediastrum* sp.

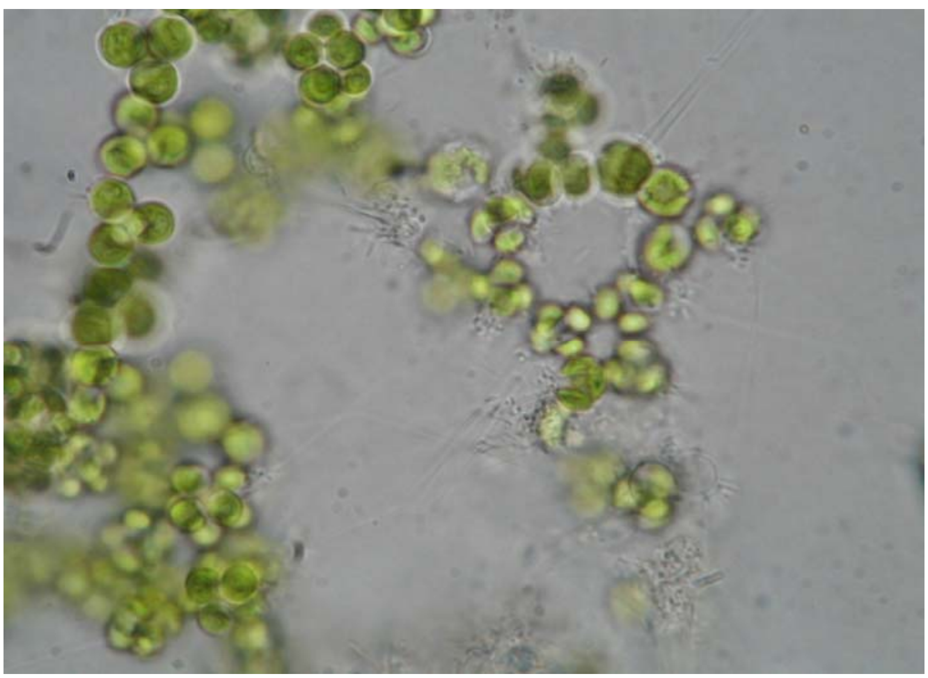

Figure 4.12: Colonial *Micractinium* sp. protected from grazing by long setae.

Figure 4.13: *Actinastrum* sp. was also abundant in the high rate algal pond, algae culture.

Figure 4.14: Grazing rotifers, such as *Philodina* sp., assist in maintaining settleable flocs by removing smaller, more buoyant algae.

4.4 Conclusions

- CODt increased in the HRAP due to the growth of algae biomass, indicated by the similarity in soluble HRAP COD and settled effluent COD, from the ASP.
- The algal biomass was effectively removed in the ASP, giving a net CODt reduction of 43%.
- Although an average 26% phosphate reduction was observed in the HRAP, this was not enough to bring effluent levels to within the $10mg.l^{-1}$ required by the discharge standards.
- Nitrate and ammonium removals were 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.
- The IHRAP, treating GDW final effluent, investigated by Clark (2001), provided effective phosphate and ammonium reduction but poor nitrate removal.
- The ATT study by Neba (2003), showed good nitrate removal but with a corresponding increase in ammonium and phosphate.
- Algae genera found in the PFP were similar to those recorded in conventional WSP. The conditions in the HRAP, however, resulted in a lower species diversity, dominated by floc forming, green algae types.

CHAPTER 5

High Rate Algal Pond Unit Operation Optimised for Nutrient Removal and Disinfection

5.1 Introduction

The South African government has identified that the continued availability of water presents one of the most serious challenges to sustainable development in Southern Africa (DEAT, 1999). It has also recognised that there is an ongoing degradation of the national water resource, further increasing stress on an already limited resource. Rose *et al*. (2002a) cites the White Paper on Pollution and Waste Management as noting that salinity and sanitation issues contribute six of the seven priority sources of pollution and the development of locally appropriate technologies to deal with this problem requires urgent attention.

As the IAPS is intended for use in smaller and rural communities, there is a high possibility that the final effluent will, either intentionally or unintentionally, be used for irrigation, livestock watering, washing or recreation. It is therefore imperative that this water is pathogen free (van Leeuwen, 1996). As these areas also, generally, have a shortage of skills, any disinfection mechanism needs to be relatively maintenance free, with low levels of operator skill and a high level of reliability required for it to function.

With this motivation, it was decided the monitoring of faecal indicator organism removal by the IAPS, was a critical area requiring scrutiny. Apart from occasional *ad hoc* sampling, no previous studies at Rhodes University had focused directly on the disinfection capacity of the IAPS under local conditions. The study reported here, therefore investigated disinfection both in the Grahamstown IAPS as well as in an IHRAP.

Research by Clark (2001), Dekker (2002) and Neba (2003) proved that the IHRAP was effective in removing nutrients, particularly N and P from GDW effluent, indicating the potential widespread use of the system as a final polishing operation that might be appended to a large number of existing small plants, especially in rural areas, that fail to meet nutrient removal requirements. The above studies also led to the proposal that the HRAP separated into a series operation might improve the nutrient and COD removal performance of the IAPS, where the HRAP was used as an averaging operation i.e. with the PFP effluent split between two parallel units. Figure 2.6 illustrates the two different configurations. One of the objectives of this study was to test this concept within the Grahamstown IAPS. Effluent from the first HRAP, providing secondary treatment was thus directed through a second HRAP, effectively functioning as an IHRAP, where tertiary polishing occurred. Thus, in addition to the disinfection findings, data on the optimisation of the HRAP tertiary functionality are also presented here.

5.2 Integrated Algal Ponding System Disinfection

5.2.1 Disinfection Mechanisms

Many different mechanisms play a role in disinfection in high rate ponds. These include predation, sunlight, temperature, dissolved oxygen, pH, sedimentation and starvation (Fallowfield *et al*., 1996). There is, however, much debate over which of these is the most important. Algal photosynthesis causes an increase in the pH due to the simultaneous removal of CO_2 and H^+ ions (Fallowfield *et al*., 1996) and the uptake of bicarbonate when the algae are carbon limited (Craggs *et al*., 1997). According to Rose *et al*. (2002a) a pH of 9.2 for 24 hours will provide a 100% kill of *E. coli* and most pathogenic bacteria and viruses. Pahad and Rao (1962) also found that *E. coli* could not grow in wastewater with a pH higher than 9.2.

Pearson *et al*. (1987) believe that faecal coliforms are adversely affected by pH and DO but not necessarily by high light intensities. The study by Craggs *et al*. (2004), however, contests this and they claim that sunlight is the most important factor, while pH and DO are only second order factors in disinfection. Although there are conflicting opinions as to the relative importance of these various factors, it appears that they all contribute to pathogen reduction. A combination of aspects might, in fact, be essential for optimal disinfection in HRAP. This concept is 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 membranes (Davies-Colley *et al*., 1999).

As the COD of the ponds is relatively low, there is, possibly, also a certain amount of starvation taking place. Sedimentation is often given as one of the reasons for pathogen reduction in conventional WSP (Almasi & Pescod, 1996; Maynard *et al.*, 1999). There is possibly a certain amount of removal taking place in the algal settling pond, however, analysis of samples taken from the HRAP showed similar *E. coli* levels as the ASP effluent (data not shown).

Many of the factors normally responsible for the elimination of indicator faecal coliforms are not active in the PFP. These include a pH of greater than pH 9 (Parhad & Rao, 1974; Sebastian & Nair, 1984; Pearson *et al*., 1988), good sunlight penetration (Craggs *et al*., 2004) and high levels of dissolved oxygen (Davies-Colley *et al*., 1999). The *E. coli* reduction in the PFP can 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).

5.2.2 Materials and Methods

Materials and methods are as described in chapter 2

5.2.3 Results and Discussion

5.2.3.1 Pathogen Removal in the Primary Facultative Pond

From Figure 5.1 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. The *E. coli* levels in the PFP effluent were, however, still far higher than those required for safe discharge into a natural water course. These levels are consistent with a number of other pathogen removal investigations in primary ponds (Pearson *et al*.,1995; Jagals & Lues, 1996; Rangeby *et al*., 1996; Almasi & Pescod, 1996)

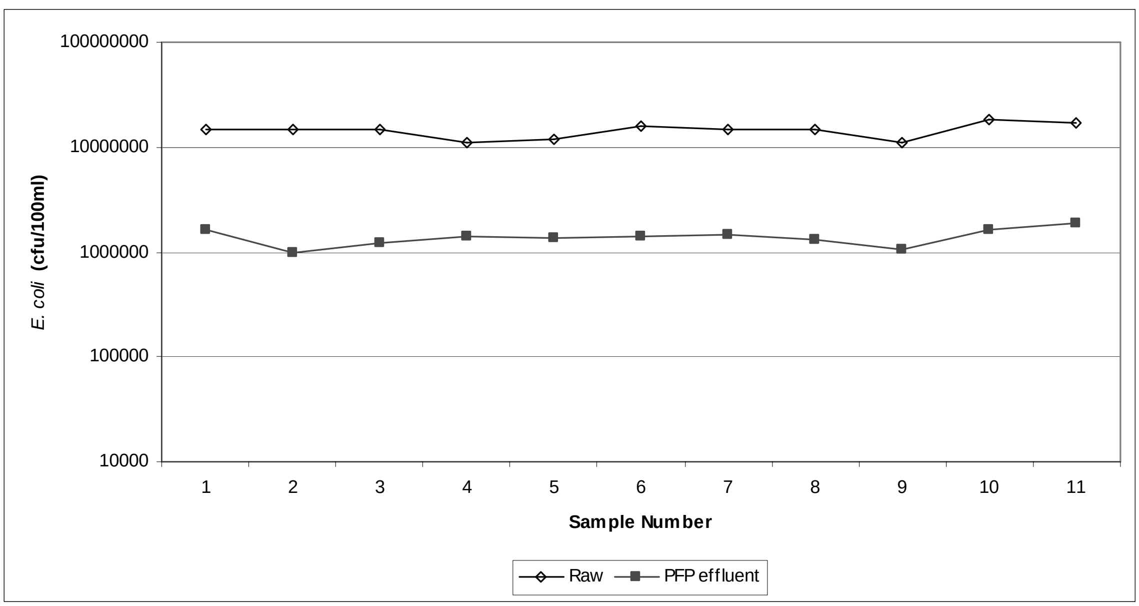

Figure 5.1: *E. coli* removal in the primary facultative pond.

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

5.2.3.2 Disinfection in the High Rate Algal Pond

Figure 5.2 illustrates the *E. coli* removal performance of the high rate ponds over the period of a year.

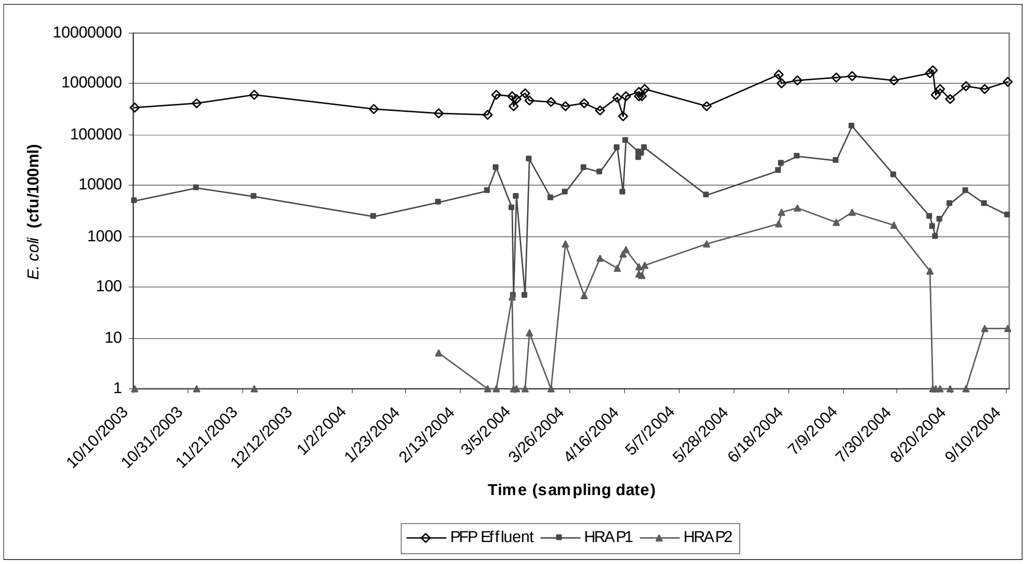

Figure 5.2: *E. coli* removal in high rate algal pond 1 and high rate algal pond 2, during the period October 2003 to September 2004.

From Figure 5.2, 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* in the region of 10^4cfu.100ml^{-1}.The second HRAP then has the capacity to completely remove the remaining *E. coli* and faecal coliforms. As the seasons moved into winter, it is evident that the indicator count in HRAP2 increased, although, apart from June and July (mid-winter), the faecal coliform remained below 1000cfu.100ml^{-1}, the South African guideline for irrigation (DWAF, 2001). By increasing the hydraulic retention time, it was possible to achieve a count of 0cfu.100ml^{-1} for both *E. coli* and faecal coliform, even under winter conditions (Figure 5.3)

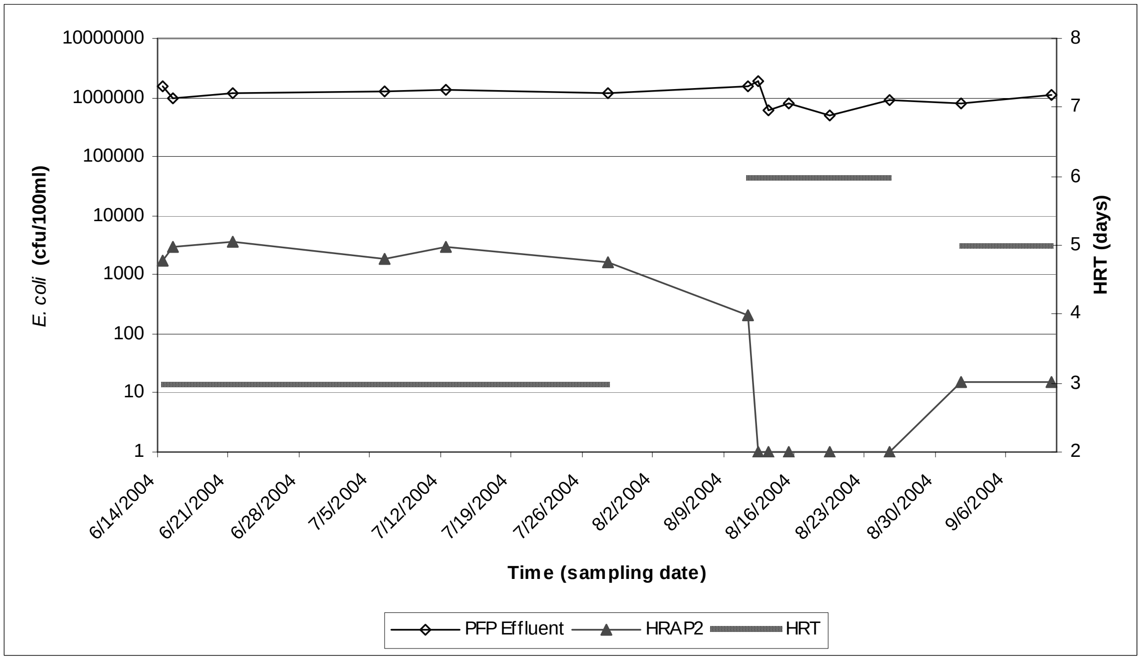

Figure 5.3: *E. coli* removal under winter conditions at different hydraulic retention times.

At a HRT of 6 days there was a 100% kill. When this retention time was decreased again to 5 days, the cfu count went up to 15cfu.100ml^{-1}. This is still a 99.999% removal but in order to attain indicator free effluent, a HRT greater than 5 days was found to be necessary. Under a 6 day HRT, total coliform counts were also 0cfu.100ml^{-1} (data not shown). These seasonal adjustments to HRT are consistent with those found by El Hamouri *et al*. (1994) in Morocco, where a winter 6 day 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.100ml^{-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% in their respective HRAP. Other studies have shown comparable disinfection rates, with faecal coliform or *E. coli* levels in HRAP effluents, less than 100cfu.100ml^{-1} (Sebastian & Nair, 1984; Davies-Colley *et al*., 2003).

From the observations made in this study, it is clear there is a compelling inverse correlation (r = -0.75) between pH and *E. coli* inactivation and this appears to be one of the key disinfection agents (Figure 5.4). As the pH drops below pH 9.5 in the HRAP, there is a sharp rise in *E. coli* count. When this pH is again increased to above pH 10, 100% removal was once again attained.

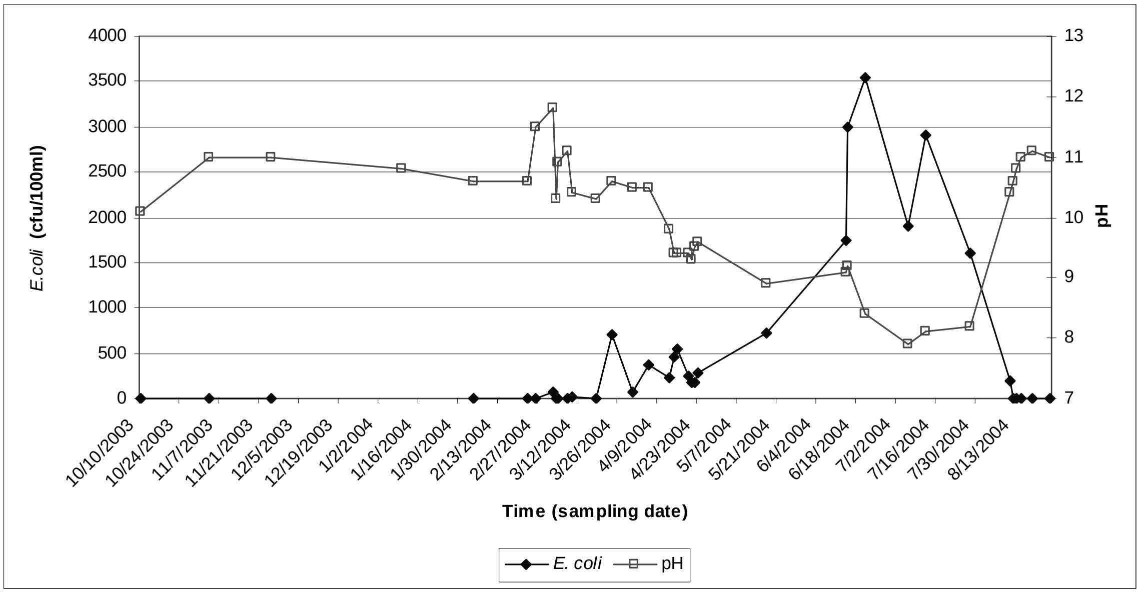

Figure 5.4: An increase in *E. coli* in the high rate algal pond correlates with a decrease in pH and *vice versa*.

Figure 5.4 depicts results of analysis done on samples taken from the demonstration HRAP in the field, which is not in a controlled environment. It is, therefore, difficult to attribute the pathogen fluctuation to pH alone as pH, temperature, DO and solar radiation all vary diurnally almost in phase with each other and are interrelated (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.

5.3 Optimisation of the High Rate Algal Pond Performance

Chapter 4 describes the relatively inconsistent nutrient removal performance of the HRAP. This was, however, greatly improved by converting the HRAP unit operation into a two pond, series configuration.

5.3.1 Materials and Methods

Materials and Methods are as described in Chapter 2.

5.3.2 Results and Discussion

5.3.2.1 Nutrient Removal

It is evident from Chapter 4 that at times there was an increase in ammonium in the first HRAP. However, even with this increase, it can be seen from Figure 5.5 that the second HRAP was able to successfully decrease the amount to below South African discharge standards ($3mg.l^{-1}$) (South African National Water Act No. 36 of 1998).

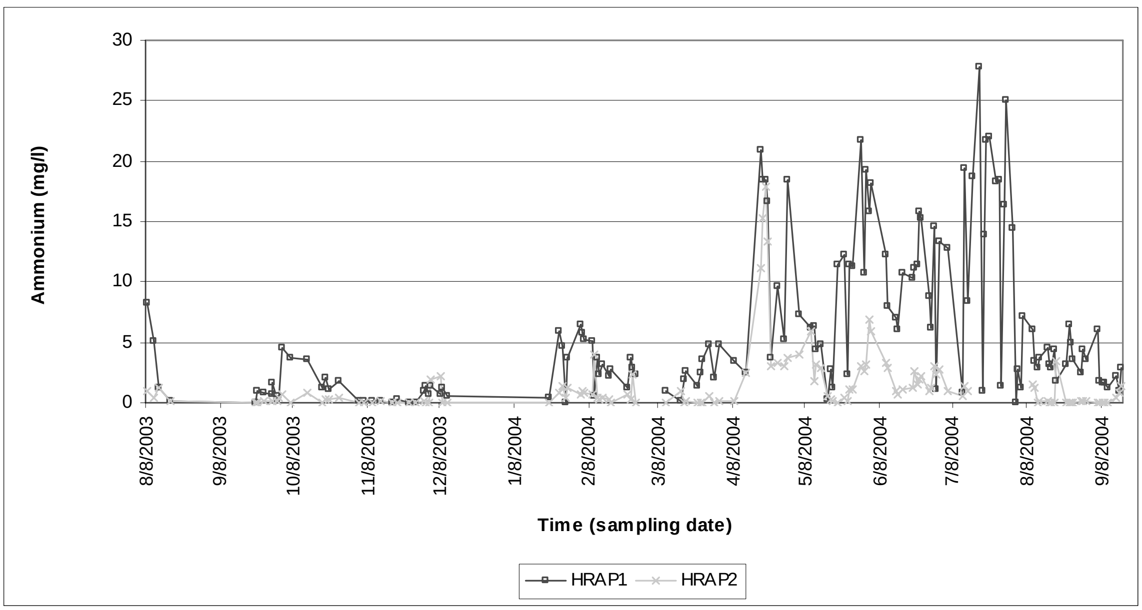

Figure 5.5: Ammonium removal in high rate algal pond 2 when the two high rate algal ponds were operated in series.

As this water is continuously mixed and also attains pH values of up to pH 11, the most probable cause of ammonium removal in HRAP2 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. Algae uptake may also be responsible for some of the ammonium removal (Nurdogan & Oswald, 1995; van der Steen *et al.*, 1998).

As can be seen from Figure 5.6, the high levels of nitrate recorded in HRAP1 and reported in Chapter 4 are not effectively removed in the second HRAP. As discussed previously, this is probably due to the aerobic conditions of the HRAP allowing nitrification to take place but inhibiting denitrification.

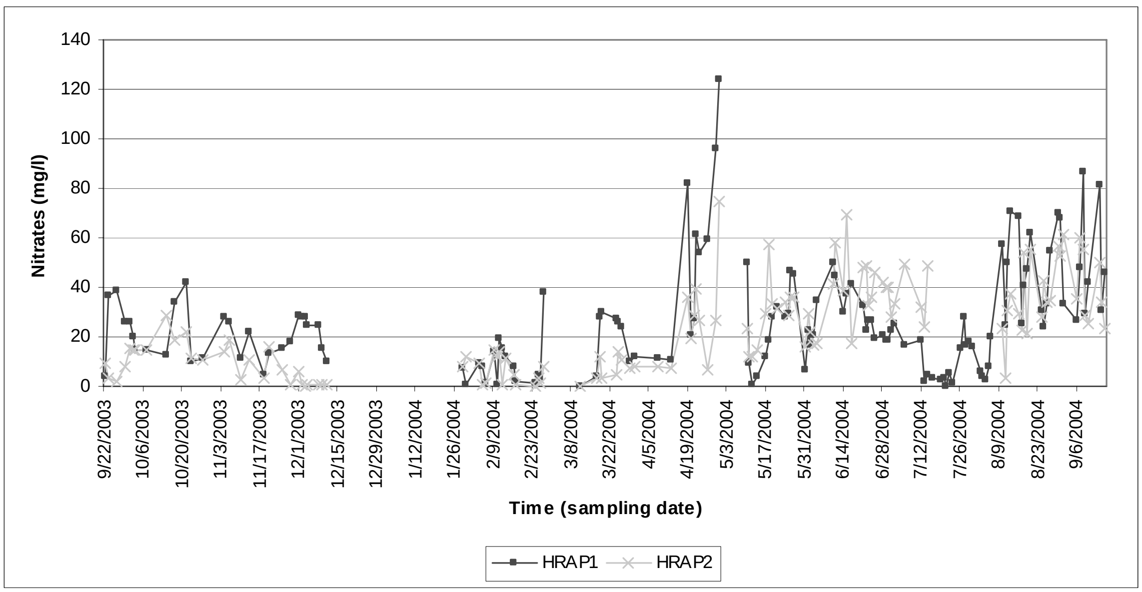

Figure 5.6: Nitrates in high rate algal pond 1 and high rate algal pond 2 for the period August 2003 to October 2004.

Although the figures described in chapter four initially discredit the HRAP's phosphate removal capacity, enhanced phosphate removal was achieved by running the second HRAP in series (Figure 5.7). Over a six month period, from late winter to mid summer, an 85% phosphate removal was achieved across the two HRAPs. Dekker (2002) realised similar results treating Grahamstown Disposal Works final effluent in the IHRAP.

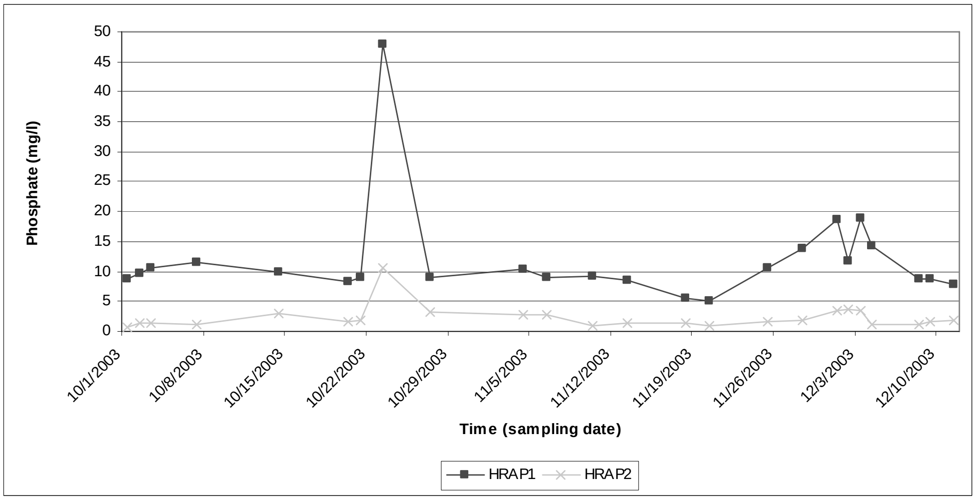

Figure 5.7: Phosphate removal between high rate algal pond 1 and 2**.**

Figure 5.8 clearly shows that during periods of low pH, there is a corresponding increase in phosphate levels and *vice versa*. Statistical analysis of the data gave a correlation coefficient r = -0.99, indicating an extremely good inverse correlation. This close correlation between pH and phosphate lends support to the theory that phosphate is removed by precipitation, although a higher pH could also be indicative of increased algal activity and, thus, greater phosphate uptake.

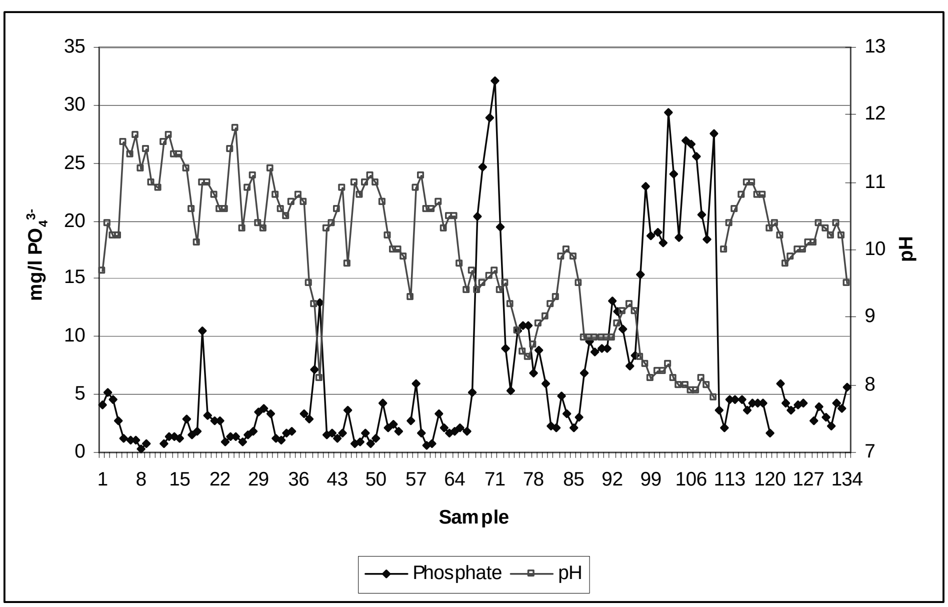

Figure 5.8: There is an inverse relationship between phosphate levels and pH.

5.3.2.2 Chemical Oxygen Demand Removal in High Rate Algal Pond 2

Although the principle objective of the series operation of the HRAP 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 4, was also observed in the second HRAP and ASP in the IAPS series, with only a slightly better oxygen demand in the final effluent. Once again, the COD in the HRAP can mostly be attributed to algal biomass.

It is evident from Figure 5.9 that the series operation of the 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 between HRAP1 and HRAP2.

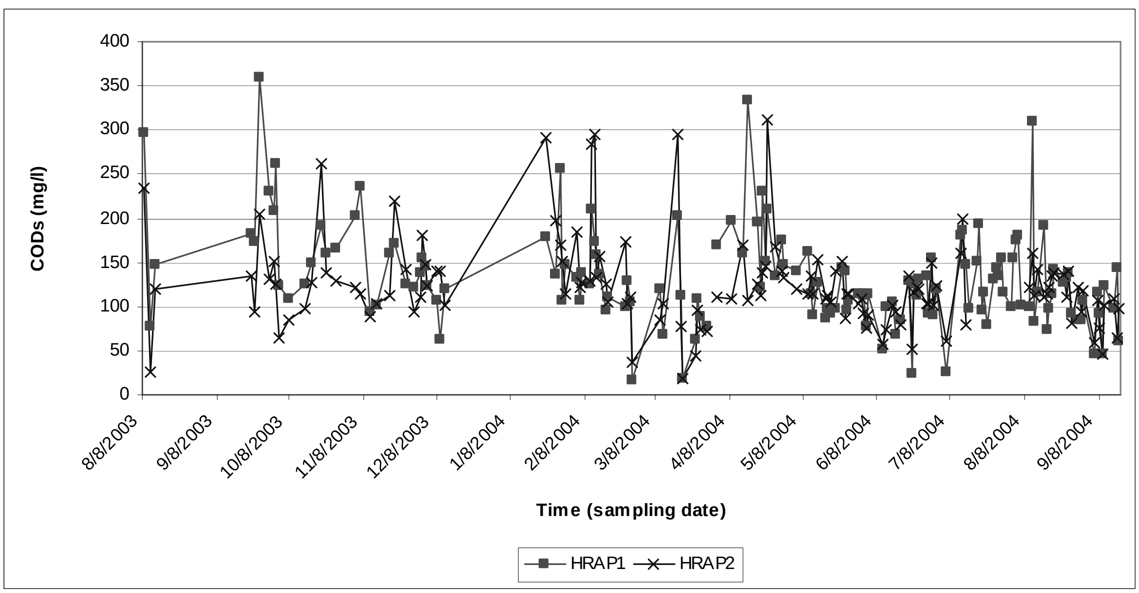

Figure 5.9: Soluble chemical oxygen demand removal between high rate pond 1 and 2.

5.4 Series Operation of HRAP

In order to check whether a second HRAP was necessary or whether simply increasing the retention time in one HRAP would be as effective, the retention time in HRAP1 was doubled and the results compared to the average effluent quality attained with the two pond configuration. After a month monitoring period the second HRAP was brought back on line. The phosphate results from this experiment are represented in Figure 5.10.

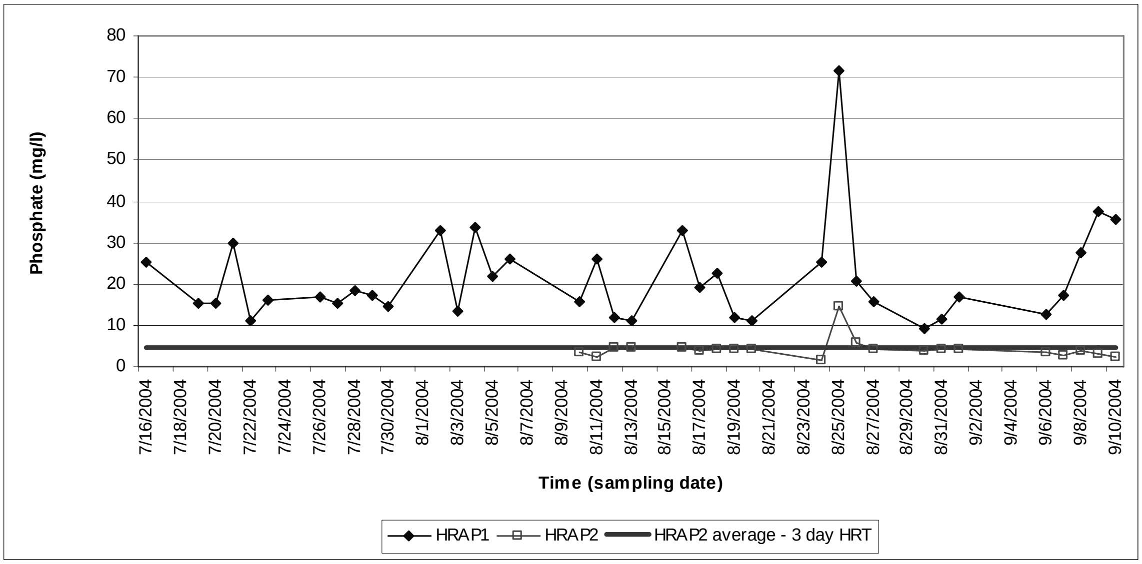

Figure 5.10: Comparison between phosphate removal in the first high rate algal pond at a 6 day hydraulic retention time with high rate algal pond 2, at a 3 day hydraulic retention time.

With a 6 day HRT, one HRAP was still not able to drop the phosphate levels to that of the second HRAP 3-day HRT average. Immediately the second HRAP was brought on line, phosphate removal improved. A similar effect was found for *E. coli* kill, where one HRAP was unable to achieve 100% inactivation even with a 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.

Table 5.1 below gives the figures for the most stable operating conditions achieved over a period of approximately three months, within the eighteen month study on the series operation of the HRAP. The phosphate reduction rate was similar to the average for the whole study at 86%. An ammonium removal rate of nearly 90% was realised during this period. The nitrate removal was better than indicated by the 18 month mean, with the final effluent containing a mean nitrate of 11mg.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.100ml^{-1}, a 0 cfu.100ml^{-1} count was observed over 80% of the time, with 1 result of 65 cfu.100ml^{-1} inflating the mean.

Table 5.1: A summary of the performance of the high rate algal pond unit operation, configured in series.

	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.100ml^{-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

5.5 Conclusions

- Under the correct configuration and process conditions it was found that the Grahamstown demonstration IAPS provided a 100% kill of *E. coli* indicator.
- A second HRAP operated in series provided far superior disinfection than one HRAP operating as an averaged treatment process within the IAPS, providing perhaps the greatest advantage of the reconfigured HRAP unit operation.
- As was shown in the IHRAP treating GDW effluent, an additional advantage of the second HRAP in the IAPS was the consistent, effective removal of phosphate and ammonia to <5mg.l^{-1} and <2mg.l^{-1} respectively.
- The second HRAP had a negligible effect on further COD or nitrate removal.

CHAPTER 6

Comparative Performance Evaluation of the Integrated Algal Ponding System in Grahamstown

6.1 Introduction

The previous chapters have discussed the data pertaining to the performance of the individual unit operations in the IAPS. These various stages are, however, designed to exploit different aspects of wastewater treatment, each contributing to the overall performance of the system. Before this study, no collation of long term monitoring data of the Grahamstown IAPS had been undertaken and although various applications of the system had been investigated, the viability of the system as an alternative treatment technology had not been evaluated over an extended period. This chapter, therefore, presents operational data, from commissioning in 1996 until October 2004, in such a way as to give an overview of the IAPS as an alternative technology for domestic wastewater treatment, comparing effluent quality with more conventional sewage works.

6.2 Organic Removal

Organic material entering a watercourse will act as a food source for the microorganisms present in the receiving water and will be oxidised in a series of oxidation reactions (Horan, 1996). The oxygen required for these reactions is obtained from the dissolved oxygen in the water, with the water consequently deaerated. At the same time the water is reaerated by oxygen transfer 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 in the water. The difference between the deaeration and subsequent reaeration 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 water course. 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 6.1 illustrates the total (unfiltered) CODt removal performance of the Grahamstown demonstration IAPS.

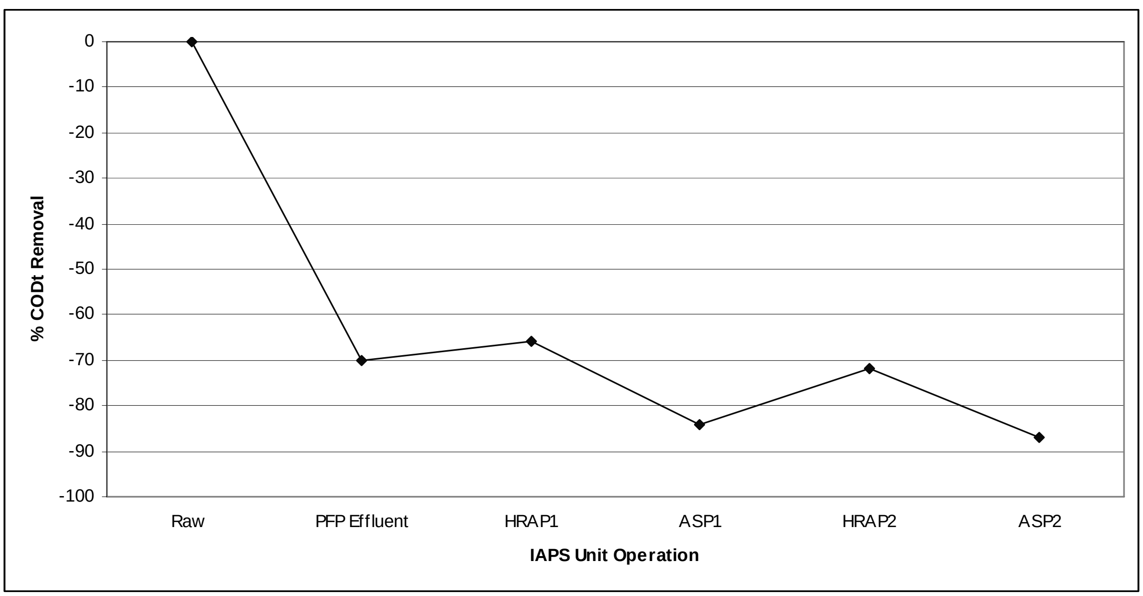

Figure 6.1: Chemical oxygen demand removal through the integrated algal ponding system. The results depicted for the raw, PFP effluent, HRAP1 and HRAP2 are averages for the entire period from July 1996 to October 2004. HRAP2 and ASP2 results are averages since these units were brought online in July 2003.

As depicted in Figure 6.1, 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 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 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 75mg.l^{-1} (DWAF, 2001). However, a large portion of this residual CODt is algae, 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 algae 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. 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 systems these have problems with rapid clogging and centrifuge size respectively (Oswald, 1988b). The drawback of any of these algae 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.

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. As it is rich in nutrients and plant hormones, the most obvious use would be as a fertiliser (Benemann *et al*., 1980). In work done with algae harvested from the Grahamstown plant, Horan and Horan (2004) 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 15.4t.ha^{-1} in soil enriched with HRAP algae, whilst commercial fertiliser only yielded 10.5t.ha^{-1} and unfertilised land, 3.2t.ha^{-1}; a combination of algae and fertiliser once again had the greatest yield at 18.5t.ha^{-1} (Horan & Horan, 2004). The growth ratio changed however, with the second harvest. The algae/fertiliser plot was still the most productive, with a yield of 35.9t.ha^{-1} but the fertiliser alone (27.8t.ha^{-1}) performed better than the algae alone (23.2t.ha^{-1}); the control plot was still considerably less than the others, yielding only 8t.ha^{-1} (Horan & Horan, 2004). As with the other two crops, an algae/fertiliser mix gave the most productive cabbage crop. This was then followed by fertiliser alone, algae alone and the untreated control (Horan & Horan, 2004). Interestingly, Horan & Horan (2004) noted that cabbage in plots treated with algae bolted, displaying a large amount of leaf growth but not producing any head.

Another potential use of the algae is as a dietary protein feed supplement for 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 algae revealed a proximate 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).

6.3 Nutrient Removal

Nutrients released in wastewater effluent can have severe impacts on receiving waters and, therefore, require removal before the discharge of effluents. Nitrogen can cause various problems depending on what form it is present in. Ammonia is toxic to aquatic organisms, especially the higher forms such as fish, at concentrations as low as $0.5 mg.l^{-1}$ (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). Where water receiving effluent discharges is used for abstraction, high nitrate levels can cause infant methaemoglobinaemia. Up to the age of about six months, infants have an incompletely developed digestive system and accumulate nitrite ions which enter the bloodstream (Barnes & Bliss, 1983). 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 algae species is beneficial to a healthy aquatic ecology, high levels of nutrients often stimulate problematic algae 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 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 physical parameters, 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 6.2 and 6.3 below illustrate the cycling of ammonium and nitrate, respectively, through the IAPS. Due to ammonification and possibly nitrogen fixation, there is an increase in ammonium in the first HRAP. This is then effectively removed in HRAP2 by the probable mechanism of volatilisation and possibly assimilation into the algae biomass. The mean ammonium level in the final effluent is 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 ammonium levels of between 10 and $40 mg.l^{-1}$ are common (Horan, 1996). It also appears to be better than ordinary WSP, where effluent ammonium 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 ammonium 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 ammonium levels of between 2 and $3 mg.l^{-1}$.

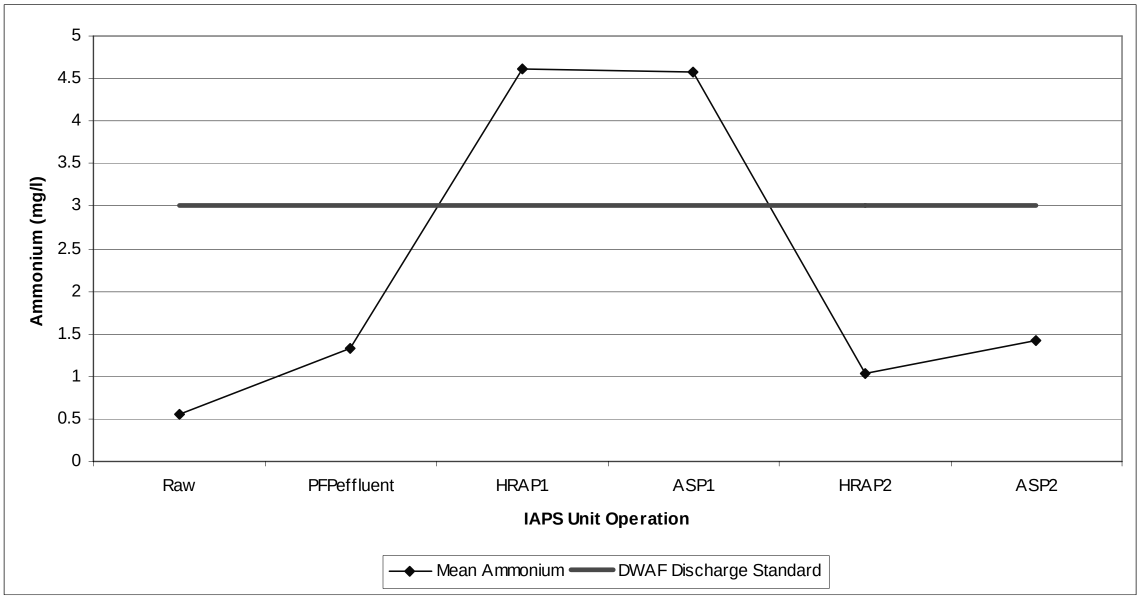

Figure 6.2: Average results for ammonium cycling through the integrated algal ponding system, monitored over the period 1997-2004 and for HRAP2 and ASP2 from 2003-2004.

The nitrate increase depicted in Figure 6.3 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 nine year life of the IAPS is below the 15mg.l^{-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. According to Horan (1996), sewage effluent routinely contains nitrates of between 5 and 30mg.l^{-1}. The variation in HRAP nitrate levels may be due to differences in algal productivity. 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 responsible 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 ($3 mg.l^{-1}$), mean total nitrogen (TKN) removal in the system was, nevertheless, 55%. 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).

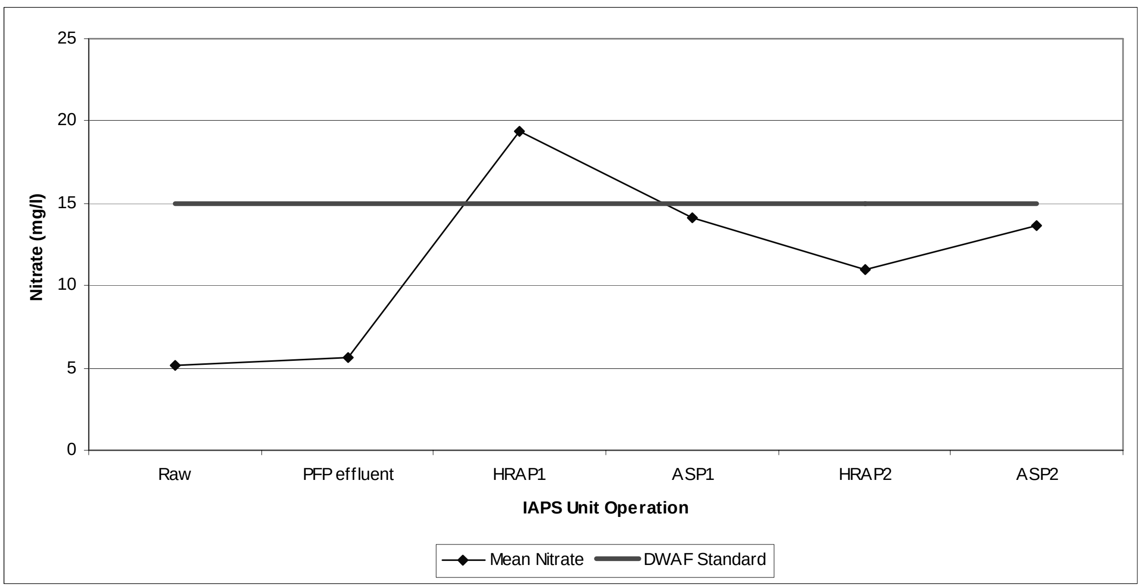

Figure 6.3: Average values for nitrate cycling through the integrated algal ponding system, monitored over the period 1997-2004 and HRAP2 and ASP2 for 2003-2004.

Phosphorous has been identified as the key limiting nutrient, stimulating the explosive growth of algal populations (Campbell, 1993). Consequently if the phosphorous containing compounds present in sewage effluents can be removed, then eutrophication can be controlled regardless of the nitrate concentration (Horan, 1996). Most biological treatment works, wanting to remove phosphate, exploit the enhanced storage of polyphosphate (poly-P) in microorganism biomass otherwise known as luxury uptake (Mino *et al*., 1998). This process is termed enhanced biological phosphate removal

(EBPR) (Horan, 1996). By circulating activated sludge through anaerobic and aerobic phases, coupled with the introduction of wastewater into the aerobic phase, poly-P accumulating organisms (PAO) are selected and grow to dominance. (Mino *et al*.,1998). Mino *et al*. (1998) explain the process as follows. PAOs are able to hydrolyse stored poly-P in order to supply energy for the anaerobic uptake of carbon sources, which are stored in the form of polyhydroxyalkanoates (PHA). This is accompanied by the degradation of poly-P and the consequent release of orthophosphate. In the subsequent aerobic phase, PAOs grow aerobically, taking up the orthophosphate. Phosphate removal is then achieved by withdrawing the excess sludge with high phosphate content. Although effluent phosphorous of less than $1mg.l^{-1}$ can be achieved in plants specifically designed to optimise EBPR, the sophisticated control that is required, make them unsuitable for developing countries (Horan, 1996).

The efficacy of phosphate removal in the IAPS is portrayed in Figure 6.4. 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 $5.4mg.l^{-1}$, 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). High rate ponds 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.

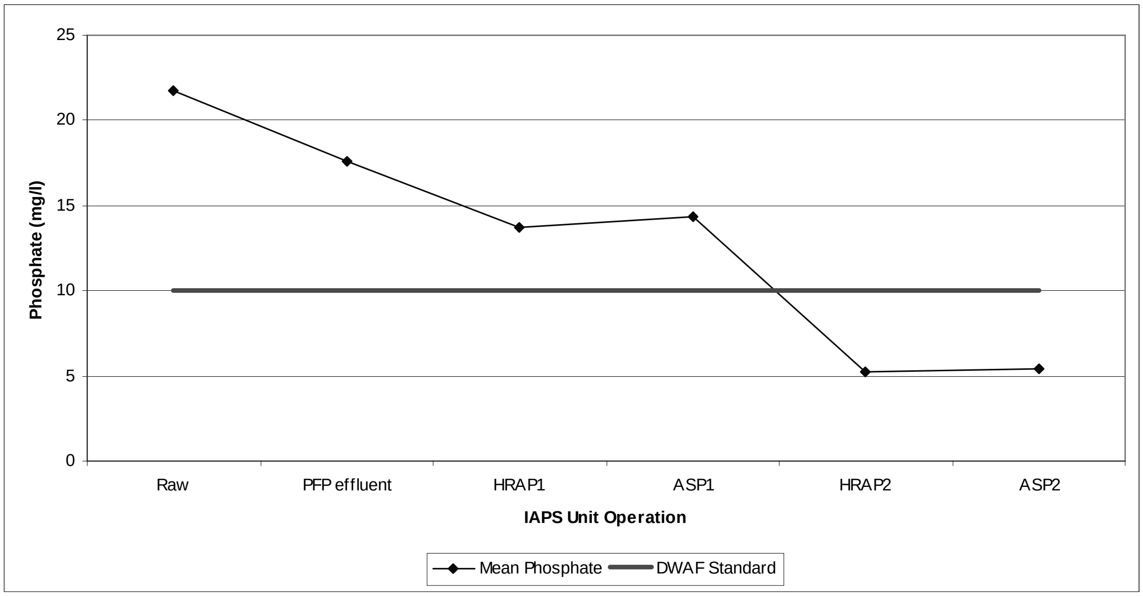

Figure 6.4: Average results for phosphate removal in the integrated algal ponding system, monitored over the period 1997-2004. HRAP2 and ASP2 averages for 2003-2004.

6.4 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 in developed countries (Stott *et al*., 2003). As has been suggested throughout this report, the IAPS technology has been researched at Rhodes University 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 <1cfu.100ml^{-1} (Law, 1996) but are often too costly for smaller municipalities (Fujioka *et al*., 1999). The use of the HRAP as a polishing step for pathogen removal was thus investigated as part of this study.

Figure 6.5 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 <1000cfu.100ml^{-1}. These figures, however, represent the mean results over the total eighteen month monitoring period, including winter periods of experimental, 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 <10cfu.100ml^{-1} (Figure 6.6). This equates to a 99.999% reduction. Zero *E. coli* were recorded in 78% of the samples.

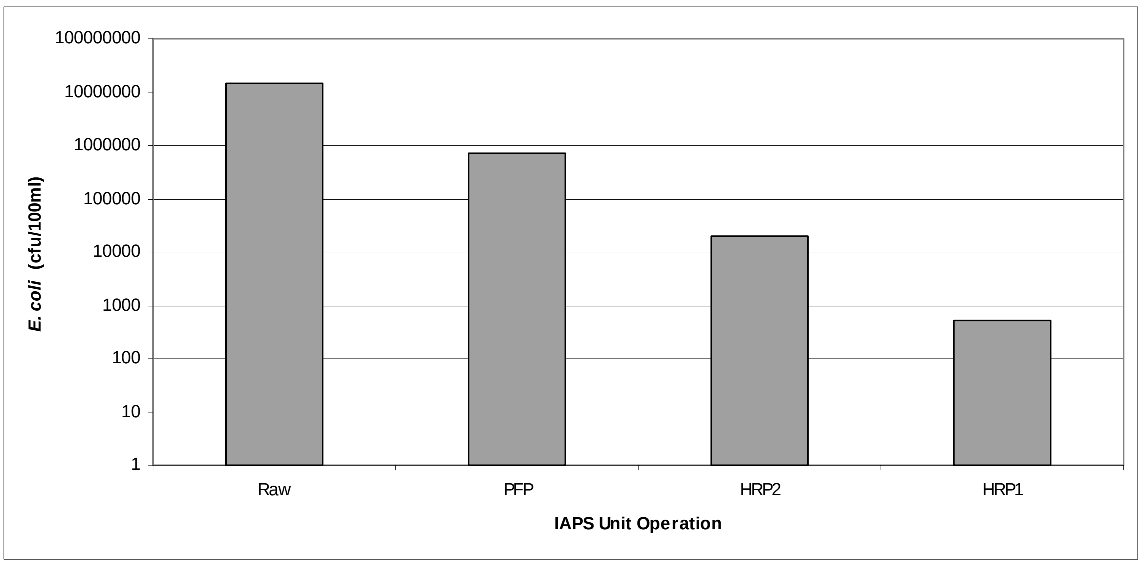

Figure 6.5: *E. coli* counts through the integrated algal pond system sequence. This figure illustrates all data from the 2003-2004 monitoring period, i.e. including results from operation with sub-optimal hydraulic retention times.

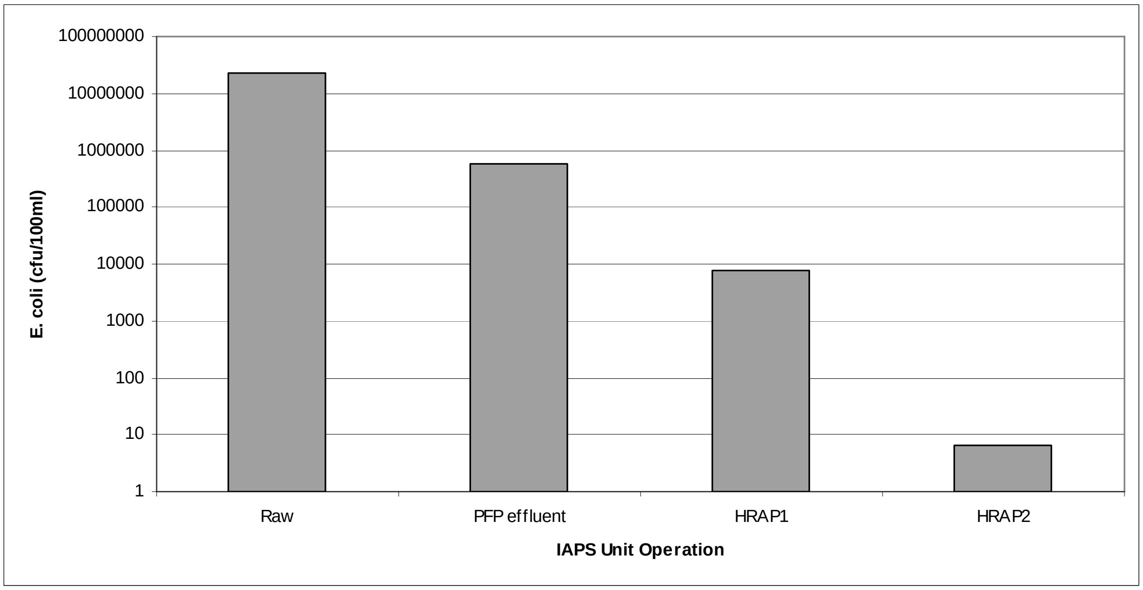

Figure 6.6: *E. coli* counts through the integrated algal pond system sequence, illustrating results only from operation under optimal hydraulic retention times, both during 2003 and 2004.

In most instances, standard WSP are unable to reduce faecal coliforms to below 1000cfu.100ml^{-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 1000cfu.100ml^{-1} (Arias *et al*., 2003; Ansola *et al*., 2003). Davies-Colley *et al*. (2003) achieved similar results to the Grahamstown plant, using a HRAP followed by a maturation pond in New Zealand. Sebastian and Nair (1984) also managed total *E. coli* removal with a 2 day contact time at pH 11 in an experimental HRAP system operated in India.

6.5 Conclusions

- IAPS provides significant organic removal, with a mean CODt reduction of almost 90% across the system.
- Effective ammonium and phosphate removal was achieved in the system, with final effluent levels below South African discharge standards.
- With correct operating conditions, it was possible to obtain an *E. coli* count of 0 cfu.100ml^{-1} in the IAPS final effluent.

- Initial trials have shown potential benefits of algae as a fertiliser and feed supplement.
- Nitrate removal persists as a weak point of the IAPS technology and remains as an area requiring further research.

CHAPTER 7

Conclusion

The IAPS has been operated at the Rhodes University Environmental Biotechnology Experimental Field Station in Grahamstown, since commissioning in 1996. Although various aspects of the system have been the focus of a number of studies, the long term performance data had not been evaluated. In addition, it was believed, the tertiary treatment capacity of the IAPS could be improved and these issues formed the basis for this investigation.

The overall assessment of the IAPS performance during the nine year study period found that COD removal was reliable and effective. The IAPS has also shown a considerable COD buffering capacity, where large fluctuations in COD load were absorbed by the system, leaving no peaks in final effluent COD. Although nutrient removal did take place in the system it was more erratic and, at times, poor.

With the reconfiguration of the HRAP unit operation into a series formation, nutrient removal was greatly enhanced, with phosphate and ammonium discharge standards consistently met. These results concurred with those from the studies on the IHRAP treating GDW final water.

None of the earlier studies on the IAPS concentrated in any detail on the disinfection efficacy of the system. The removal of faecal pathogens is, however, essential if this system is to be implemented in rural areas where reuse of final effluent is desirable and likely. In this investigation it was found that the IAPS has the ability to reduce the *E. coli* and faecal coliform count to 0 cfu.100ml^{-1} in the final effluent. This was only possible with the two pond series operation of the HRAP unit. Although a HRT in each HRAP of 3 days was sufficient under summer operating conditions, it was observed that this needed to be doubled to a 6 day HRT in winter, in order to achieve comparable pathogen removal results. The removal of faecal coliforms in the IHRAP is extremely important for the application of the system in small municipalities and rural communities, where treatment plant effluents rarely meet pathogen discharge standards (Antrobus, 2002). Future studies should assess the effect of the HRAP unit operation on VBNC, as this will present a more in-depth measure of the disinfection value of the IAPS than the cfu results discussed here.

TKN removal in the IAPS was substantial. However, since commissioning, nitrate removal in the IAPS has proved fickle. The ATT reactor effectively removed nitrates but at the expense of increased ammonium and phosphate levels. Some preliminary manipulation of the organic loading to the HRAP unit operation has shown potential for improving nitrate reduction. This aspect of the IAPS needs further investigation and should form the focus of future studies.

In addition to the water treatment capacity of the IAPS, the system also produces a beneficial, cost-effective by-product in the form of algal biomass. Preliminary results suggest that algae, harvested from the HRAP, has value as a fertiliser and animal feed supplement. This inexpensive source of protein and nutrients enhances the appeal of the system for application in rural areas.

The collation and evaluation of the entire set of monitoring data, together with the results of the most recent examination of the system, have led to a number of design and operation recommendations. For optimal nutrient removal and disinfection, a two stage HRAP unit operation is advised i.e. the effluent from the first HRAP, after algae removal in the ASP, is directed into a second HRAP before discharge. In order to ensure complete disinfection, a six day HRT in each HRAP is required. Alternative use could be made of the extra HRAP capacity in summer, when the HRT needed is only 3 days.

As Oswald (1995) has noted, ponds remain the most economical system for liquid waste management and utilisation of solar energy. The results of this study indicate that, although the IAPS is a low cost, low maintenance treatment option, requiring minimal anthropogenic energy input, it is also capable of providing effective tertiary treatment, removing nutrients and faecal pathogens. There is no universally applicable best technology for domestic wastewater management as each situation necessitates individual appraisal of its specific conditions and objectives. However in South Africa, where more emphasis is being placed on alternative water sources, and particularly, the smaller communities of the Eastern Cape, where sanitation has been neglected, it is suggested that the IAPS has application as a viable treatment technology.

The separation of processes within the HRAP unit operation was found to hold advantages in optimising tertiary treatment, enhancing nutrient removal and disinfection, thus providing an effective polishing unit and verifying the hypothesis on which this research programme has been based.

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