Performance of an integrated algal pond for treatment of domestic sewage: a process audit.

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Abstract

Integrated algae pond systems (IAPS) are energy efficient, robust, passive systems that use the principles of fermentation, photosynthesis and microbial metabolism to remediate wastewater, producing a good quality effluent with reuse potential. In addition to the treatment of wastewater, IAPS have the ability to generate two additional product streams viz. biogas and biomass. The latter adds to the attractiveness of the system. However, the implementation of this technology, like many passive systems, has remained limited at a commercial scale, and the inclination is still towards grey technologies. The aim of this research was to investigate the capabilities and potential of a demonstration-scale IAPS and use results obtained to establish a process audit framework. The aspects considered for the audit included performance efficiency, effluent water quality, biomass composition, quantity and productivity within the ponds, and cost analysis of operation and maintenance over a 9-year period. Plant performance was closely monitored during the course of the study and this led to a review of previously adopted plant management strategies. Troubleshooting exercises were also carried out when plant performance declined. Results showed that IAPS efficiently reduced standard water parameters with the exception of pH, dissolved oxygen, and nitrate whose values increased from raw influent to final effluent. The following water quality parameters were established for the final effluent: total suspended solids 55 ± 7.1 mg. L^{-1} (n = 28); chemical oxygen demand 94.1 ± 10.6 mg. L^{-1} (n = 28) (after removal of algae); pH 9.9 ± 0.01 (n = 26); ammonium nitrogen 1.7 ± 0.3 mg. L^{-1} (n = 25); nitrate 3.3 ± 0.6 mg. L^{-1} (n = 25); ortho-phosphate 1.6 ± 0.2 mg. L^{-1} (n = 25); electrical conductivity 98.7 ± 2.0 mS m^{-1} (n = 26) and faecal coliforms (per 100 mL) 1482.6 ± 636.0 (n = 24). The final effluent measured consistently high chemical oxygen demand and total suspended solids, however close analysis showed that total suspended solids could be controlled by increasing the frequency of removal of settled biomass within the settling ponds. Biomass produced contained microalgae, bacteria, metazoa, and protozoa. The biomass productivity achieved was as high as 130.6 kg ha^{-1} d^{-1}; however, about 33% was lost to the final effluent due to inadequate settling. Results obtained during the course of this study and outcomes of earlier work on IAPS are taken as the baseline to determine parameters needed for the development of the process audit framework. Techniques utilised to derive the blue print process audit protocol for IAPS included a turtle diagram, a flow diagram and a checklist. Attention to

plant management proved vital in determining overall performance. Cost, including operating and maintenance, of treating water using the demonstration scale system on a per person equivalent per year basis was determined as ZAR 123.87 (where, ZAR to USD = 0.07).

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List of Abbreviations

AFP Advanced Facultative Pond

AIWPS Advanced Integrated Wastewater Pond System

AOP Advanced Oxidation Process

ASP Algal Settling Pond

AS Activated Sludge

BOD Biochemical Oxygen Demand

CFU Colony forming Unit

CO_2 Carbon dioxide

COD Chemical Oxygen Demand

CW Constructed Wetland

DB Drying Bed

DO Dissolved Oxygen

DWA Department of Water Affairs

DWS Department of Water and Sanitation

EBRU Institute for Environmental Biotechnology, Rhodes University

EC Electrical Conductivity

EPS Extracellular Polymeric Substances

FC Faecal Coliforms

FP Fermentation Pit

HRAOP High Rate Algal Oxidation Pond

HRT Hydraulic Retention Time

IAPS Integrated Algae Pond Systems

IPD In pond digester

MaB-flocs Microalgal-bacterial-flocs

MLSS Mixed Liquor Suspended Solids

NBS Nature based system

SB Splitter Box

SE Standard Error

TSS Total Suspended Solids

WSP Waste Stabilisation Ponds

WWT Wastewater Treatment

WWTW Wastewater Treatment Work

Chapter 1 Literature Review

1.1 Introduction

The developing world is faced with numerous challenges, which include food insecurity, cost of energy, water scarcity, and sanitation deficiency. A possible mitigation strategy to alleviate some of these challenges is to safeguard and conserve water through the adequate treatment and reuse of domestic, municipal, and industrial wastewater.

Domestic wastewater is actually 99.9 % water (Tebbutt 2013); however, it contains contaminants that must be removed to reduce its threat to humans and the environment (Sonune and Ghate, 2004). Wastewater treatment reduces the concentration of contaminants and also allows for the recovery of water, nutrients, and energy for reuse (Craggs et al. 2014). Wastewater treatment has the potential to:

- Reduce current and future water scarcity by providing an alternative source of water and reducing the freshwater demand (Ding 2017).
- Increase access to safe drinking water by preventing river pollution (WWAP 2018).
- Recover valuable resources (C, N, P) from wastewater which can be utilised in agriculture (Van Den Hende et al. 2016)

Wastewater treatment is not a recent phenomenon as wastewater treatment plants (WWTP) have been operating since the early 20th century in some parts of the world (Lofrano and Brown 2010). In South Africa, about 824 wastewater treatment plants treat 5128.8 Ml d^{-1} with an estimated operational cost of more than ZAR 4 million y^{-1} system^{-1} (DWA: Department of Water Affairs 2013; Cowan 2016). One of the reasons why the full potential of these wastewater treatment systems are not being exploited is the fact that the majority of WWTP release inadequately treated, non-compliant effluent. Compliance standards ensure that effluent released from WWTP is not harmful or polluting to the receiving environment. General authorization limits applicable to South Africa for the discharge of effluent into a water resource (Table 1.1) and irrigation of any land or property up to 2000 kL were established under section 39 of the National Water Act, 1998 and later amended (DWA, 2013) (Table 1.2). The most stringent limits are for the irrigation of land or property up to 2000 kL; however, as the volume of wastewater utilised decreases, limits become relaxed. The reasons for this poor performance in South Africa, among others are; over capacitated

WWTP, inadequately skilled technicians operating WWTP, mismanagement, defective WWTP, and lack of operating resources. The major challenge being experienced in most municipalities is the lack of finances to operate, maintain and upgrade infrastructure, optimize the treatment efficiencies or replace already existing WWTPs (Ntombela et al. 2016; Oberholster et al. 2018). Cowan and co-workers (2016) also argue that the choice of wastewater treatment technologies (WWTT) adopted in South Africa is poor, and this has possibly exacerbated the situation.

Table 1:1: Wastewater limit values for the discharge of wastewater into a water resource (Department of water affairs, 2013)

Parameter	General limit
pH	5.5-9.5
Suspended Solids (mg. L^{-1})	25
Electrical Conductivity	70 mS m^{-1} above intake to a maximum of 150 mS m^{-1}
Chlorine as free Chlorine (mg. L^{-1})	0.25
Orthophosphate as phosphorous (mg. L^{-1})	10
Nitrate/Nitrite as Nitrogen (mg. L^{-1})	15
Ammonia (ionised and un-ionised) as Nitrogen (mg. L^{-1})	6
Chemical Oxygen Demand (mg. L^{-1})	75
Soap, Oil, and Grease (mg. L^{-1})	2.5
Fluoride (mg. L^{-1})	1
Faecal coliforms (per 100 mL)	1000
Dissolved arsenic (mg. L^{-1})	0.02
Dissolved cadmium (mg. L^{-1})	0.005
Dissolved Chromium (VI) (mg. L^{-1})	0.05
Dissolved Copper (mg. L^{-1})	0.01
Dissolved Cyanide (mg. L^{-1})	0.02
Dissolved Iron (mg. L^{-1})	0.3
Dissolved Lead (mg. L^{-1})	0.01
Dissolved Manganese (mg. L^{-1})	0.1
Dissolved selenium (mg. L^{-1})	0.02
Dissolved Zinc (mg. L^{-1})	0.1
Mercury and its compounds (mg. L^{-1})	0.005
Boron (mg. L^{-1})	1

Table 1:2: Wastewater limit values applicable for the irrigation of between 50 – 2000 kL domestic and biodegradable industrial wastewater of any land or property on any given (Department of water affairs, 2013).

Parameter	Up to 2000 kL	Up to 500 kL	Up to 50 kL
pH	5.5-9.5	6 - 9	6 - 9
Suspended Solids ($mg. L^{-1}$)	25		
Electrical Conductivity	70 $mS\ m^{-1}$ above intake to a maximum of 150 $mS\ m^{-1}$	200 $mS\ m^{-1}$	
Chlorine as free Chlorine ($mg. L^{-1}$)	0.25		
Orthophosphate as phosphorous ($mg. L^{-1}$)	10		
Nitrate/Nitrite as Nitrogen ($mg. L^{-1}$)	15		
Ammonia (ionised and un-ionised) as Nitrogen ($mg. L^{-1}$)	3		
Chemical Oxygen Demand ($mg. L^{-1}$)	75	400	5000
Soap, Oil, and Grease ($mg. L^{-1}$)	2.5		
Fluoride ($mg. L^{-1}$)	1		
Faecal coliforms (per 100 mL)	1000	100 000	100 000
Sodium Adsorption Ratio		5	5

A wide range of WWTT are currently implemented in South Africa; however, the most deployed technologies include waste stabilization ponds (WSP), activated sludge (AS), and bio-filtration in the ratios shown in Figure 1.1 (Cowan et al. 2016). Among the drawbacks of the currently utilised systems are the need for large land area, inconsistent water treatment efficiency which also reduces with time, massive energy input requirement, need for biomass/sludge handling/management, production of biomass/sludge with little reuse potential, and high expenses associated with construction, maintenance and operation of these ‗hi-tech' technologies (Nemadire 2011; Oswald 1991). Therefore , to curb these drawbacks the United Nations has proposed the adoption of more passive, nature-based solutions (NBS) for the treatment of wastewater (WWAP 2018).

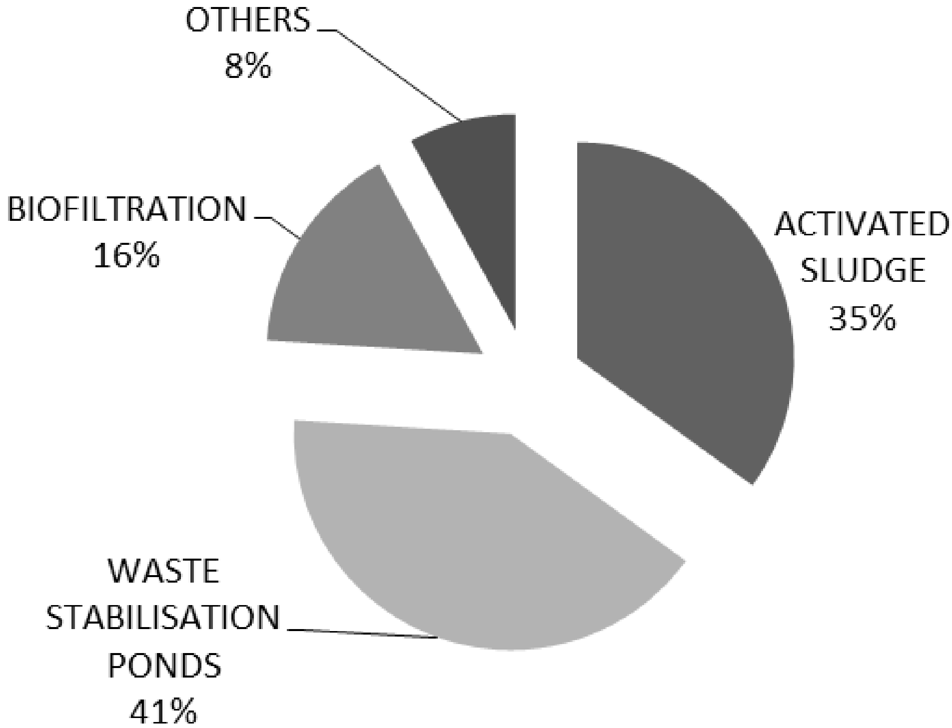


Figure 1:1: Proportion of Wastewater treatment technologies currently in use in South Africa (Cowan et al. 2016).

Nature-based solutions mimic natural occurrences and include safeguarding or upgrading already existing natural ecosystems and or constructing artificial ecosystems, thus providing water management. Examples of such systems include Advanced Integrated Wastewater Pond Systems (AIWPS®), wetlands, waste stabilisation ponds, water hyacinth system, and evapotranspiration systems. NBS remain a scarce, underappreciated and underutilised option, and the inclination is still towards grey infrastructure. This is due to the perception that NBS are inefficient and unpredictable, lack of awareness and lack of research that fully defines the capabilities of NBS has decelerated their use and general lack of inclusion and acceptance at both community and government level has also remained a challenge.

1.2 Wastewater treatment

Wastewater can be viewed as a water supply that has been contaminated by community use. It is classified into domestic, sanitary, industrial, combined and storm wastewater. Domestic wastewater, also known as sewage, consists mainly of household, human and animal waste, greywater and black water, and a small concentration of groundwater infiltrations and industrial waste (Spellman 2013).

The development of waste water treatment was more prominent in the 20th century, and in the years preceding, sewage was disposed of untreated or after dilution (Lofrano and Brown 2010). From then, wastewater technologies have evolved from mere gravity sedimentation to more advanced technologies, as shown in Figure 1.2.

From the community, wastewater is accumulated and transported to wastewater treatment works (WWTW) via wastewater collection and/or sewerage systems that collect and convey wastewater either through gravity systems, vacuum systems and or force main systems. The first point of entry of the effluent at the WWTW is the head of works or preliminary treatment area. Preliminary treatment is the removal of large material at the point of entry and includes some or all of the following processes: influent pumping, screening, shredding, grit removal, flow measurement, pre-aeration, chemical addition, and flow equalization. The aim of preliminary treatment is to reduce maintenance and operational problems within the WWTW, protect equipment and reduce the volume of wastewater, this is achieved by removal of material which could potentially block pipes and pumps and prematurely age plant equipment (Spellman 2013). From the head of works, wastewater goes through primary, secondary, and tertiary/advanced treatment methods depending on the level of effluent standards required.

Primary treatment involves the removal of settleable and floatable solid matter, water is either retained in a quiescent state, or its velocity is lowered to a point where the heavy suspended material sinks to the bottom and lighter material floats to the top of the basin. The end result of primary treatment is a reduction in the biological load. In a typical treatment unit, an upper limit of reduction of 95% settleable solids, 60% total suspended solids (TSS), and 35% biological oxygen demand (BOD) may be observed. Primary treatment was, for a long time, the only form of treatment employed until discharge limits became stricter. The earlier technologies utilised for this phase included trenches and pits, cesspits, sedimentation, septic, and Imhoff tanks (Lofrano and Brown 2010).

Secondary treatment further reduces pollution from primary treated water to produce stable suspended solids, which can subsequently be removed or discharged into the environment without being a nuisance to the receiving body. Secondary treatment is mostly biological and takes advantage of the metabolic activities of microorganisms to produce less reactive compounds. The biological processes of secondary treatment are divided into attached and suspended growth systems. According to the clean water act, the final effluent of secondary treatment has a BOD and TSS of less than 30 mg. L^{-1} (an equivalent of 85-95% removal). Nutrient removal and disinfection may also be targeted at this stage (Metcalf and Eddy 2008; Spellman 2013). Depending on the final effluent use and discharge standard requirements, wastewater can be further treated during an additional tertiary stage.

The term tertiary treatment is often used synonymously with advanced treatment and includes any additional purification of effluent that follows secondary treatment. Tertiary treatment achieves the removal of nutrients and residual organic and inorganic solids; disinfection is also targeted during this stage of treatment. A wide range of advanced technologies, which include advanced oxidation, activated carbon processes, low-pressure membrane techniques and high-pressure membrane technique, may be utilized during this phase of wastewater treatment (Muralikrishna et al. 2017; Dhodapkar and Gandhi 2019).

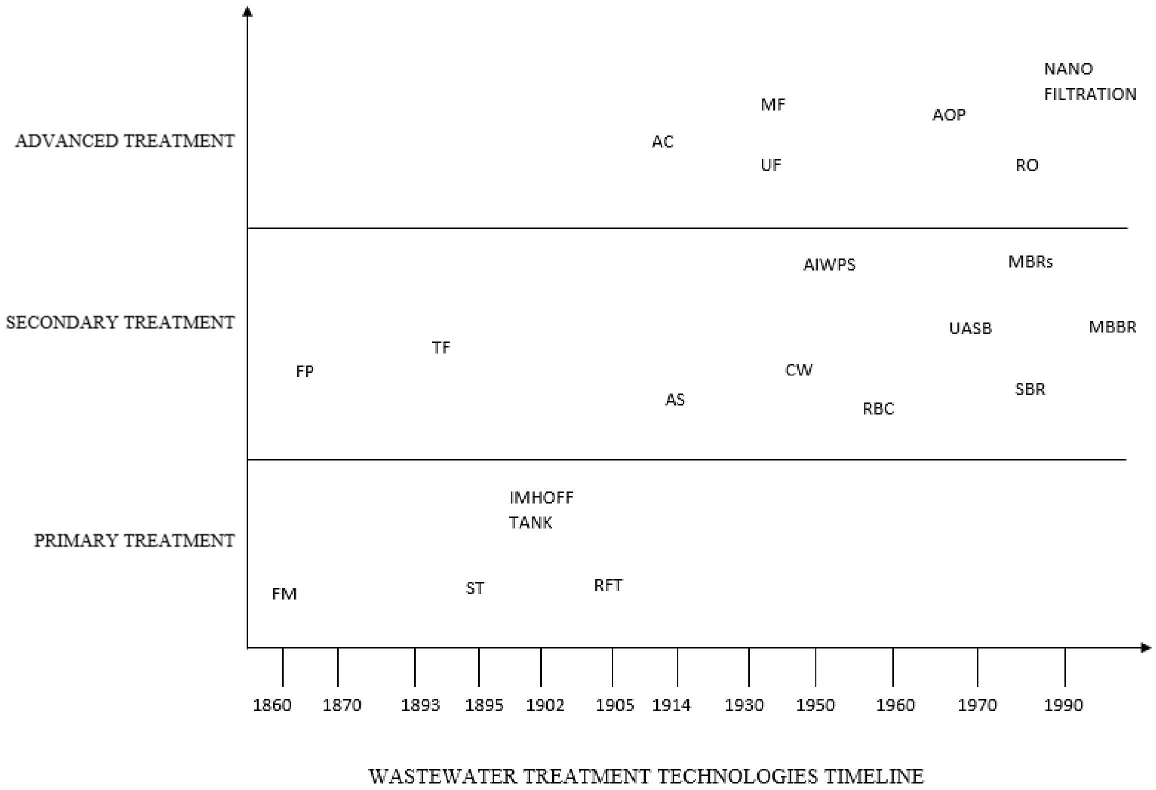


Figure 1:2: Evolution of wastewater treatment. (Adapted from Lofrano and Brown 2010). **FM-**Fosses Mouras**;** **ST**- Septic tank**,** **RFT**- Radial flow tank**;** **FP-** Filtration process; **TF-** Trickling Filter**;** **AIWPS-** Advanced integrated wastewater pond system**;** **AS -** Activated sludge**;** **CW -** constructed wetlands**;** **RBC -** Rotating biological reactors**;** **UASB -** Upward-flow anaerobic sludge blanket**;** **MBRs -** Membrane biological reactors**,** **SBR** - Sequencing Batch Reactors**;** **MBBR -** Moving Bed Biofilm Reactors**;** **AC –** Activated carbon**;** **UF –** Ultra filtration**;** **MF** – Membrane filtration; **AOP –**Advanced oxidation process**;** **RO –**Reverse Osmosis

1.3 Selecting wastewater treatment technologies

The selection of wastewater treatment technologies is often model based and, takes into account a number of factors which include economic, environmental, technical, and social issues (Arroyo and Molinos-Senate 2018). In South Africa, the selection has been largely biased toward biological treatment with a very small fraction being other types of wastewater treatment technologies (Figure 1.1). A lot of variations exist within biological treatment technologies, with some being simple and economic, though with a large land footprint, e.g., waste stabilisation ponds and advanced integrated wastewater pond systems. Others are more widely accepted mechanic types, which are often costly to operate, for example activated sludge systems. There is, therefore, a great need to select appropriate technologies which address the social, economic, environmental, and technical needs of a community where it is located.

1.4 The Advanced Integrated Wastewater Pond System

The Advanced Integrated Wastewater Pond Systems (AIWPS®) was developed by Professor Oswald and co-workers in the 1950s. This passive yet robust system uses the principles of fermentation, photosynthesis, and microbial metabolism to remediate wastewater producing a good quality effluent with reuse potential (Banat et al. 1990; Oswald 1995; Downing et al. 2002). The AIWPS® was fully established after Professor Oswald was approached and asked to suggest an innovative wastewater treatment technology for the City of St. Helena. During that time, the most common wastewater treatment technologies were the trickling filter, activated sludge, and waste stabilisation ponds. The disadvantages associated with these systems are they were expensive to build and operate, they were prone to odour, and they became less efficient with time. In his thoughts, Oswald saw it befitting to develop an economic alternative wastewater treatment system that would combine methane, fertiliser, or feed production with efficient wastewater treatment without odour problems. This then led to the development of the AIWPS, which consists of four ponds in a series. The first pond in the series is a facultative pond that houses an anaerobic pit at the base, this is followed by a shallow paddlewheel mixed high rate pond, and the third pond in the series is a settling pond which removes algae. The last pond in the series is the maturation pond, which provides additional treatment and storage (Oswald and Asce 1990; Rose et al. 2007). AIWPS® is categorised into two designs, generations one and two. First generation AIWPS® treat wastewater to standards suitable for irrigation, and the HRAOPs are mixed using mechanical means. Second generation AIWPS® HRAOP contents are mixed using a paddlewheel and this system produces an effluent suitable for discharge and unrestricted use. Second generation AIWPS® also offers recovery, harvest, and utilisation of methane produced in the AFP and biomass produced in the HRAOP (Green et al. 1995). The term AIWPS® is commonly used in the USA, similar systems which include integrated algae pond systems (IAPS), enhanced pond systems (EPS), advanced pond system (APS) and microalgae wastewater treatment plant model (ALGA) shown in Figure 1.3 have been constructed and studied over the past 70 years (Grönlund et al. 2004; Craggs et al. 2004; Rose et al. 2007; Craggs et al. 2015). Extensive studies on AIWPS® and similar systems have shown that when properly designed and adequately managed this algal technology offers a simple, reliable, affordable, effective, sustainable and nuisance free wastewater treatment option (Oswald and Asce 1990; Banat et al. 1990; Tadesse et al. 2004; Craggs et al. 2015; Young et al. 2017).

Figure 1:3: Examples of AIWPS and similar systems located in the USA, South Africa, and New Zealand. CA = California, NZ = New Zealand, ZA = South Africa. AIWPS = Algal Integrated Wastewater Pond System, IAPS = Integrated Algal Pond System, EPS = Enhanced Pond System. AIWPS has been implemented at pilot, demonstration, and full commercial scale.

Research on the few available AIWPS and derivative systems suggests that it is an effective technology capable of treating wastewater, producing an effluent of secondary/ partial tertiary water quality (Rawat et al. 2011). High removal rates of standard water quality parameters irrespective of technology location and wastewater origin have been observed. Efficiencies in the 90th percentile for biological and chemical oxygen demand, *Escherichia coli* removal, and varying nutrient stripping rates have been previously reported (Tadesse et al. 2004; Banat et al. 1990; Grönlund et al. 2004; Wells et al. 2005).

An advanced facultative pond (AFP) can operate for years without the need for sludge removal (Oswald 1991). This is attributed to the low up flow velocity, which allows for the sedimentation of most solids and stable methane production, which occurs in the advanced facultative ponds. In IAPS carbon integration into methane formation in the AFP, reduces the need for daily sludge handling and disposal typical of activated sludge systems and waste stabilisation ponds (WSP). Conventional ponds and activated sludge systems generate large amounts of sludge and need to be de-sludged often. In WSP accumulation of sludge also results in a reduction in pond volume and treatment efficiency, odour production, the need for sludge removal, and ultimately reduction in pond life expectancy. Unlike WSP, the reduced sedimentation rates and the algae at the aerobic layer of the AFP aid in odour control (Cowan and Render 2012; Green et al. 1995).

When compared to WSP, less evaporative losses are experienced in the HRAOP unit due to the shorter hydraulic retention time (HRT) and the smaller surface area. Ultimately this results in more water being recovered for reuse (Buchanan et al. 2018).

One of the single most attractive features of AIWPS is low energy consumption. When properly or commercially designed, water flows from one pond to another by gravity, eliminating the need to pump. On a commercial scale, motors from the head of works and paddlewheels become the only component that utilises energy. Paddlewheel mixing is vital in HRAOP as it prevents sedimentation of algae and microalgal bacterial flocs (MaB-flocs), laminar flows, and oxygen stratification. The gentle mixing facilitates exposure to light at alternating intervals, thus creating favourable conditions for microalgae production (Cowan and Render, 2012; Rogers et al. 2014; Sutherland et al. 2014). It is estimated that HRAOP paddlewheels require 90% less energy than mechanical aeration to achieve optimum mixing (Craggs et al. 2014). Low energy consumption also indirectly offsets greenhouse gas emissions (Gupta and Singh, 2012).

AIWPS has been promoted as a more economic wastewater treatment option to conventional systems due to the lower costs of construction and low maintenance and operational costs associated with the system (Oswald et al. 1957; Craggs et al. 2015). Conventional WWTPs are expensive to build, operate, and maintain. The features which make IAPS a cheaper option to operate are; the use of a less energy intensive paddlewheel to provide aeration, infrequent de-sludging and cheaper construction costs (Banat et al. 1990; Green et al. 1995; Young et al. 2017).

Although IAPS are smaller than conventional ponds, one major disadvantage of AIWPS is the large footprint required for implementation at scale. The size requirement for AIWPS is massive, and this limits technology implementation to certain areas (Young et al. 2017). Advocates of algae technology propose that this can be overlooked because this system not only effectively treats water at a low cost but provides profitable and sustainable water reuse, energy and food opportunities (Oswald 1995; Craggs et al. 2014; Mehrabadi et al. 2015; Acién et al. 2016). However, technology innovations are still required to reduce system size. Final effluent produced by IAPS is often characterised by consistently high chemical and biological oxygen demand and total suspended solids. This has been attributed to apoptosis, extracellular polymeric substances, and inefficient biomass removal, thus suggesting the need for additional treatment (Jimoh and Cowan, 2017; Rose *et al.*, 2007). Although some regulators have provision for residual algae (Tadesse et al. 2004), the installation of other phases of sewage treatment ensures that regulatory standards for irrigation and discharge are met. IAPS also requires the implementation of a tertiary unit to ensure compliance with wastewater discharge limits (García et al. 2006; Mambo et al. 2014). This is a disadvantage because the incorporation of a tertiary treatment unit results in an increase in land footprint and cost. Maturation ponds, rock filters, slow sand filters, advanced oxidation, and reverse osmosis have been successfully administered as tertiary systems (Downing *et al.*, 2002; Jimoh, 2017; Mambo *et al.*, 2014). More compact options which do not significantly increase land requirement such as advanced oxidation and reverse osmosis might be attractive considerations. Although these options could result in significant increases in costs, Downing and co-workers (2002) concluded that because the running costs of IAPS alone are low, it allows for the use of more vigorous tertiary reclamation techniques that would be otherwise costly to use in conjunction with conventional activated sludge secondary treatment.

Table 1:3 Advantages and disadvantages associated with AIWPS

Advantages	Disadvantages
• Effective water treatment system	• Large land footprint
• Low energy costs	• Longer hydraulic retention time
• Economic	
• Little to no sludge handling	
• Less water loss by evaporation	
• Less odour	
• High life expectancy	

Oswald (1995) also predicted that in addition to water treatment, these ponds would become vital for the provision of food and energy both directly and indirectly. To date, a few

AIWPS® exist all over the world either as a full AIWPS® or as variations. Like many passive NBS, AIWPS® remains a scarce, underappreciated, and underutilised technology with a lot of potential to alleviate the water, food, and energy challenges in this 21st century.

1.5 Integrated Algae Pond Systems (IAPS)

Integrated algal ponding systems (IAPS) are a derivation of AIWPS®. These consist of the first three ponds of the AIWPS® and were developed as part of a technology transfer initiative, in association with the developers of the AIWPS® funded by water research commission (WRC) (Rose et al. 2007). The first IAPS was built in 1994 at the institute for environmental biotechnology (EBRU) in Grahamstown, South Africa. The core of the partnership between WRC and Rhodes University was to inform the South African water sector of the benefits of using this method to treat municipal waste water.

A typical IAPS process flow begins with a primary treatment unit. The primary unit employed is an advanced facultative pond (AFP), which houses a fermentation pit. This pond is utilized as the leading unit due to its consistent high efficiency and its ability to resist process changes (Tadesse et al. 2004). Bulk COD removal is experienced at this stage due to sedimentation and fermentation. The resultant effluent then flows into the high rate algal oxidation ponds (HRAOP). These are shallow, open raceway ponds whose contents are mixed with a paddlewheel at a low velocity. This microalgal-bacterial unit doubles as a secondary and partial tertiary treatment centre (Rawat et al. 2011), stripping the effluent of organic carbon, nutrients, and pathogens. All this is achieved because of the symbiosis which exists between the algae and the bacteria aided by design, operational conditions, and climate conditions. Algae removal occurs in algae settling ponds (ASP) (Cowan and Render, 2012). The efficiency in this unit determines the quality of the final effluent as algae constitute 50-65% of the organic load (Tadesse et al. 2004; Mara et al. 1983) and the efficiency of biomass harvest yield.

Research on IAPS spans over 25 years in South Africa, and results indicate that this system provides an effective and sustainable treatment option for the reclamation of wastewater. Plant performance efficiencies report reductions of 87% COD, 76% phosphate, 55% nitrogen, and 99.999% *Escherichia coli* (Wells et al. 2005). The effluent produced by the Belmont Valley demonstration scale plant is characterised by consistently high chemical oxygen demand (COD) and TSS (Cowan et al. 2016; Jimoh 2017). Other standard water parameters, as set in the Water Act (1958), are however often within discharge limits as summarised in Table 1.3.

Table 1:4: A summary of water quality data of the final effluent from the Belmont Valley WWTW IAPS obtained from earlier studies 2002 – 2016 (Cowan et al. 2016; Jimoh 2017).

Parameter	Water quality of effluent				
	2016	2013-14	2012-13	2006	2002
pH	8.1 ± 0.2	9.1 ± 1	9.4 ± 1	9.5	10.5
Dissolved oxygen (mg. L^{-1})	7.3 ± 0.7	5.7 2	5.5 ± 2	NA	NA
Electrical conductivity (mS m^{-1})	125.1 ± 5.6	112 ± 14	108 ± 19	NA	NA
COD (mg. L^{-1})	81.3 ± 6.7	66 ± 12	72 ± 13	60	65
TSS (mg. L^{-1})	28.6 ± 3.4	35 ± 14	34 ± 13	60	60
Nitrate/nitrite-N (mg. L-^{1})	11.2 ±1.9	2.3 ± 2	12 ± 1	15	17.5
Ammonium-N (mg. L^{-1})	1.9 ± 0.4	2.6 ± 1	2.9 ± 1	1.5	7
Ortho-phosphate (mg. L^{-1})	2.3 ± 0.2	4.3 ± 2	5.3 ± 2	5.4	2.6
Faecal coliforms (CFU 100 mL^{-1})	$1.6 \times 10^5 \pm 0.3 \times 10^5$	> 1000	> 1000	>1000	>1000
Reference	(Jimoh 2017)	(Cowan et al. 2016)			

NA= Not available

IAPS presents opportunities to address future constraints at the water-energy-food nexus (Oswald 1995). In addition to the treatment of wastewater, IAPS has the ability to generate two additional product streams viz. biogas and biomass.

Biogas production takes place in the AFP of an IAPS. A theoretical output of 40 m^3 biogas$\cdot d^{-1}$ for a 600 person equivalent (PE) plant is possible; however, actual measurements point to only 28% of theoretical yield being produced (Cowan et al. 2016; Green, Bernstone, et al. 1995).

Biomass in IAPS is mixed liquor of suspended solids comprising algae, bacteria, and other microbes. Biomass is produced in high rate oxidation ponds as flocs – known colloquially as MaB-flocs or microalgal-bacterial aggregates (MABAs) (Vulsteke et al. 2017; Quijano et al.

2017). Mixed liquor suspended solids (MLSS) are the biomass component typically produced in the aeration tank during an activated sludge water treatment process. In brief, this involves the addition of bacterial granules to the aeration basin, and in the presence of oxygenated air and agitation these bacteria abstract nutrients from the water, grow and produce substantial biomass. Similarly, in high rate algal oxidation ponds, the MaB-flocs abstract nutrients from wastewater in the presence of photosynthetically produced oxygen and sunlight to produce sludge or a bio-mass. Consequently, the mixed liquor suspended solids are similarly produced to MaB-flocs. In high rate algal oxidation ponds, a symbiotic relationship exists between microalgae and bacteria. Bacteria facilitate the oxidation of organic matter to produce carbon dioxide, ammonia, and phosphate, which are then utilised by microalgae during photosynthesis. The oxygen released during photosynthesis is, in turn, utilized by bacteria. It is this interaction, aided by continual paddlewheel mixing in the HRAOP and the production of extracellular polymeric substance (EPS), which results in the bio-flocculation of these microbes hence the formation of MaB-flocs (Jimoh et al. 2019). Other studies make use of chlorophyll measurements and mixed liquor volatile solids (MLVSS) to quantify biomass so as to include only the algal and organic component of the biomass produced, respectively (Park et al. 2013; Sutherland et al. 2014). However, for the purposes of estimating the amount of biomass produced in HRAOP and dried in drying beds, MLSS is adequate, then depending on the final use of the biomass, that is when only utilizing the algal component or the organic component then further quantification may be carried out.

The biomass produced in high rate algal oxidation ponds is, in essence, equivalent to the sludge. In any traditional waste water treatment plant, this sludge would require handling. Sometimes this involves drying and dumping and, in some cases, palletisation and transfer to landfill. In a situation where high rate algal oxidation ponds are the reactors producing the biomass, it is the beneficiation of this sludge into value added products that is crucial to the success of the technology. These MaB-flocs may serve as a feedstock for the production of biogas, bioethanol, biohydrogen, biodiesel, and bio-crude oil (Cowan and Laubscher, 2010; Craggs et al. 2014; Oswald & Golueke, 1957; Park and Craggs, 2010). MaB-flocs can also be used in the production of fertiliser and as an animal feed (Cowan and Mlambo, 2015; Oswald, 1991, 1995).

Algae technologies like IAPS combine wastewater treatment with the production of sustainable energy sources and food, thereby reducing the total environmental impacts of the system. However, these benefits can only be fully realised with the commercialisation of the technology. Australia and New Zealand have had success in the implementation, validation, and commercialisation of IAPS technology (Craggs et al. 2012; Sutherland et al. 2017; Fallowfield et al. 2018). However, the implementation of this system at a large scale in Africa has been limited, and the inclination is still towards unsustainable mechanical technologies. The possible cause for this status quo is the perception that mechanical systems perform better with less risk, a general lack of inclusion of the technology by waste management planners and engineers has also halted the upscaling of this technology, and lastly, a general lack of understanding and awareness of the benefits that the system can offer has also hammered technology progress.

1.6 Factors affecting IAPS performance

The structure and process flow of IAPS has undergone no dramatic changes since its inception, with derivatives of the system being utilised worldwide (Tadesse et al. 2004; Mahadevaswamy and Venkatamaran 1986; Banat et al. 1990; Gröndlund et al. 2004; Wells et al. 2005). Optimization strategies have been developed to improve water quality and biomass yield; however, IAPS remains an underused technology (Park et al. 2013; Sutherland et al. 2014a; Sutherland et al. 2014b, Sutherland et al. 2016).

Like all other wastewater treatment systems, operation and management protocols should be followed to ensure optimum plant performance. For the Belmont Valley IAPS, two high rate algal oxidation ponds are operated in series; the first pond has a hydraulic retention time of 2 days, and the second pond has a hydraulic retention of four days. This operating protocol was adopted to improve water quality. When the ponds were operated as single units, the effluent produced did not meet standards of discharge with regards to phosphates and faecal coliforms counts. Operating the ponds in series also improved the MaB floc structure and settleability in the second HRAOP. (Rose et al. 2007). For the Belmont Valley demonstration scale IAPS, when inefficient hydraulic retention time was experienced in the HRAOP due to improper pipe installation, it caused a spike in the faecal coliform count (Mambo et al. 2014).

Seasonal and diurnal changes occur within IAPS, with effects being more prominent in the HRAOP (Sutherland et al. 2018a; Tadesse et al. 2004). Biomass yield, disinfection, and nutrient removal experience seasonal variations, with higher productivities and better performance being noted during the hotter summer months (Al-Shayji et al. 1994; Sutherland et al. 2018b; Rose et al. 2007). Increasing winter time HRAOP HRT has also been shown to improve water quality during the cold months (Rose et al. 2007) however, the study of García and coworkers (2006) highlighted that although daily variations occur in HRAOP the changes are insignificant hence it would not affect the ability of the system to treat wastewater efficiently to a consistent standard.

Changes in algae dominance occur within the ponds, and this has been attributed to alterations in water quality parameters (Sutherland et al. 2017). At demonstration scale, the change of dominance from *Pediastrum*/*Scenedesmus* to *Cyclotella* affected water quality. Dominance by *Cyclotella* increased the concentration of NH_4-N in the final effluent to levels that exceeded compliance. The authors, however, acknowledge that this increase cannot be entirely attributed to a change of dominance (Cowan et al. 2016).

Light availability has a direct impact on biomass productivity and disinfection within the HRAOPs (Davies-Colley et al. 2003; Sutherland et al. 2015). Light is the driving force of photosynthesis, and its intensity is a direct measure of the rate of algal growth rate. An increase in light intensity increases biomass productivity, provided nutrients are not limiting, and the temperature is optimal. The maximum light intensity required for growth is species dependent; however, values greater than 200 µmol. m^{-2} s^{-1} have an inhibitory effect on most algae. An increase beyond this point results in photoinhibition and cell death. As photosynthesis increases, so does algal concentration, and this increases dark cycles experienced by cells within the HRAOP, thus decreasing the amount of light available between the pond (Richmond 2004; Park et al. 2011). Optimal light required is catered for in the HRAOP design as the ponds are shallow and include a paddle wheel. The gentle mixing facilitates exposure to light at alternating intervals. For disinfection to occur within the HRAOP sunlight interacts with other factors which include pH and dissolved oxygen to enhance the decline if faecal indicators (Davies-Colley et al. 2003; Craggs et al. 2004)

Temperature affects biomass production within the HRAOP however, the effect of temperature is also species dependent with optimum temperature ranges from 15 to 35 degrees (Bouterfas et al. 2002). At low temperature algae are prone to induced photo inhibition, as temperature increases towards optimum an increase is noted in algal productivity as this is concomitant with an increase in light intensity, beyond optimum photorespiration and respiration are favoured. The effect of temperature is notable only when CO_2, light, and nutrients are not limiting (Bouterfas et al. 2002; Vonshak *et al.*, 2001; Pulz, 2001). Although possible in closed systems, temperature regulation and manipulation has not been achieved with IAPS. (Slegers et al. 2013) suggest summer and spring time operation of

IAPS if the system is operated for the purposes of biomass production; however where the main purpose of IAPS is wastewater treatment changes in operation conditions to optimise other influential factors and a tertiary unit implementation should be sufficient to meet discharge regulations.

Adequate carbon supply is essential in ensuring optimum biomass and photosynthetic productivity, nutrient removal, and pH control (Benemann 2003; Park and Craggs 2010). CO_2 in IAPS is supplied by fixation from the air and bacterial metabolism. Even so, carbon abatement is inevitable as photosynthesis progresses due to its utilisation and pH induced decline, which shifts inorganic carbon species equilibrium towards the production of HCO_3^- and CO_3^{2-} (Sutherland et al. 2015). The addition of CO_2 has been demonstrated as a practical option to curb the effects of CO_2 limitation, and its application is gaining momentum, especially in full scale systems. The most sustainable and cost-efficient sources of CO_2 are flue and exhaust gas. Carbon capture from flue gases will not only play a role in reducing the effects of climate change but can also provide additional revenue for IAPS plants as carbon capture incentives are becoming available (Raven 2017; Acien et al. 2012).

HRAOP are biological systems and these function within a narrow pH range. A wide range of species are found within this pond, and each microbe has its own pH optima. Whilst some are optimal at pH 8, e.g. *Chaetoceros sp* and *Chlorella sp*, others can function at pH 10, e.g., *Ankistrodesmus sp* (Park et al. 2011). A pH above or below optimum has a negative effect on algal productivity. High pH values exceeding 11 are prone in HRAOP during the day. These elevated pH levels interfere with the symbiosis within the pond, impact nutrient removal and assimilation and affect photosynthesis. CO_2 utilisation during photosynthesis causes an increase in pH, which is further enhanced when no form of carbon augmentation is remediated (Larsdotter 2006). High pH > 9 favour the production of HCO_3^- and CO_3^{2-} thereby reducing the amount of available $CO_{2,}$ thus halting photosynthesis for CO_2 dependent species (Sutherland et al. 2015). Additional CO_2 cannot be supplied by bacteria oxidation as aerobic bacteria inhibition occurs at a pH of 8.3, and this consequently affects nutrient supply required for algal growth(Park and Craggs 2010). Nutrient uptake by assimilation into biomass is also affected at high pH as ammonia stripping, and auto-flocculation take precedence. Ammonia stripping also increases the concentration of free ammonia. At high free ammonia levels, algal growth is inhibited; however, this has been shown to vary between species (Ogbonna et al. 2000). At elevated pH, pH induced flocculation may occur in some species; although this is an advantage in the ASP, it affects photosynthesis light and nutrient assimilation (Larsdotter 2006; Sutherland et al. 2015). pH can be controlled by increasing depth, adding carbon dioxide and increasing organic load (Sutherland et al. 2014; Larsdotter 2006).

Nitrogen and phosphorus are key nutrients required for microalgae growth; other macronutrients and micronutrients are also required to avoid nutrient limitation. Biomass productivity has been shown to be associated with an increase in nitrogen and phosphorus with optimum levels exceeding 25 and 2 mg. L^{-1}. Although a wide range of N : P ratios are documented showing species dependents, a nitrogen supply ratio of NO_3^{2-}N: NH_3^{+}-N of 3: 1, gave the highest production rates (Mostert and Grobbelaar 1987).

Influent dilution can also be adopted to reduce the organic load in the AFP. This has been successfully implemented at a South African plant using final effluent to dilute influent wastewater, and this resulted in effluent with improved quality however incorporation of the tertiary treatment unit was required to ensure that discharge regulations were achieved with regards to faecal coliforms and COD (Cowan et al. 2016; Mambo et al. 2014). High amounts of rainfall increase the amount of water being pumped in the wastewater treatment plant. This results in decreased HRT and dilution of wastewater (Rose et al. 2007; Buchanan 2014).

The contamination of HRAOP by zooplankton has been shown to occur within the HRAOP. Zooplanktonic grazing can be detrimental to the quantity of biomass within the HRAOP, and it reduces nutrient removal; however, the presence of grazers improves settleability. Factors that induce the establishment of zooplankton include neutral pH, high food ratio, and change of dominance (Montemezzani et al. 2016). Zooplanktonic populations can be controlled using physical and chemical methods (Montemezzani et al. 2015); unfortunately, this can increase the operational costs of the WWTW.

The effectiveness of the ASP is a function of how efficient the HRAOP is. The simple gravitational method in this unit is desirable as it is cheap; however, its major limitations are species composition, hydraulic retention time, algae size, charge, and density (Park et al., 2011; Park et al. 2013).

The above factors which affect performance show that unit operation audits are required so as to fully optimise the plant as there is no single fit for all plants. It should be noted; however, that implementation of optimisation strategies increase in cost is likely in most cases.

1.7 The bio-refinery potential of IAPS

Bio-refineries have been in existence for a long time and have been applied in a number of industries. Biorefining is defined as the sustainable processing of biomass into a range of market value products (e.g., food, feed, materials, and chemicals) and energy. IAPS have the potential of being a refinery because, in addition to purifying water, it generates biomass with volarisable potential. However, in order to become a refinery, the most sustainable and economical route of biomass processing should be utilised. Knowing the structure and composition of the biomass produced is essential for biomass processing. Biomass can be converted into a number of products shown in Figure 1.4 using various technologies and biochemical conversions. The factors that have been shown to influence the conversion route applied are quantity of biomass, moisture content, protein, lipid and carbohydrate composition (Hirano et al. 1997; McKendry 2002; Yen and Brune 2007; Bohutskyi et al. 2018)

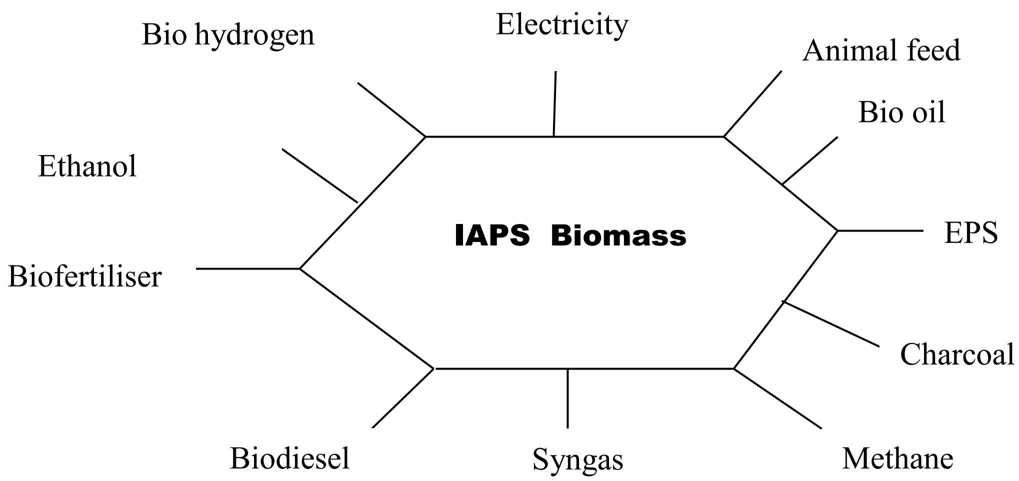


Figure 1:4: Examples for possible platforms in IAPS bio refinery system (Brennan and Owende 2010)

As mentioned above of as much importance as quality, the quantity of biomass produced will determine if becoming a biorefinery would be sustainable. Biomass productivity is generally low in open raceways (Norsker et al. 2011), and productivity is affected by several factors, which include climate, algae carbon source, seasonal changes, and mixing efficiency (Park et al. 2011). Despite the low biomass production, HRAOPs are still the production system of choice when considering a biorefinery due to their low-cost investment with the assumption that the system has already been built for wastewater treatment. Biomass is produced as MaB-floc in the HRAOP of IAPS (Jimoh et al. 2019). A lot of methods have been used to quantify biomass and biomass productivity (Al Shayji et al. 1994; Park et al. 2013; Sutherland et al. 2014; Jimoh and Cowan 2017). Algae productivity is defined as the mass of algae produced per day in a unit area. A theoretical maximum algae productivity value of 200 grams. biomass m^{-2} day is given based on upper limits of production and best case scenarios (Weyer et al. 2010) while a modest productivity is calculated from average solar to give a value of only 13% of the theoretical maximum (Park et al. 2011). Values for algae productivity based on experimental measurements provide in literature range from 20 - 110 tonnes. $ha^{-1}year^{-1}$ (Slegers et al. 2013), and this discourse is in part due to an unstandardized method of determination.

1.8 Process Audit

A process audit is defined as —a audit of a process against agreed upon requirements" (Russell 2009). It entails the evaluation of results to determine whether a process is operating within set limits to achieve process objectives. It also verifies whether the results are achieved within an efficiently and effectively managed process. Process audits may be carried out in order to:

- Assess the effectiveness and efficiency of a process
- Verify compliance to standards
- Trouble shoot problems which may arise during the process (Russell 2009; Hoyle 2001)

Process audit techniques include the use of tree diagrams, turtle diagrams (Figure 4.1), flow diagrams and checklists.

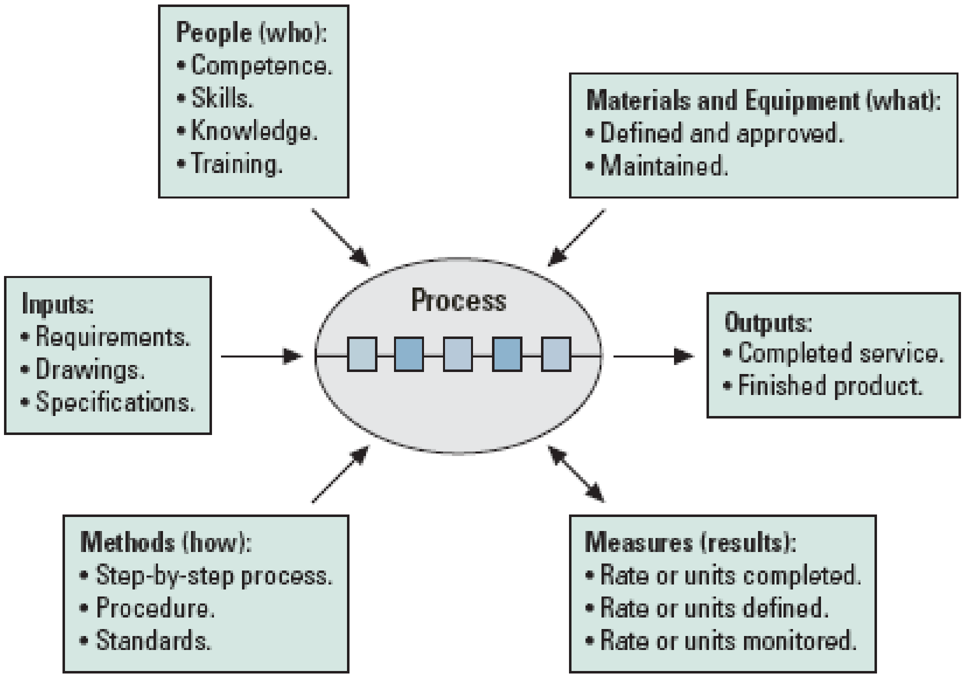


Figure 1:5: Example of process audit turtle diagram.

1.9 Background

Construction on the Belmont valley integrated algae pond system began in 1994, in 1996 the plant started operating and was officially opened in 1997 as a research and reference plant. A lot of research which has spanned the plant's entire life span has defined the plant treatment efficiency, biogas output, biomass quantity, and composition, accounted for consistently high total suspended solids and chemical oxygen demand and recommended suitable tertiary treatment for this system (Potts 1998; Wells et al. 2005; Rose et al. 2007; Johnson 2010; Mambo et al. 2014; Cowan and Mlambo 2015; Jimoh 2017). Research by Kubura (2010) and Cowan and co-workers (2016) carried out on the Belmont Valley IAPS revealed that this 500 PE demonstration-scale system produced approximately 28 ML. y^{-1} treated water for recycle/re-use, 1880 kg $CH_4.y^{-1}$ methane-rich biogas, and >3 tonnes dry weight. y^{-1} biomass. The approximate biochemical composition of the microalgae bacterial biomass is protein 41.5% protein, 4.8% lipid, and 35.1% carbohydrate (Potts, 1998). Plant performance efficiencies are stated earlier and a summary of the final effluent water quality data from earlier research is shown in Table 1.3.

According to the wastewater Act (1998), effluent exiting a wastewater treatment facility should meet the “agreed” standards for either discharge to a water course that is not a listed water course or irrigation. The above mentioned IAPS differs from typical systems that have a tertiary component as it was introduced into South Africa primarily for irrigation of agricultural/horticultural crops in the peri-urban space. Hence the final effluent is characterised by high TSS and COD (Mambo et al. 2014). South African guidelines for discharge and irrigation and previous research results on the Belmont Valley IAPS (Potts 1998; Wells et al. 2005; Rose et al. 2007; Johnson 2010; Mambo et al. 2014; Cowan and Mlambo 2015; Jimoh 2017) then provides a basis of agreed requirements in regards to treatment efficiency, final water quality, biomass quantity, and quality and gives insight on the effectiveness of the process.

Decades later, efforts towards commercialisation of the technology have failed, IAPS has remained at the demonstration scale and has become a "wasted technology" (Cowan and Laubscher 2019). Despite its good performance efficiencies, projects for the construction and implementation of a full-scale IAPS have failed despite the availability of full/ partial funds.

Although a vast amount of research has been carried out since the commissioning of the Belmont Valley IAPS, this process audit will help to fully define the capabilities of this and other IAPS and provide more recent performance data.

1.10 Aims and objectives

The aim of the project was to develop a process audit framework for IAPS during the commissioning and optimisation phase of process implementation.

Objectives include:

1. Assess compliance of final effluent to general limits of discharge.
2. Assess plant performance efficiency and the effectiveness of each IAPS treatment unit (AFP, HRAOPs and ASPs) in treating effluent.
3. Investigate biomass composition, productivity, and settleability.
4. Investigate the effect of plant management on plant performance and troubleshooting.
5. Analysis of costs incurred for plant maintenance and operation during the period 2009-2017

Chapter 2 : Materials and Methods

2.1 IAPS Components and Operation

The demonstration-scale IAPS facility used in this study was constructed adjacent to the Belmont Valley Wastewater Treatment Works (WWTW) (33W 190 07″ South, 26W 330 25″ East). The plant was built to treat wastewater for 500-person equivalent (PE), with an estimated maximum organic load of 40 kg day^{-1} and a continuous inflow rate of 75 kL d^{-1}. The system comprises of a series of synergistic ponds whose operational design is based on hydraulic retention time (HRT). These are; a 225 kL, in-pond digester (IPD) or fermentation pit (FP) submerged 6 m below a primary facultative pond (PFP) with a volume of 1500 kL, two 150 kL high rate algal oxidation ponds (HRAOP), two 19 kL algae settling ponds (ASP) and four drying beds with a surface area of 10 m^2 each (Cowan and Render 2012). The configuration and flow diagram of the pilot scale IAPS at Belmont are shown in Figure 2.1 below.

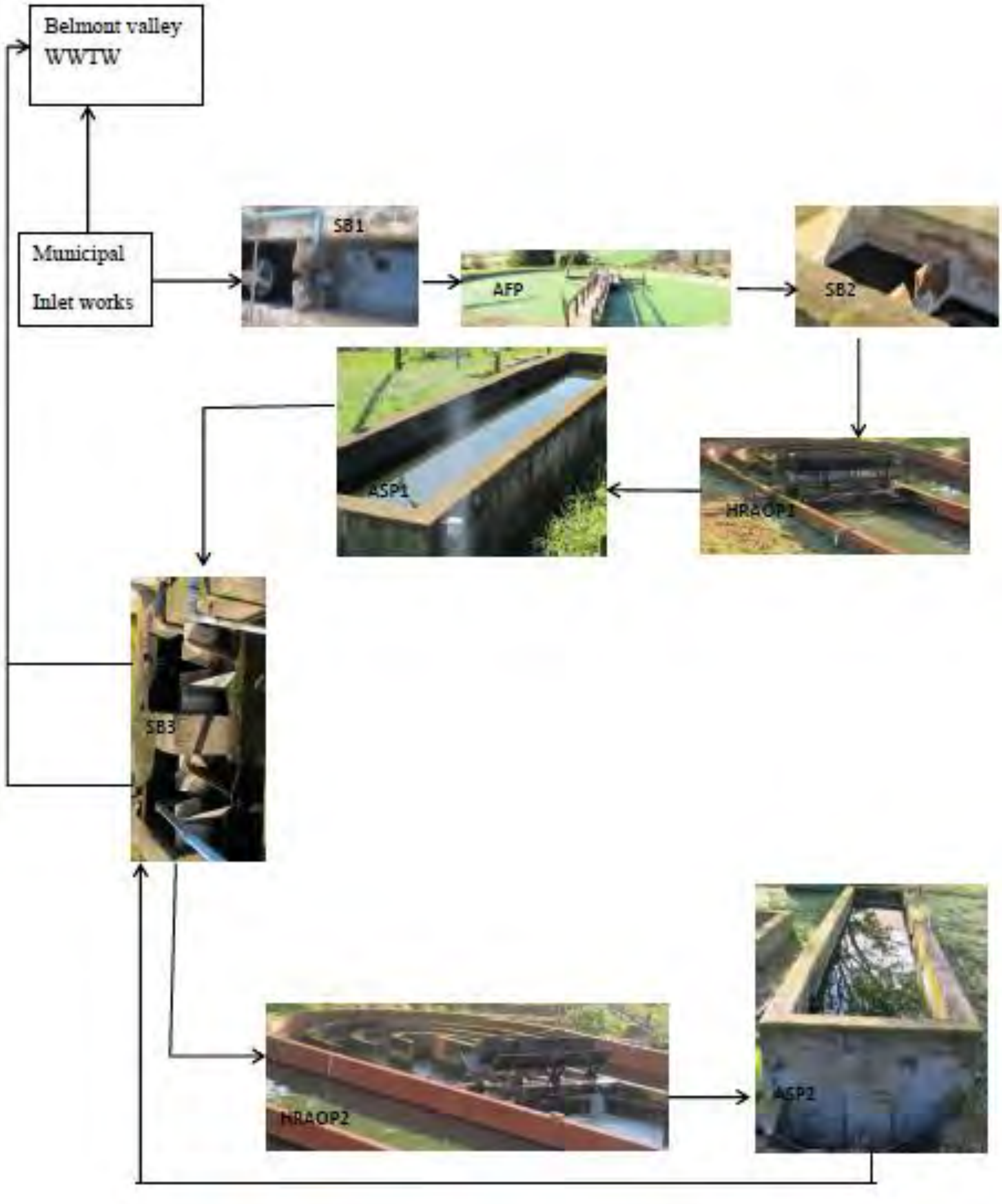


Figure 2:1: Belmont Valley IAPS process flow for wastewater treatment. SB=splitter box; AFP = advanced facultative pond; IPD = in-pond digester; HRAOP = high rate algae oxidation pond; ASP = algae settling pond

The flow rate and the volume of the treatment unit form the basis for the theoretic estimation of hydraulic retention time within the system, according to Equation 2.1. The resultant HRTs are shown in Table 2.1

$$HRT = \frac{Volume\ of\ treatment\ unit\ \ (kL)}{flow\ rate\ (kL\ d^{-1})}$$ ....Equation 2.1

Table 2:1: Summary of unit operations hydraulic retention time

Treatment unit	Volume (kL)	Flow rate ($kL \cdot d^{-1}$)	HRT (d)
Fermentation pit (FP)	225	75	3
Primary facultative pond (PFP)	1500	75	20
High rate algal oxidation pond 1 (HRAOP 1)	150	75	2
Algae settling pond 1 (ASP 1)	19	75	0.253 (6.08 h)
High rate algal oxidation pond 2 (HRAOP 2)	150	37.5	4
Algae settling pond 2 (ASP 2)	19	37.5	0.51 (12.16 h)

EBRU receives flow after entry into the Belmont Valley Wastewater Treatment Works inlet, which traditionally has grit removal. The first point of entry for the incoming sewage within IAPS, once it has passed through the first splitter box (SB1) servicing the IAPS, is the bottom of the fermentation pit, which is located about 6 m below the water level of the PFP. The fermentation pit has a depth of 4.5 m and maintains an up-flow velocity rate of 1.5 m. d^{-1}. This allows for the settling of suspended solids, helminth ova, and other parasite cysts. The total HRT in the FP and the surrounding PFP is 3 and 20 d, respectively. Together, the IPD and PFP are termed advanced facultative pond (AFP). The AFP surface was covered by a layer of duckweed throughout the sampling period. Its presence resulted in anaerobic conditions throughout the pond length. The resultant effluent then decants and flows by gravity into the first HRAOP, where it is retained for a further 2 d. After two days, the HRAOP effluent flows into the first ASP, which has a HRT of 6 h. The treated effluent flows from the first ASP into a second splitter box (SB 2), where approximately 50 % of the volume is pumped into a second HRAOP at a rate of 37.5 kL d^{-1}. The residual volume drains into a concrete collection sump. A retention time of 4 d is designated in the second HRAOP.

Microalgae bacterial biomass in both HRAOPs is kept in suspension by paddlewheels, which mix the effluent and propel it around the pond at a velocity of 30 cm. s^{-1}. From the HRAOP, water flows into a second ASP, which has an HRT of 12 h. The effluent leaving the second ASP decants into a second splitter box where it mixes with the residual volume of effluent from the first ASP. This mixed effluent drains into a concrete collection sump where it is pumped to the Belmont Valley WWTW inlet via the first splitter box. The settled biomass slurry from each of the ASPs is pumped onto the drying beds containing coarse gravel. Water drains through the gravel bed into a concrete collection sump, where it is disposed of back to the Belmont Valley WWTW inlet. The algae biomass is sun-dried, and the dried algae is raked of the drying beds and composted.

During the first three months of sampling, the plant was operated in recycle mode. During this period, no raw effluent was fed into the plant; instead, effluent from the ASP1 and 2 was channelled back into the AFP. However, when AFP water levels dropped below the pipe, raw water was pumped into the AFP. The plant started operating in continuous mode from the 17th of July 2018.

2.2 Culture medium for microbial analysis

m-Fc agar, Chromocult® Coliform Agar.

2.3 Chemicals and reagents

Ammonium (N-NH_4^+) 14752 Merck KGaA, Darmstadt kit, Nitrate (NO_3^-) 14773 Merck KGaA, Darmstadt kit, Phosphate (PO_4^-) 14848 Merck KGaA, Darmstadt kit, High and low range COD digestion solutions, Sulfuric acid reagent, Potassium hydrogen phthalate standard

2.4 Water sampling and determination of quality

2.4.1 Sampling. Composite samples were collected from the raw inlet point (first splitter box), the V-notch weir between the AFP and the first HROAP, in both HRAOPs at the discharge points and both ASPs at the weirs. These sampling points were selected to allow for the evaluation of both plant and unit process performances. Calculation of percentage removals for COD and TSS was done according to equation 2.2 below (Spellman 2013).

$$\%\ \text{Removal} = \frac{[\text{Influent Concentration} - \text{Effluent Concentration}]}{\text{Influent Concentration}} \times 100 \ldots\ldots\ldots \text{Equation 2.2}$$

Composite samples were collected weekly over a period of 9 months (May 2018- February 2019). Sampling was carried out between 0800 h – 1700 h. This sampling period and sampling time allowed for the incorporation of both diurnal and seasonal changes experienced within the IAPS. Approximately 500 mL aliquots were collected diurnally between 0800-0900 h, 1200-1300 h, and 1600-1700 h into sterile, acid washed Duran Schott bottles, at each sampling point. Samples were refrigerated at 4 $^{\circ}$C immediately after sampling. At the end of the sampling period, samples from each time interval were combined. A 100 mL aliquot was poured into a sterile bottle; this was used for bacteriological analysis and all tests that required unfiltered samples. Another 100 mL aliquot was vacuum filtered (pore size 0.45μm), for determinations that required filtered samples. Samples were then stored at 4 $^{\circ}$C and analysed the next morning.

2.4.2 Analytical Procedures

General authorization limits applicable to South Africa for the discharge of effluent into a water resource and irrigation were established under section 39 of the National Water Act, 1998. The recommendations shown in Table 1.1 apply.

Physicochemical parameters which include temperature, dissolved oxygen, pH, and electrical conductivity, were measured on-site at each sampling time interval. Seasonal and daily variations have been observed previously in HRAOP and AIWPS™ although changes have been shown to be more prominent in oxidation ponds than facultative ponds (Tadesse *et al.*, 2004; Jimoh, 2017). The measurement of each parameter at each time interval, therefore, served as a process control to ensure that the respective profiles were observed. Electrical conductivity and pH are included in the wastewater discharge limits; hence their measurements were part of the water quality check routine. Temperature and dissolved oxygen were measured using an IP67 Combo (Water Quality Meter, China). Electrical conductivity was measured using an EC Testr11 Dual range 68X 546 501 detector (Eutech Instrument, Singapore), whilst pH was measured using a Hanna HI 8424 microcomputer pH meter (Hanna Instrument, Romania).

Bacteriological analysis was carried out using the spread plate method described in (Prescott and Klein 2002). Chromocult agar and MF-c agar were used for the enumeration of *E.coli* and faecal coliforms, respectively. Both agars were used to allow for comparison with metadata (Wells et al. 2005; Mambo et al. 2014; Jimoh 2017). The diluted sample was spread on agar plates prepared according to the manufacturer's instructions. Chromocult agar plates were incubated at 37 ˚C, whilst MFC plates were incubated at 45 ˚C for 24hrs prior to the enumeration of colonies. Colony forming units per 100 mL (CFU. 100 ml^{-1}) were estimated according to Equation 2.3.

$$cfu\ 100\ Ml^{-1} = \frac{(number\ of\ colonies \times dilution\ factor)}{volume\ of\ sample} \times 100 \text{……..Equation 2.3}$$

The parameters of total suspended solids (TSS) and chemical oxygen demand (COD) were analysed according to APHA method 2540 D and 5220D, respectively. For TSS, a known volume of sample was filtered through a pre-dried and pre-weighed Whatman glass microfiber filter (47 mm and 1.2 µm pore size (grade MGC obtained from Munktell Ahlstrom)**.** The filters were dried in an oven at 105 ˚C for 24 h. The TSS concentration was calculated according to Equation 2.4. TSS concentration was also used for other calculations, as will be explained below.

$$mg\ TSS.L^{-1} = \frac{(A-B)\times 1000}{sample\ volume\ (ml)} \text{………Equation 2.4}$$

Nutrients. Ammonium ($N\text{-}NH4^{+}$), nitrate ($NO3^{-}$), phosphate (PO_4^{-}) were measured according to the manufacturer's instructions following the 14752, 14773, and 14848 Merck KGaA, Darmstadt, Germany photometric methods test kits respectively. Standard solutions were prepared weekly for each test, and absorbance was read using a Thermo Spectronic Aquamate spectrophotometer (ThermoFisher Scientific, Waltham, MA). The resultant standard curve was used to interpolate values for the final concentrations (Appendix Figure A1-A4).

2.5 Biomass composition and productivity

2.5.1 Microscopy

The microscopic analysis of HRAOP2 water was carried out weekly. The pond water was microscopically examined using a Zeiss Axiostar plus light microscope (Carl Zeiss, Jena, Germany) at low (10×) and high (40×) resolution. Images were captured using a Canon PowerShot G12 (Canon Inc., Japan). The purpose of this analysis was to observe species composition, diversity, and presence of microalgal-bacterial flocs.

2.5.2 Biomass Productivity and Loss

TSS concentration, as described earlier, was used to estimate biomass concentration within the ponds. Biomass productivity is presented in kg. ha^{-1} d^{-1} and was calculated according to Equation 2.5 (Al-Shayji et al. 1994). Final effluent TSS was used to calculate the lost biomass. Therefore, actual settled biomass was calculated as the difference between HRAOP 2 biomass productivity and residual (unsettled) biomass in final effluent.

$P = 10 \times \frac{d}{t} \times n \times TSS$……………… Equation 2.5

Where: P is pond productivity (kg. ha^{-1} d^{-1}),

d = pond depth (m),

t = hydraulic retention time of the pond,

TSS = total suspended solids (mg. L^{-1})

n = algae ratio in the TSS (0.9-1.0, as estimated by Al-Shayji *et al.* (1994).

2.5.3 Source of metadata

Data from previous research and research that was carried out concurrently with this work was collected to provide the bacterial and biochemical composition of biomass. Bacterial composition was obtained from Jimoh (2017) and Masudi (2019). The collection of MLSS samples for the isolation of bacteria was carried out in HRAOP 2. Serial dilution of MLSS sample was carried out first then 1.0 mL aliquot of sample was spread on nutrient agar plates. Successive sub-culturing was carried out by streaking on new agar plates until a pure culture was attained. Pure isolates were then identified by genomic DNA extraction and pyrosequencing (Jimoh 2017; Masudi 2019).

The average lipid, carbohydrate and protein values were calculated from three research studies (Potss 1998, Johnson 2010; Sibelo 2019). Lipids were quantified using the Folch method (Johnson 2010) and SPV method (Sibelo 2019). Carbohydrates were quantified using modifications of the phenol-sulfuric-acid assay as described by Dubois (Johnson 2010), nitrogen free extracts (NFE) as a measure of the soluble carbohydrates (Potts 1998) and the sulfuric acid-UV method (Sibelo 2019). Protein was quantified using the Bradford protein dye-binding method (Johnson 2010) and micro-Kjeldahl protein analysis method (Potts 1988).

2.6 Plant management

The following management procedures were carried out during the course of the study in order to maintain optimum plant working conditions as set out in the Standard operating procedure (Cowan et al. 2016).

2.6.1 Flow measurements. Wastewater flow was estimated throughout the sampling period using the bucket and stopwatch method as described by (Spellman 2013). The time it took to fill a 10 L bucket was recorded, and the flow rate was calculated and is presented as kilolitres per day kL d^{-1}. Flow measurements were taken at the inlet point (SB 1), HRAOP 1 inlet (decant from the AFP), ASP 1 weir decant and the final effluent at the second splitter box. This measurement allowed for the estimation of the HRT in the PFP and both HRAOPs. When inflow rates deviated from the set flow rates as detailed by (Rose et al. 2007), the flow rate was adjusted accordingly.

2.6.2 Algae settling pond cleaning and weir debris removal. Algae settling ponds (ASP) are scheduled for emptying out and cleaning every fortnight to harvest microalgae bacterial biomass and avoid decomposition of settled solids, thereby maintaining effective unit performance (Cowan et al. 2016). Cleaning dates and estimation of the next cleaning date were therefore noted as part of standard operating procedures. Debris accumulation at the ASP weirs was checked occasionally and immediately removed.

2.7 Plant troubleshooting

Extensive research has previously been carried out on the IAPS; hence the resultant metadata provided the basis for ‗normal condition recognition'. Biomass structure is one such parameter. When biomass found in the HRAOP was no longer observed as aggregates, a number of troubleshooting exercises were carried out. The trouble shooting exercises carried out included linear velocity, volute measurements, species diversity, population dynamics, and MaB-flocs identification and settleability.

2.7.1 Volute measurement. During sampling, a period of no floc formation (June 2018-September 2018) was experienced. The volute was then measured as a troubleshooting tool to establish the type of flow within the ponds. The paddlewheels were stopped for this procedure. The distance from the depression, including the width of the paddlewheel, was measured and recorded. The width of the paddlewheel was also measured and recorded. The volute was recorded as the difference between these values.

2.7.2 Linear velocity of water flow. An estimation of the linear velocity was carried out by measuring the time it took for a floating plastic table tennis ball to travel around the HRAOP. A table tennis ball was thrown randomly into the pond, and the moment it landed on the water surface, a timer was started. The time it took the ball to travel from the front of the paddlewheel around the HRAOP to return to the paddle wheel was recorded. Velocity was calculated according to Equation 2.6

$$velocity\ m.s^{-1} = \frac{distance\ (m)}{time\ (s)}$$ …………Equation 2.6

2.7.3 Microscopy.

Microscopy was carried out as described in section 2.5.1. The purpose of this analysis was to observe species composition, diversity, and presence of microalgal-bacterial flocs. Continual changes in algae dominance are prone in HRAOP, and this affects plants performance. Variations have been attributed to changes in operational conditions and or environmental conditions, biological selection pressures and changes in influent water quality parameters. (Sutherland et al. 2017; Cowan et al. 2016). Knowledge of the variations in population dynamics will assist with predicting probable changes in operational conditions and effluent water quality. Species identification was achieved by referencing previous studies on the HRAOP (Johnson 2010; Jimoh 2017).

2.7.4 Settleability. MaB-flocs identified at the Belmont IAPS have been characterised as readily settleable (within two hours) distinct constituents. Settleability tests were carried out in Imhoff cones in accordance with APHA method 2540 F (APHA, 1988). An Imhoff cone was filled to the 1-L mark with a well-mixed sample and left to stand uninterrupted for 1 hour. After an hour, 50 ml supernatant liquor was pipetted out at the halfway point between the surface of the settled material and the liquid surface. TSS measurement was carried out for this liquor, and this represented non-settleable solids. Percentage settleability was calculated conforming to Equation 2.7.

$$\%\ settleable\ solids = \frac{(TSS\ mg.L^{-1} - mg\ non\text{-}settleable\ solids\ L^{-1})}{TSS\ mg.L^{-1}} \times 100$$ ……Equation 2.7

2.8 Operational and maintenance costs

The Belmont Valley plant utilises 34 126 $kWh.y^{-1}$, it was built to treat wastewater for 500-person equivalent (PE), at a continuous inflow rate of 75 kL d^{-1}. Therefore operational costs were calculated as shown in Equation 2.8.

$Operational\ costs\ (ZAR) = 34\ 126 kWh.y^{-1} \times$
$average\ cost\ of\ electricity\ per\ year$……………………Equation 2.8

To calculate Operation and maintenance costs per person equivalent (ZAR. $PE^{-1}.y^{-1}$) Equation 2.9 was used whilst Equation 2.10 was used to calculate total costs per litre treated (ZAR. $L^{-1}.y^{-1}$).

$\frac{operational + maintenace\ costs\ (ZAR)}{500\ (PE)}$……………..Equation 2.9

$\frac{operational + maintenace\ costs\ (ZAR)}{75\ 000\ (L)}$……………Equation 2.10

2.9 Statistical Analysis

All data were computed and analysed on Microsoft Excel 2010 and Sigma Plot Version 11, (Systat Software Inc., USA). A one-tailed distribution *t*-test (Microsoft Excel, 2010) at alpha level (p) 0.05 was used to determine the level of significance between the mean concentrations at different treatment units for all data sets. A correlation test was performed to determine the factors which affect faecal coliform counts.

2.10 Process audit framework

The tools used for the development of the process audit framework included; a turtle diagram, a flow diagram, and a checklist. The process audit turtle diagram was created by answering the questions supplied in the ISO 9001 template shown in Figure 2.2. The process flow diagram presented in Russell (2009) was used as a guideline to prepare the IAPS process flow diagram. The ISO 9001 management checklist template was used to develop checklist questions for IAPS.

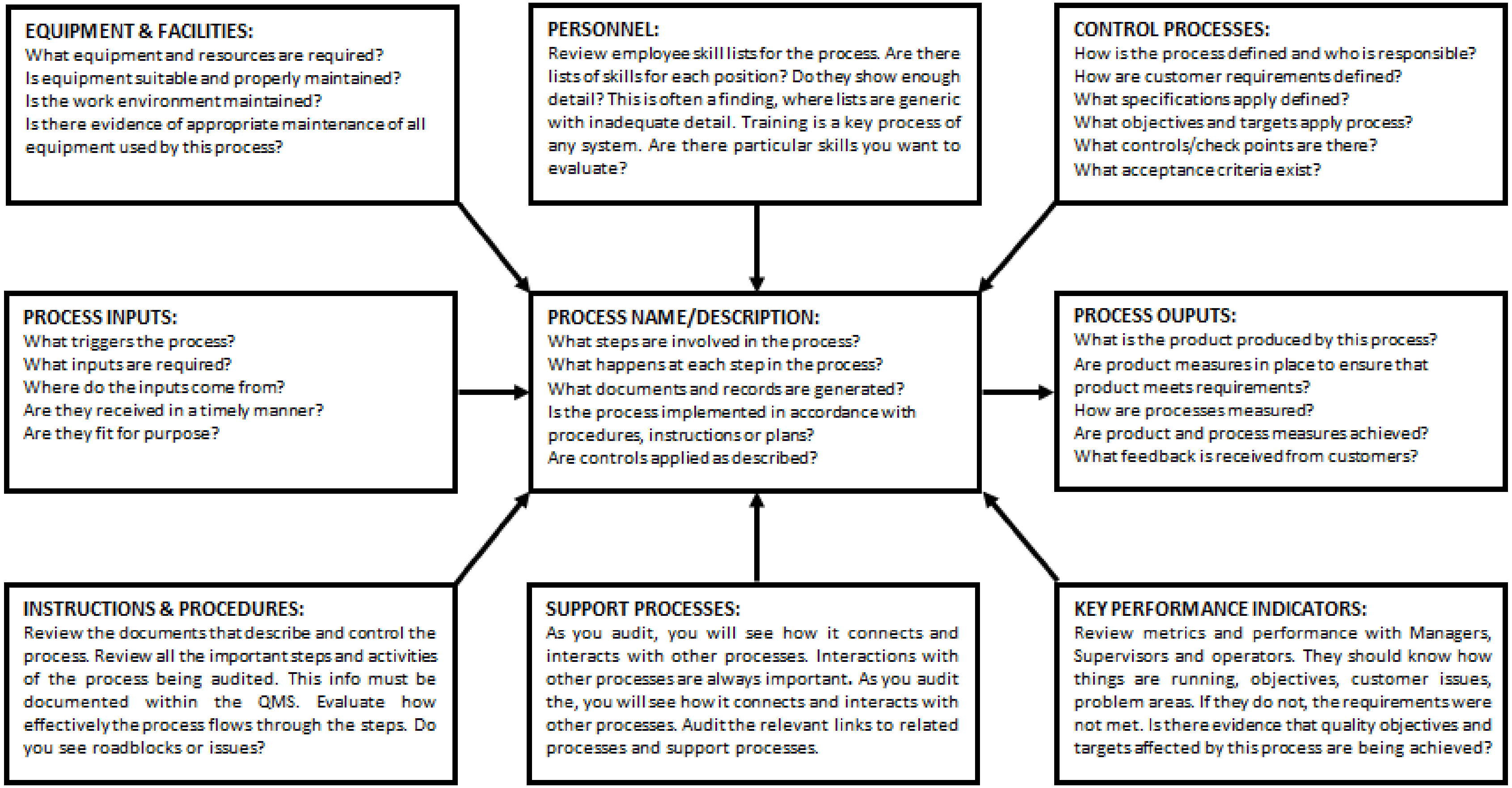


Figure 2:2: ISO 9001 management systems process audit turtle diagram template.

Parameters that were used to develop the IAPS process audit framework are shown in Table 2.2 below.

Table 2:2: Process audit parameters determined weekly for Belmont Valley IAPS during 9 month sampling period (May 2018- February 2019).

Parameter	Monitoring point	Sample description	Purpose
Flow rate (L sec^{-1})	Raw inlet in AFP AFP outlet into P1 Settlers at splitter box	Volume recovered per unit time	Plant management
Linear velocity (m sec^{-1})	HRAOP	Distance travelled per unit time	
TSS (mg. L^{-1}) COD (mg. L^{-1}) Nutrients (mg. L^{-1})	Raw effluent AFP effluent HRAOP pond contents ASP effluents	Composite	Water quality tests
pH, DO (mg. L^{-1}), EC (mS m^{-1}), temperature ($^{\circ}$C)	Onsite at all sampling points	Diurnal average (n=3)	Water quality and process control. Diurnal profiles determined
Settleability	HRAOP water	Composite	
Algal genera	HRAOP water	Grab	Process management and troubleshooting.
Volute size measurement	HRAOP	The gap between HRAOP base and paddlewheel.	Troubleshooting

AFP= Advanced facultative pond. ASP= Algae settling pond. HRAOP= High rate algae oxidation pond

Chapter 3 : Water quality and efficiency of integrated algal pond systems

3.1 Introduction

Water demand is increasing rapidly each year, and its availability is currently threatened by climate change and water pollution. Already close to 50% of the world's population is already living in a water scarce region; hence there is a need to safeguard and restore water quality (WWAP 2018). Proper wastewater treatment practices provide avenues for ensuring that water supplies and rivers are protected from pollution.

Integrated algal pond systems (IAPS) have been examined and evaluated for more than 70 years and are considered as an effective, economic, simple and sustainable treatment option for the reclamation of wastewater (Oswald and Asce 1990). This technology, based on the symbiotic relationship between algae and bacteria and powered by solar energy, is efficient and capable of removing chemical and biological oxygen demand and reducing nutrient load and coliform bacteria.

As discussed in Chapter 1, wastewater treatment plant should treat influent to a level that is acceptable for discharge and or irrigation. In South Africa, the wastewater limit values for the discharge of wastewater into a water resource and irrigation are shown in Table 2.2 and 2.3. The first system evaluation on the Belmont Valley IAPS revealed that when the HRAOP were configured in series, the effluent produced met standards of discharge and irrigation as set out in the Water Act (1998) with regards to nutrient levels. COD and faecal coliform counts were non-compliant. Plant performance reductions reported for the Belmont Valley demonstration scale IAPS were 87% COD, 76% phosphate, 55% nitrogen and 99.999% *E.coli* (Wells et al. 2005; Rose et al. 2007). Subsequent research (Mambo et al. 2014; Jimoh 2017) focused on the incorporation of tertiary treatment units to improve water quality. Results of these studies showed that indeed in order for IAPS to comply with regulatory standards of discharge and irrigation of land of up to 2000 cubic metres, a tertiary treatment unit must be installed.

Algae research has significantly advanced over the years, and the interest is microalgae biomass due to its vast application to day-to-day life and its ability to be produced on unproductive land at a faster growth rate than crops (Campbell et al. 2011). Professor Oswald (1995) hypothesized a pond revolution in the 21st century, which would not only serve as wastewater treatment vessels but feedstock. With the advancement in technology, it has been shown that the potential of biomass includes energy, therapeutics, pharmaceuticals, and cosmetics production (ref). IAPS provides a low-cost option in the production of biomass as it is produced as a by-product of wastewater treatment. The most recent studies on the Belmont Valley HRAOP 2 have characterised the quantity and quality of biomass produced (Johnson 2010; Jimoh 2017). Results indicate that the biomass produced aggregates into MaB-flocs is composed of microalgae and bacteria, and settles rapidly. Protein, carbohydrate and lipid concentrations were 36%, 19% and 16% respectively. Biomass productivity was shown to undergo seasonal and diurnal variations, and values ranged from 58 kg ha^{-1} d^{-1} in spring to 215 kg. ha^{-1} d^{-1} in summer. This chapter provides a re-evaluation of the performance efficiency and effectiveness of the Belmont Valley IAPS in treating domestic effluent. Biomass productivity and composition in the HRAOP 2 is assessed, and the ability of the ASP to recover biomass is also investigated. The effect of plant management on plant performance is investigated; control processes and key performance indicators within the system are also identified

3.2 Results

3.2.1 Treatment efficiency of IAPS (May 2018-February 2019)

Both DO and pH showed a continual increase throughout the system, as shown in Figure 3.1. DO increased from 2.5 ± 0.3 mg. L^{-1} in the raw influent to 6.0 ± 0.4 mg. L^{-1} in the final effluent. AFP effluent was anoxic (1.8 ± 0.08 mg. L^{-1}) throughout the sampling period possibly due to the duckweed which covered the pond throughout the sampling period. It was also noted that each HRAOP contained water with a higher DO concentration than that of the ASP it deposited water into. pH showed a steady increase throughout the system from 7.56 ± 0.4 to 9.9 ± 0.1 in the final effluent. Values for EC were generally within discharge limits throughout the system; however, a decrease from 122.6 ± 6 to 98.7 ± 2 mS. m^{-1} was still achieved. The lowest EC values were recorded when the plant was in recycle mode with values less than 100 mS. m^{-1} being recorded from inlet to outlet.

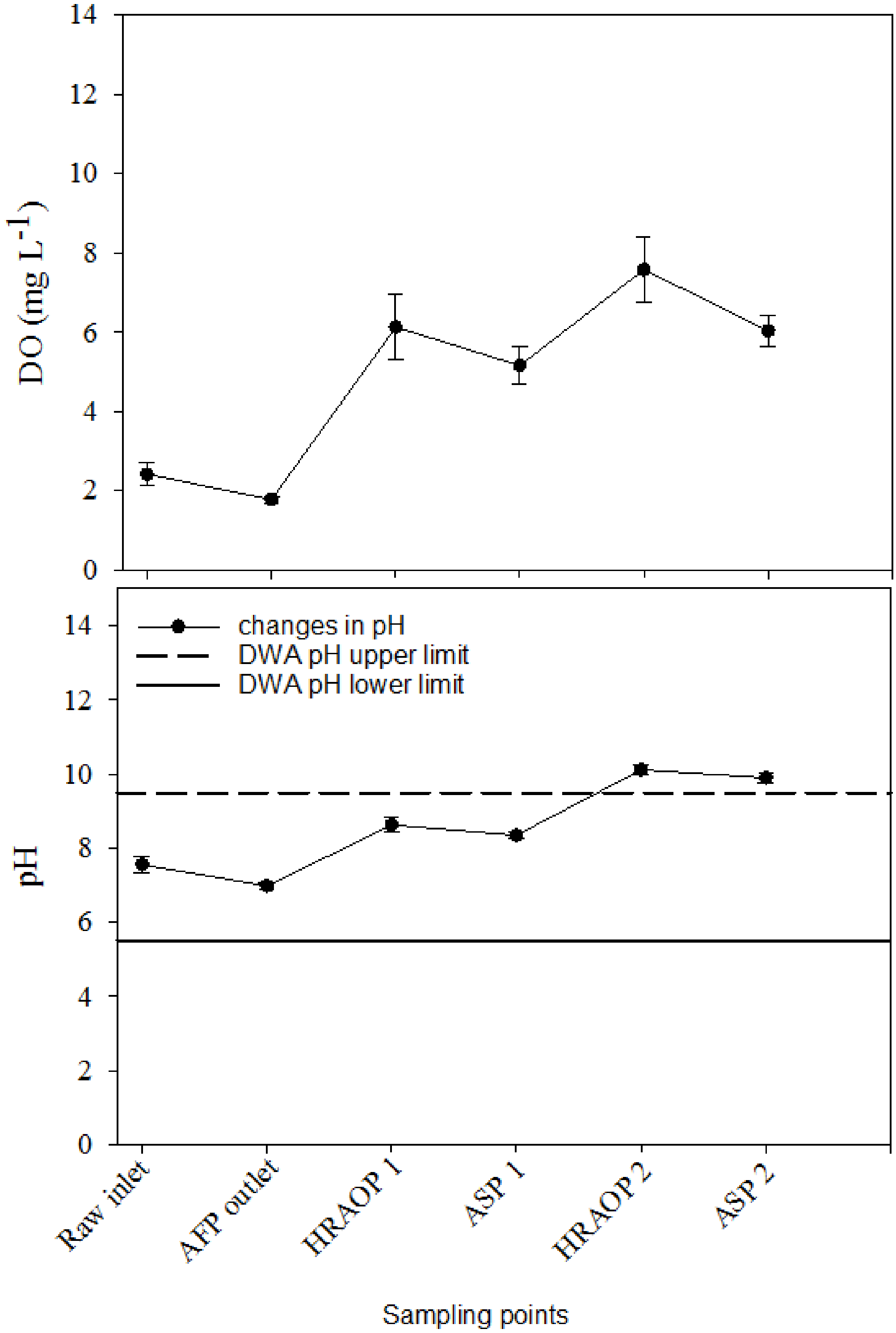


Figure 3:1: Treatment efficiency and changes within the IAPS treatment units from inlet to final effluent. Number of sampling times is 26. Bars indicate standard error.

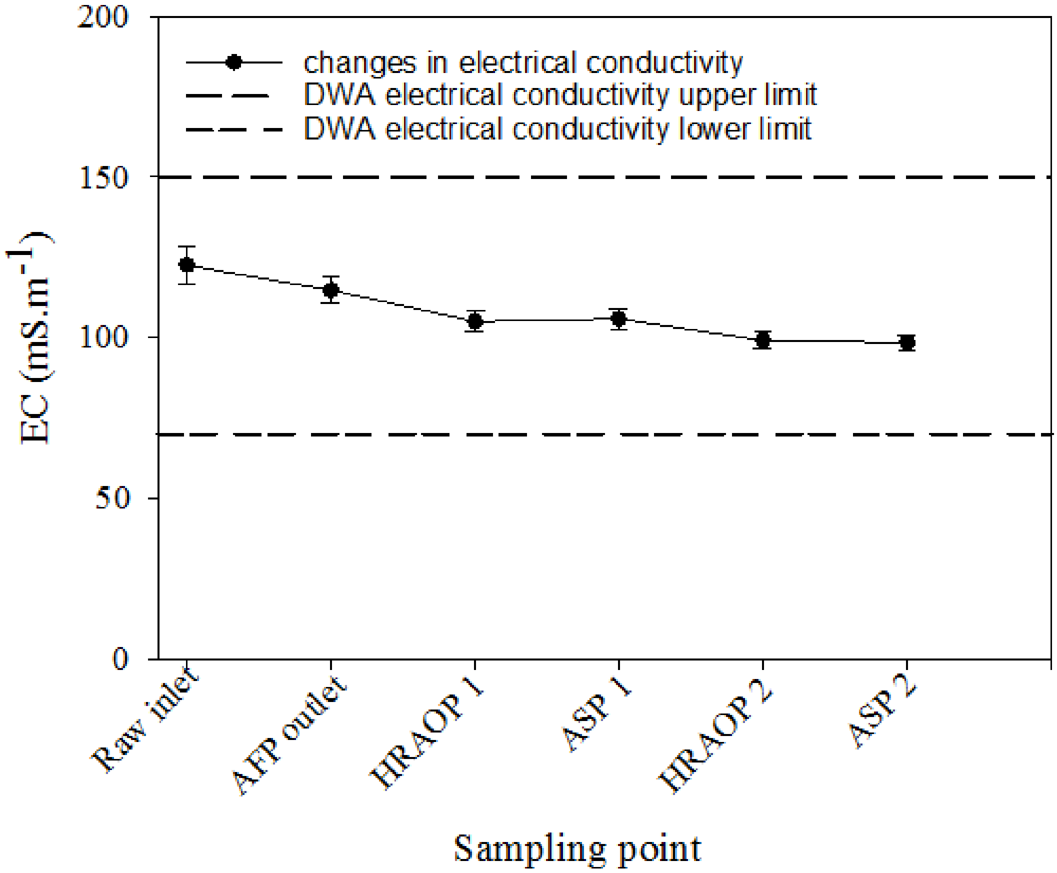


Figure 3:2: Treatment efficiency and changes within the IAPS treatment units from inlet to final effluent. Number of samples 26. Bars indicate standard error.

The average COD and TSS reduction throughout the sampling period were 89.8 ± 1% and 85.0 ± 3.9% respectively. Increases in concentration were noted in both parameters in HRAOP 1 and 2, presumably due to an increase in microalgae bacterial biomass within the ponds. The bulk removal with regards to both parameters was achieved in the AFP with efficiencies greater than 70%. A slight reduction in TSS removal efficiency of 3% was observed from the AFP effluent to the final effluent however statistical analysis (t-test, $p > 0.05$) confirmed that the increase was insignificant ($p = 0.155$). Algae Settling Ponds (ASP) 1 and 2 effectively removed TSS by 54.7 ± 9.3 % and 84.3 ± 3.7% respectively when MaB-floc were present and 29.7 ± 7.9% and 36.7 ± 7.8% respectively when MaB-flocs were absent. The average TSS for the final effluent during MaB floc absence and presence were 81.6 ± 8.7 mg. L^{-1} and 25.4 ± 4.5 mg. L^{-1}. The average inlet COD during the sampling period was 721.1 ± 141.4 mg. L^{-1} with minimum and maximum values of 125 ± 2 mg. L^{-1} and 3625 ± 5 mg. L^{-1}.

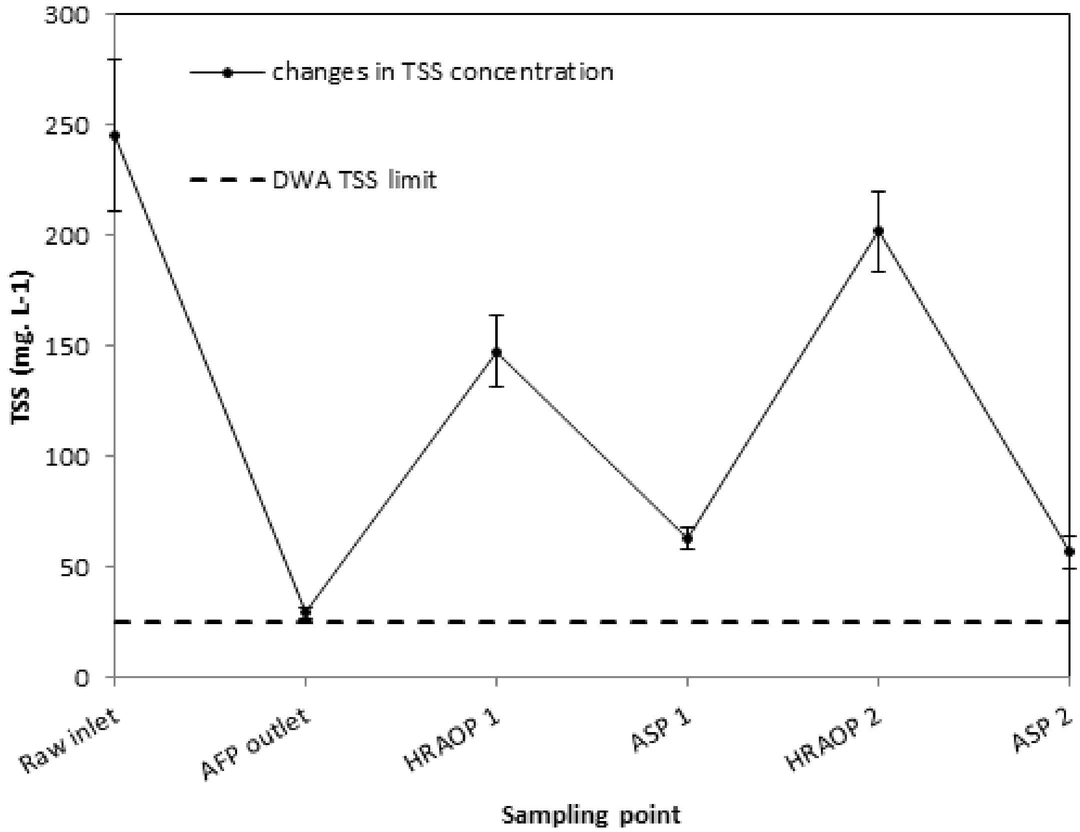


Figure 3:3: Treatment efficiency and changes within the IAPS treatment units from inlet to final effluent for TSS. Number of samples 28. Bars indicate standard error.

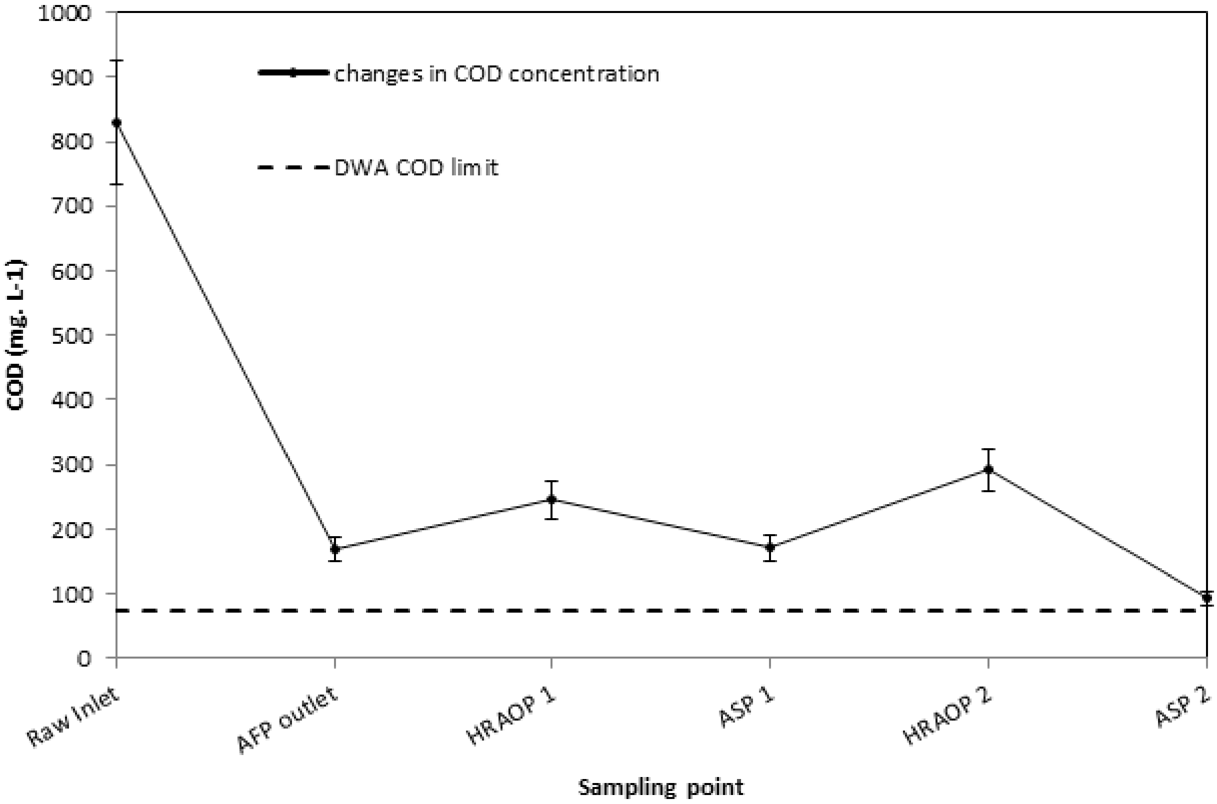


Figure 3:4: Treatment efficiency and changes within the IAPS treatment units from inlet to final effluent for COD. Number of samples 28. Bars indicate standard error.

99.9% faecal coliform reduction was achieved during the sampling period; however, the average faecal coliform count was non-compliant to discharge limits. High values above 1000 cfu.100mL^{-1} for final effluent were recorded during the autumn and winter months. Zero faecal coliform counts were achieved for 59% of the samples analysed during the 9-month monitoring period. Towards the end of winter from the 29th of August, faecal coliform counts of less than 1000 were consistently achieved.

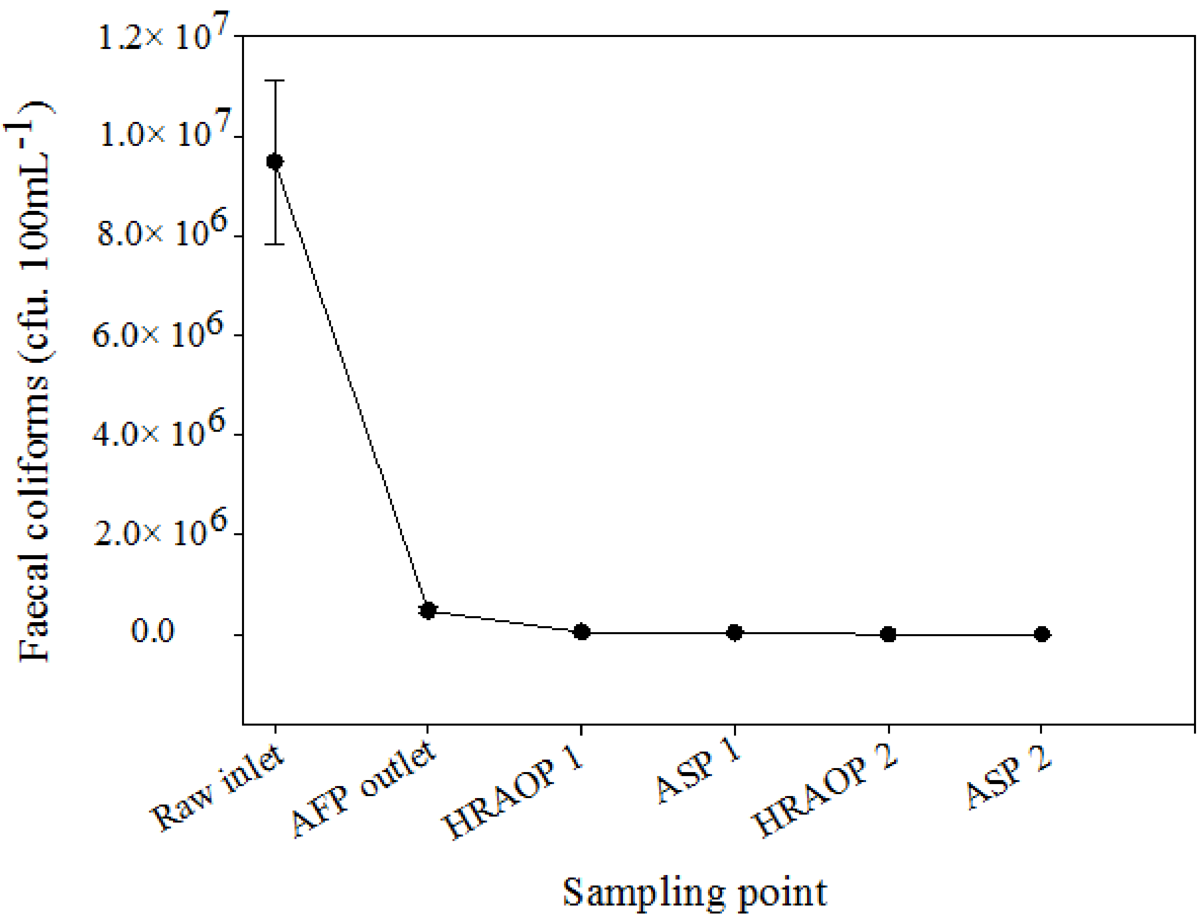


Figure 3:5: Treatment efficiency and changes in faecal coliforms within the IAPS treatment units from inlet to final effluent. Number of samples 24. Bars indicate standard error.

High nutrient removal rates were achieved within the system with regards to ortho-phosphate and ammonium-N whilst nitrate concentration increased. Nutrient concentrations for the final effluent complied with discharge limits (Figure 3.6). Ammonium-N and ortho-phosphate concentrations were reduced by 93.0% and 81.6% respectively. Influent ammonium-N concentrations as high as 30 mg. L^{-1} was recorded in the raw effluent; however final ammonium concentration was never above 4 mg. L^{-1}. The greatest reductions were experienced in HRAOP 2, were values dropped from 8.2 ± 1.1 mg. L^{-1} to 1.2 ± 0.2 mg. L^{-1} for ortho-phosphate and 5.8 ± 0.6 to 1.4 ± 0.2 mg. L^{-1}. Increases in ammonium-N and ortho-phosphate concentration experienced for the final effluent was insignificant ($p = 0.06$ and 0.4 respectively). Nitrate concentration increased by 81% and recorded final effluent concentrations of 3.3 ± 0.64 mg. L^{-1} from the initial 0.7 ± 0.2 mg. L^{-1} in the raw influent. The highest nitrate concentration of 3.8 ± 0.8 mg. L^{-1} was observed in HRAOP1. Overall nitrate concentrations were lower compared to those of previous research (Table 1.2), but this may be attributed to the recycling than occurred for more than half the year.

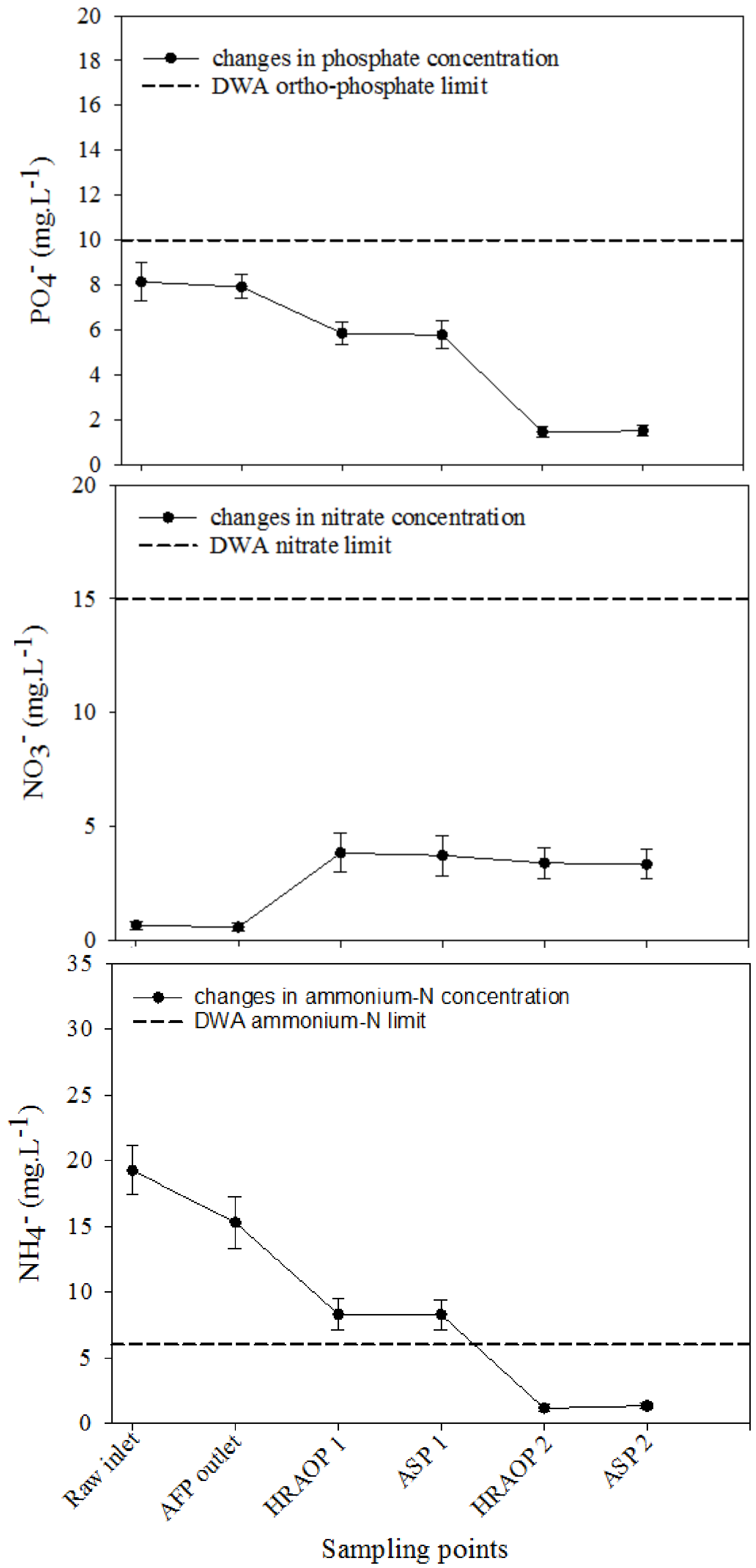


Figure 3:6: Treatment efficiency and changes in nutrient concentrations within the IAPS treatment units from inlet to final effluent. Number of samples 25. Bars indicate standard error.

The graphs in Figure 3.1- 3.6 detail the overall performance efficiency of IAPS over a period of 9 months. Results showed that IAPS was very efficient in reducing all standard water parameters will the exception of pH, DO, and nitrate, which increased in concentration in comparison with the raw effluent as expected.

3.2.2 IAPS effluent water quality

A summary of the physical, chemical and microbial composition of the IAPS final effluent analysed from May 2018 - February 2019 is presented in Table 3.1. Dissolved oxygen (DO) and electrical conductivity (EC) were within the limits of discharge, with an average of 6.0 ± 0.4 mg. L^{-1} and 98.7 ± 2 mS. m^{-1}. Average pH was slightly above discharge limits with values ranging from 8.6 to 11.6 throughout the sampling period, and the lowest pH values were recorded between June and August 2018.

Nutrient composition of final effluent was well within general limits with values of 1.6 ± 0.2 mg. L^{-1} for ortho-phosphate, 1.7 ± 0.3 mg. L^{-1} for ammonium-N and 3.3 ± 0.6 mg. L^{-1} for nitrate.

Expectedly total suspended solids (TSS), chemical oxygen demand (COD) and faecal coliforms did not regularly meet the standards of discharge. A lot of variation was observed in the TSS concentration, which recorded a very high average of 55 ± 7.1 mg. L^{-1}. The highest TSS concentration of 128 mg. L^{-1} was attained during the period of no MaB floc formation, yet the lowest value of 5 mg. L^{-1} was achieved during good floc formation and *Daphnia* presence. COD concentration was routinely higher than the 75 mg. L^{-1} discharge limit with an average of 94.1 ± 10.6 mg. L^{-1} and minimum and maximum values of 30 ± 0 mg. L^{-1} and 260 ± 5 mg. L^{-1} respectively.

Faecal coliforms were slightly above the general limit of 1000 cfu.100 mL^{-1} with an average of 1482.0 ± 636.0 cfu.100 mL^{-1}. Surprisingly the autumn and winter months recorded values greater than 1000 cfu. 100 mL^{-1} despite the fact that the system was in recycle mode. From the 29th of August, 2019, complete faecal coliforms removal was routinely achieved (Figure A6 in appendix). Some variations in water quality during the sampling period were due to system operation and maintenance and varying weather conditions.

Table 3:1: IAPS effluent water quality over a period of 9 months. Data are presented as the mean ± SE of samples collected between May 2018 and February 2019.

Parameter	Concentration
pH	**9.9 ± 0.1**, (n = 26)
Dissolved oxygen (mg. L^{-1})	6.0 ± 0.4, (n = 15)
Electrical conductivity (mS. m^{-1})	98.7 ± 2, (n = 26)
COD (mg. L^{-1})	**94.1 ± 10.6**, (n = 28)
TSS (mg. L^{-1})	**55 ± 7.1**, (n = 28)
Nitrate/nitrite-N (mg. L^{-1})	3.3 ± 0.6, (n = 25)
Ammonium-N (mg. L^{-1})	1.7 ± 0.3, (n = 25)
Ortho-phosphate (mg. L^{-1})	1.6 ± 0.2, (n = 25)
Faecal coliforms (CFU. 100 mL^{-1})	**1482.6 ± 636.0**, (n = 24)

Numbers in the parenthesis represent the number of data used to generate the means. Values highlighted in bold do not comply with the General limits for discharge.

3.2.3 Biomass composition and productivity.

During the course of the research, two phases were observed in the structure of the biomass, i.e. a period of MaB floc presence and absence, as shown in Figure 3.7 below. The period of MaB floc presence coincided with a richer microbial diversity whilst during MaB floc absence only the microalgae *Scenedesmus* and *Chlorella* were observed. Table 3.2 shows the microorganisms that were observed microscopically during the course of this work. *Closterium*, *Chlamydomonas*, *Pyrobotrys*, *Euglena*, *Actinastrum*, and *Dictyosphaerium*, have been cited in previous research on the Belmont demonstration scale however, they were not observed during the course of this study (Johnson 2010; Jimoh 2017). The bacterial component was taken from research that was carried out concurrently with this work (Masudi 2019). The range of microbes observed and isolated included microalgae, bacteria, metazoa and protozoa.

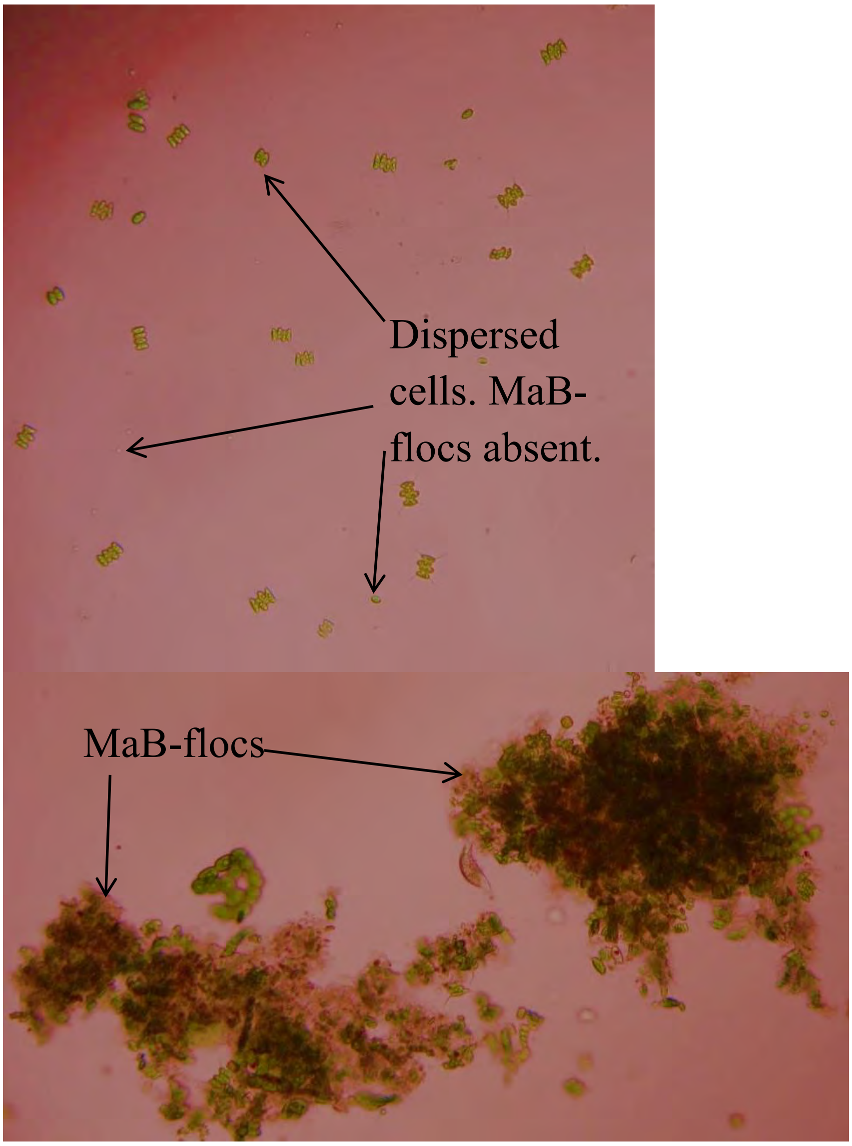


Figure 3:7: Low resolution (×100) light microscope analysis of high rate algal oxidation pond 2 water. Light microscope images during (a) MaB floc absence (June-September 2018), (b) MaB floc presence (October 2018-February 2019).

Table 3:2: Microbial composition of biomass 2010-2019 based on microscopic observations and genomic DNA extraction and pyrosequencing.

Bacteria	Algae	Protozoa	Metazoa
Planococcus maitriens, Bacillus, Fictibacillus, Aeromonas, Exiguobacterium, Arthrobacter, Enterobacter, Microbacterium, Pseudomonas, Ancylobacter	*Pediastrum, Scenedesmus, Micractinium, Diatoms, Chlorella, Closterium, Chlamydomonas, Pyrobotrys, Euglena, Actinastrum, Dictyosphaerium, Ankistrodesmus sp., Navicula sp, Nitzchia sp*	*Vorticella, Aspidisca*	*Daphnia, Brachionus, Conochilus, Lecane*

The biomass productivity recorded during the course of the study within the HRAOP 2 was 130.6 ± 11.2 kg. $ha^{-1}d^{-1}$ however, it is estimated that the actual settled biomass recovered within the ASP was only 95.1 ± 15.1 kg. $ha^{-1}d^{-1}$.

Table 3:3: Biomass productivity and losses observed in high rate algal ponds. Number of samples 28.

	Productivity kg. $ha^{-1}d^{-1}$
HRAOP 2	130.6 ± 11.2
Settled in ASP 2	95.1 ± 15.1
Unsettled biomass (lost to final effluent)	35.5 ± 5.7

The results in this section are an average of three research outcomes carried out on the Belmont Valley HRAOP 2 from as early as 1998 (Potts 1998; Johnson 2010; Sibelo 2019) There is a lot of variation which exists within the biochemical composition of the Belmont

Valley biomass from 1998- 2019. The range of values for protein, lipid and carbohydrate concentration is 10.7 - 46%, 4.8-19 % and 16 – 35.1 % respectively.

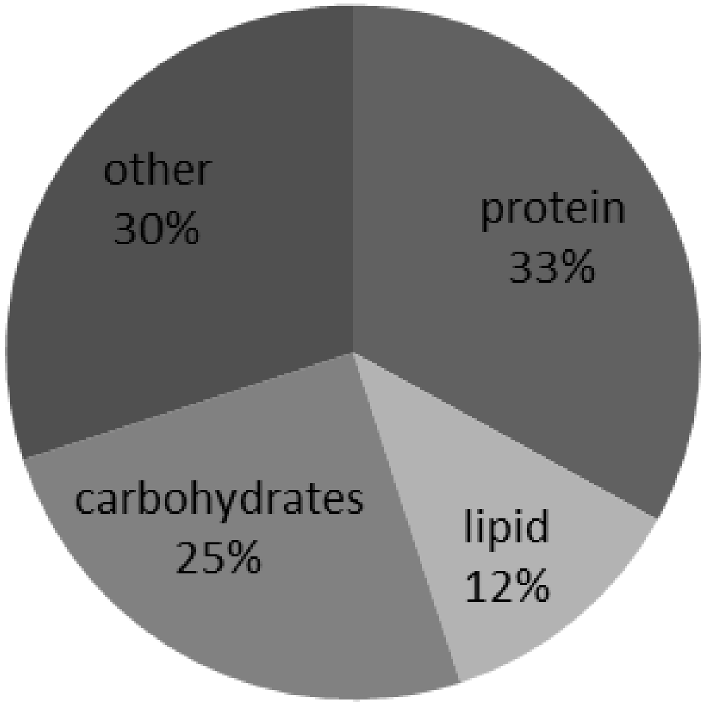


Figure 3:8: Average biochemical composition of produced biomass. Results are an average of three research outcomes carried out on the Belmont Valley biomass from as early as 1998 (Potts 1998; Johnson 2010; Sibelo 2019)

3.2.4 Plant management, process control and key performance indicators

Biomass removal after settling.

The algae settling ponds (ASP) for the Belmont Valley IAPS are scheduled for cleaning every fortnight. During sampling, fluctuations were observed in final effluent TSS data, as shown in Figure 3.9, with values ranging from 5 ± 0 mg. L^{-1} to 142 ± 2.8 mg. L^{-1}. However, a closer look revealed that every time the ASP was emptied out and cleaned TSS levels would reduce, then rise steadily until the next cleaning date after which a decrease was experienced again after cleaning. This was true irrespective of MaB floc absence or presence. A series of exercises were then carried out to closely monitor the changes in TSS concentration during different cleaning cycles.

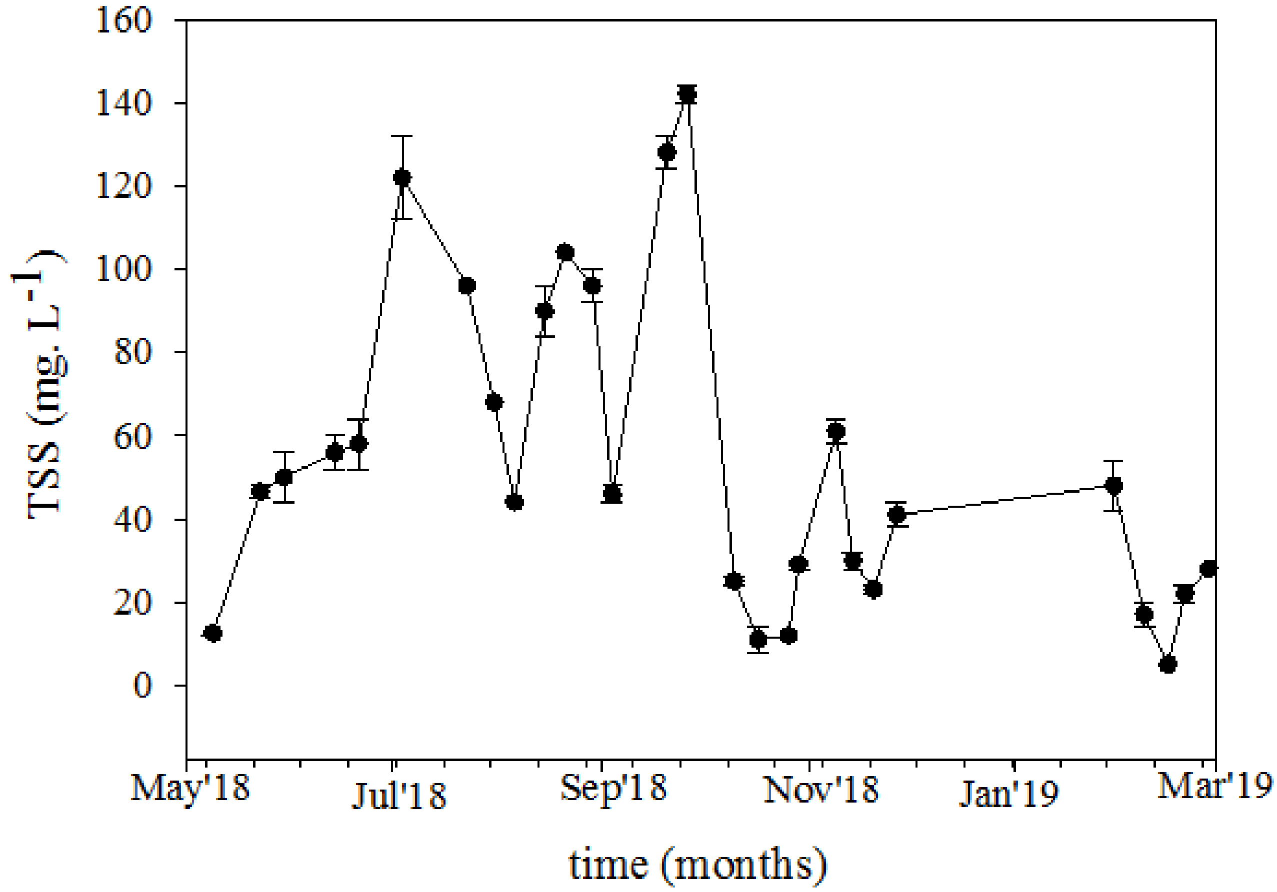


Figure 3:9: Fluctuations in total suspended solids during sampling. Data are presented as the average of duplicate measurements. Bars indicate standard error.

The results presented in Figure 3.10 – 3.13 show the change in TSS between cleaning periods. During floc absence Figure 3.10, TSS would increase from 80 ± 8 mg. L^{-1} to over 100 mg. L^{-1} in the first week to 142 ± 2 mg. L^{-1} after two weeks of pond cleaning.

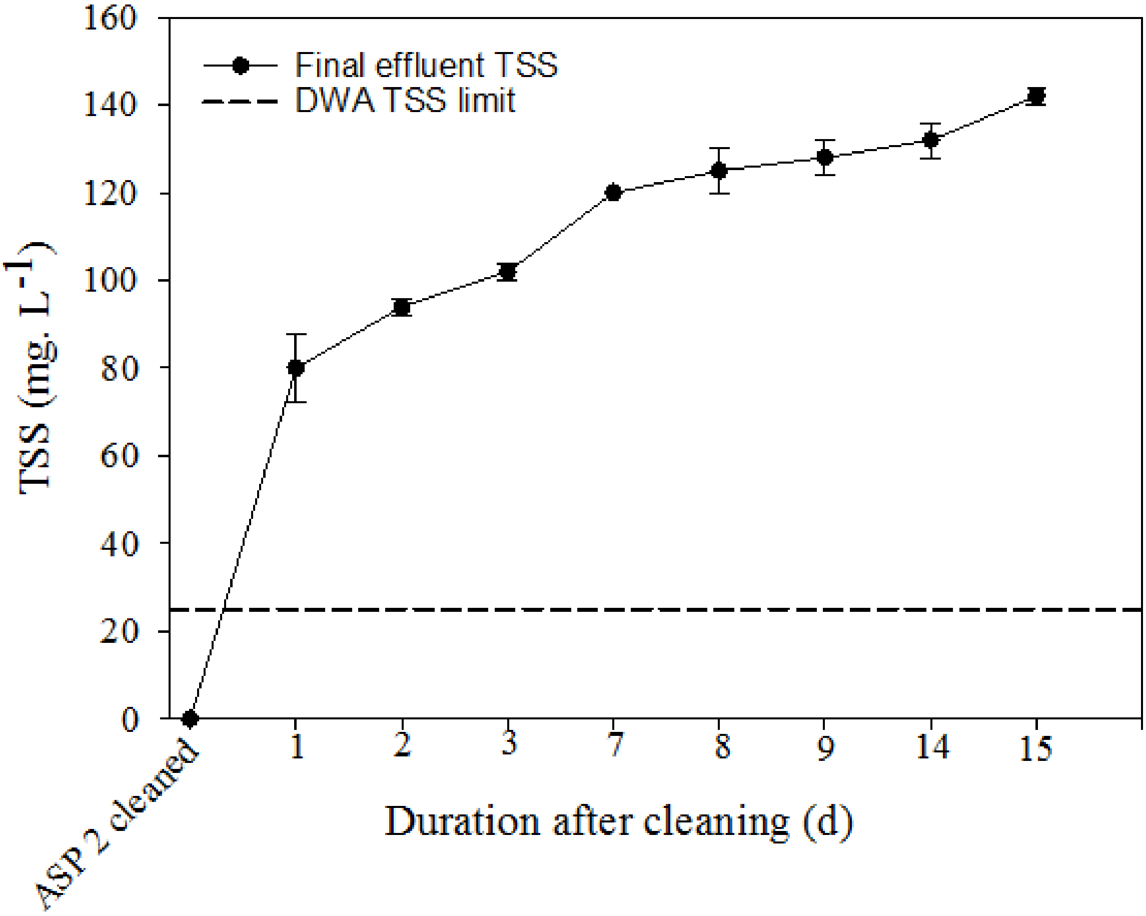


Figure 3:10: Effects of algae settling pond cleaning on integrated algae ponds final water quality. Changes in TSS during MaB floc absence. Data are presented as the average of duplicate measurements. Bars indicate standard error.

TSS also showed a steady increase during MaB floc formation as shown in Figure 3.11 and 3.12. TSS concentration would remain within the discharge limit during the first week; however, during the second-week, values above discharge limits were recorded. Of interest was the decrease in TSS levels during the second week after cleaning after a steady increase had been noted (Figure 3.12). This corresponded with the high population of Daphnia inside the settler. TSS concentration reduced from 27 ± 2 mg. L^{-1} on the 14th of October 2018 11 ± 3 mg. L^{-1} on the 17th of October 2018. During this period, no rainfall was experienced; hence dilution of water was not a contributing factor.

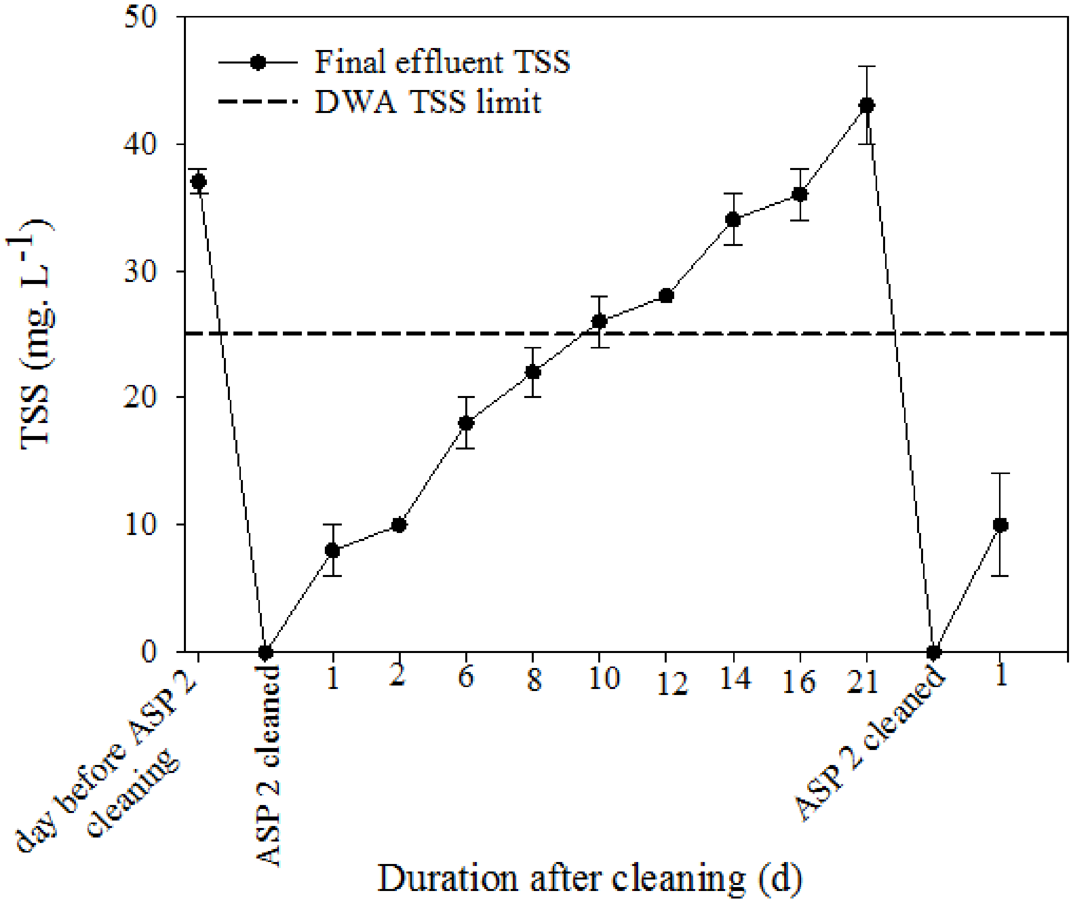


Figure 3:11: Effects of algae settling pond cleaning on integrated algae ponds final water quality. Changes in TSS concentration during MaB floc presence. Data are presented as the average of duplicate measurements. Bars indicate standard error.

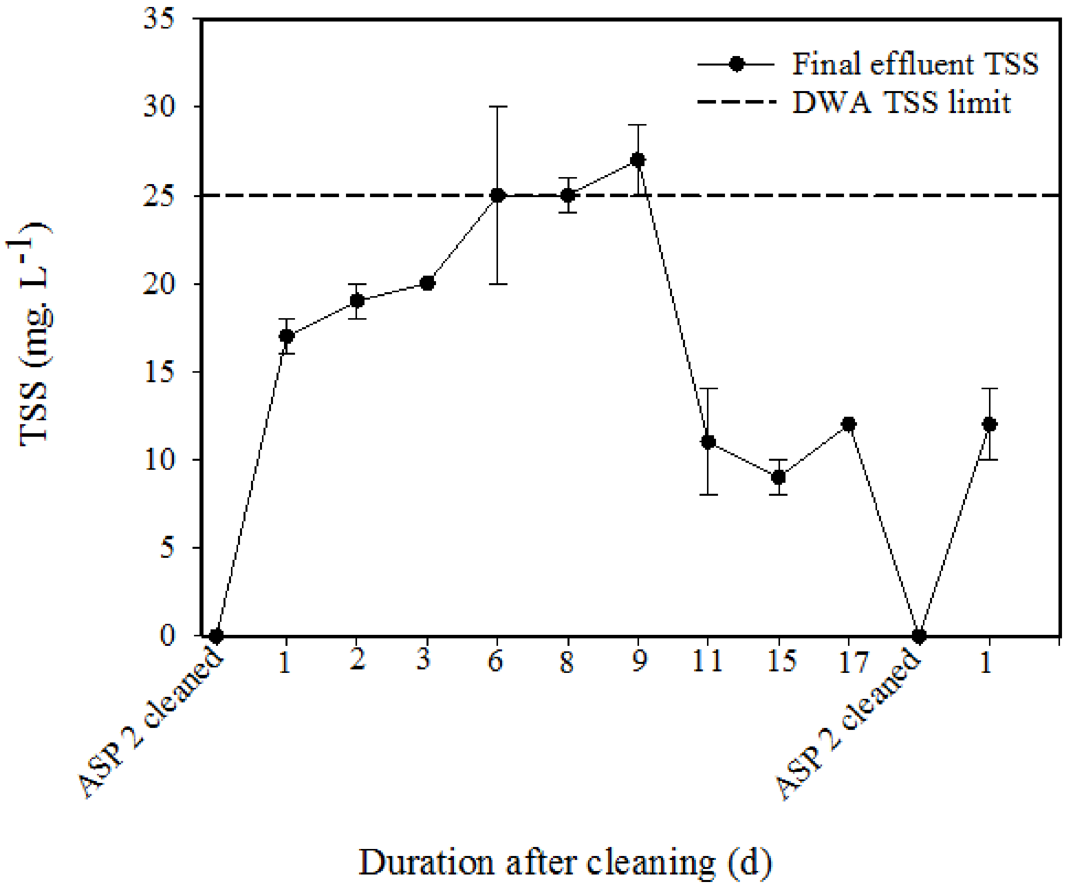
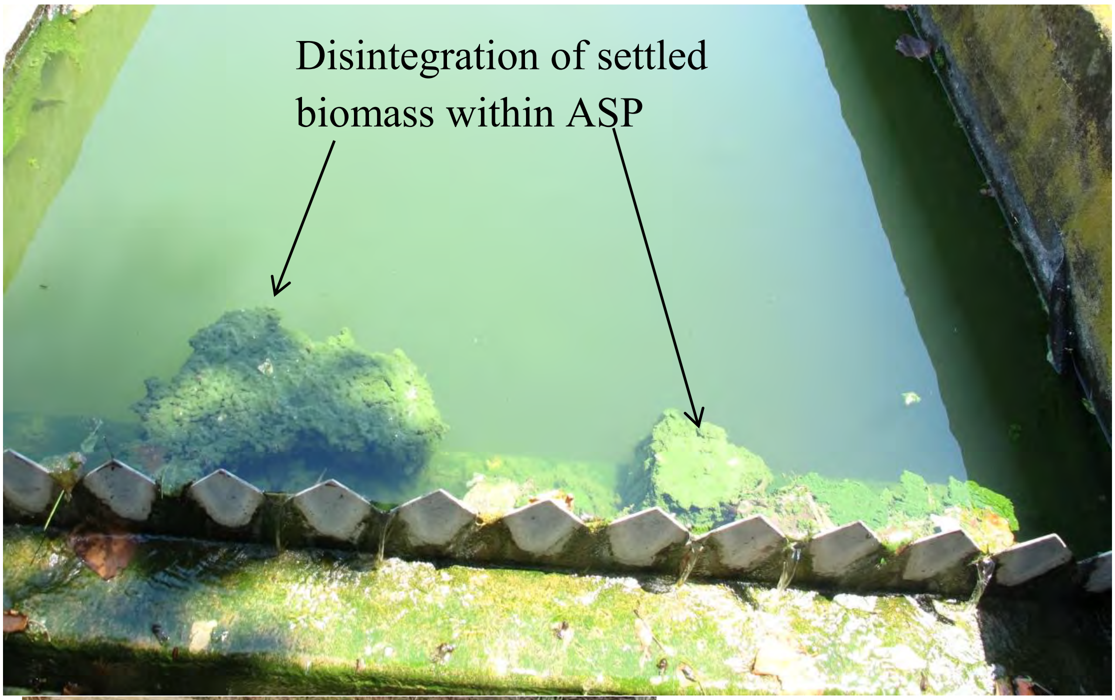


Figure 3:12: Effects of algae settling pond cleaning on integrated algae ponds final water quality. Changes in TSS concentration during MaB floc and daphnia presence. Data are presented as the average of duplicate measurements. Bars indicate standard error.

When the settler was not cleaned out for more than three weeks, TSS concentration exceeded 40 mg. L^{-1} and the settled microalgae bacterial biomass would disintegrate and float above the surface, as shown in Figure 3.13. This was unexpected as previous reports showed that biomass remains concentrated at the bottom of the settler for months.



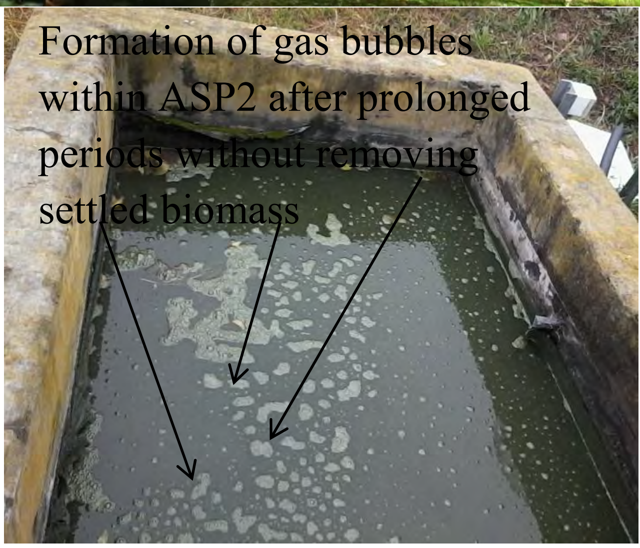


Figure 3:13: Effects of algae settling pond cleaning on integrated algae ponds final water quality. The disintegration of settled biomass within settler was noted after three weeks without cleaning.

The colour changes of the final effluent taken at 7-day intervals were also visible, as shown in Figure 3.14. This corresponded with the increase in TSS in the final effluent described above.

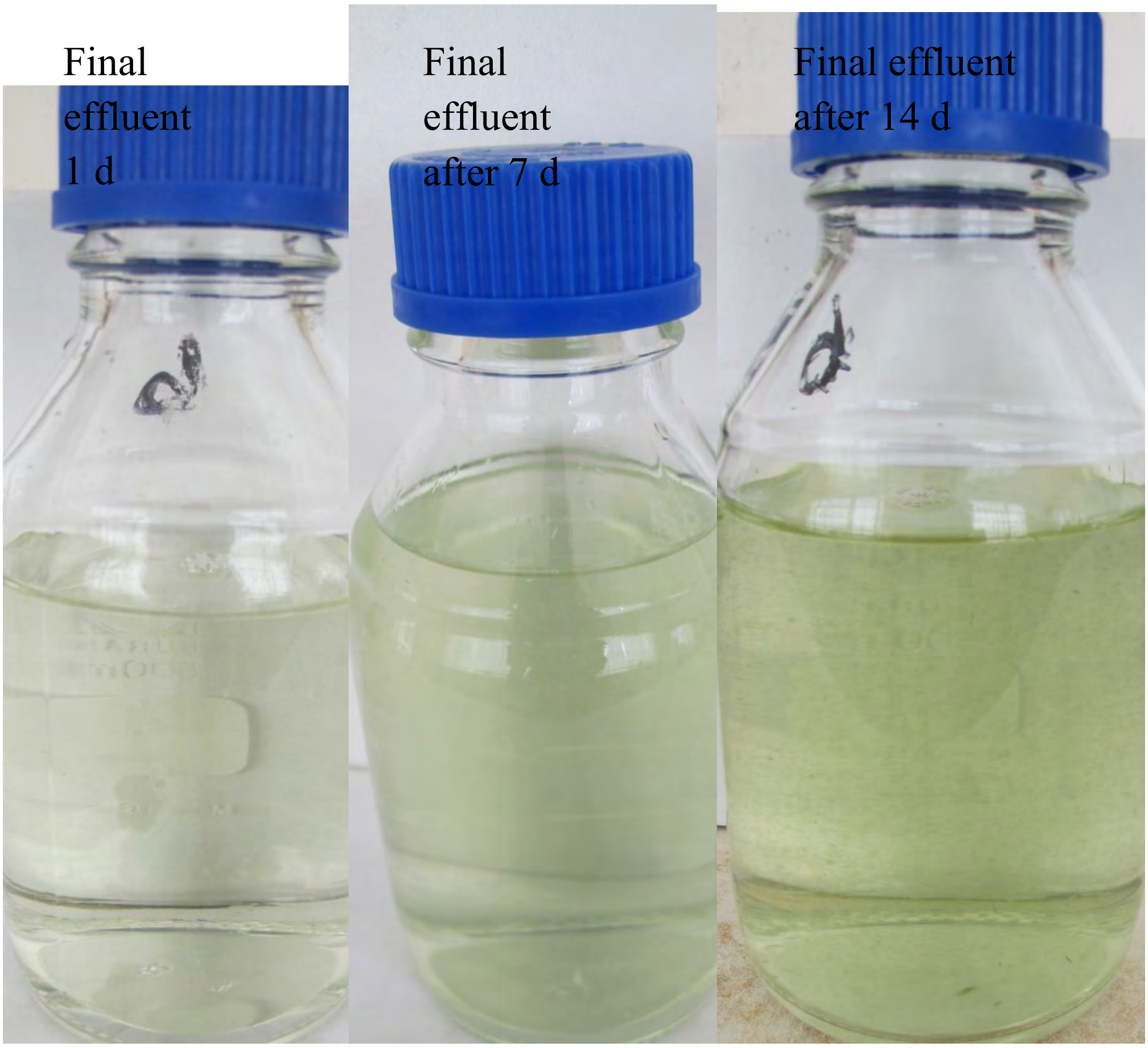


Figure 3:14: Effects of algae settling pond cleaning on integrated algae ponds final water quality. Changes in final effluent colour intensity during cleaning intervals.

During the period of May 2018 to June 2018, final TSS values began to rise excessively as shown earlier in Figure 3.15, and the final effluent had a green colour (Figure 3.15 a). Microscopic analysis revealed that the microalgal bacterial biomass was no longer aggregated as MaB-flocs and species diversity with the ponds had reduced as only *Scenedesmus* and *Chlorella* could be seen as shown in Figure 3.7 (a). In an attempt to troubleshoot, a number of exercises were undertaken to determine the cause.

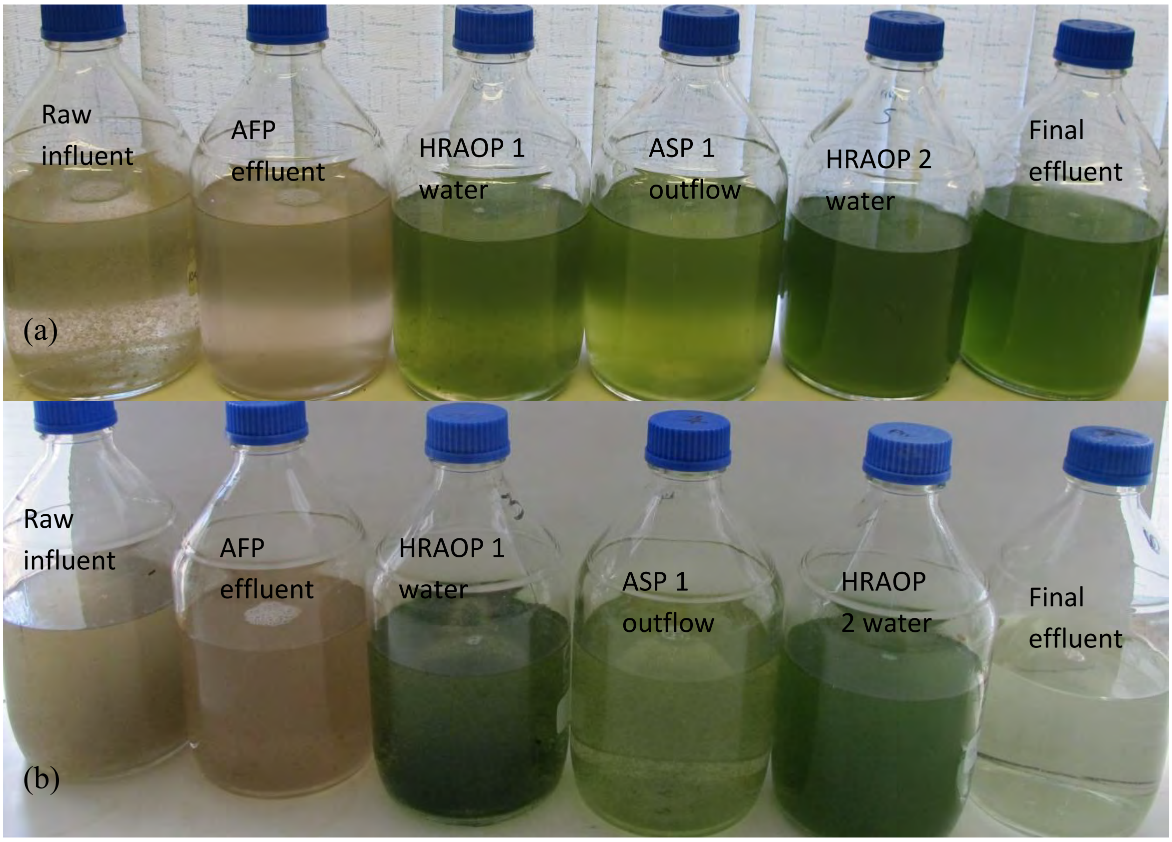


Figure 3:15: Efficacy of IAPS treatment on water clarity. Photographs show changes in the colour of IAPS treatment units water contents throughout the system during (a) MaB floc absence and (b) MaB floc presence. Left to right: raw influent, AFP effluent, HRAOP 1 water, settler 1 outflow, HRAOP 2 water, final effluent.

Firstly, recycling was stopped on the 17^{th} of July and the system started receiving raw effluent continuously in an attempt to increase nutrient concentrations within the HRAOP, which had dropped way below the usually recorded values. A month later, nutrient concentration increased; however, no aggregation of biomass was being observed. This prompted the measurement of the volute and linear velocity.

Volute measurements were taken for both ponds to determine the type of flow and paddlewheel efficiency. HRAOP 1 and 2 had volute values of 52 ± 0.06 mm and 50 ± 0.03 mm, which was considered to be within range as the paddle wheel should only be raised by a few millimetres. This exercise showed that the paddlewheel was indeed efficient, and a linear flow was still being achieved. Results for velocity measurements are shown in Table 4.1, and both ponds recorded velocities higher than the previously detailed 200-300 mm s^{-1}. This was not adjusted during the sampling period; however; this meant that more electricity was being consumed. Unfortunately, none of the troubleshooting mechanisms used aided in determining why bio-flocculation was not occurring within the ponds.

Table 3:4: Velocity of IAPS paddlewheels during the period of MaB-floc absence. Data are presented as the mean ± SE of two independent days. 15 replicates were carried out for each determination on each day.

IAPS treatment unit	Velocity (mm s^{-1})
HRAOP 1	331 ± 0.01
HRAOP 2	391 ± 0.007

MaB-floc aggregations were finally observed in October 2018 after a pump failure which resulted in the system being shut down for two days. The presence of flocs corresponded with an increase in microbial diversity (Figure 3.7 (b)). Higher microalgal species richness, metazoa (rotifers and *daphnia*) and protozoa (ciliates) were observed until the end of the sampling period in February 2019. All this led to improved effluent clarity (Figure 3.15 (b)), high settleability rates and lower TSS.

Effect of MaB floc absence and presence on settleability.

Changes in microalgal bacterial biomass settleability throughout the sampling period are shown in Figure 3.16. Indications were that the absence of flocs resulted in poor settleability during the months June-August. Percentage settleability during this period ranged from 20 - 27%. Unexpectedly good settleability (64 ± 8%) was observed in September despite the absence of flocs. During this month, a very low inflow (average 19 ± 4 kL d^{-1}) was experienced due to the installation of an old GF mono-industrial pump. TSS was also very high within the pond, and some Pediastrum was observed within the HRAOP although in small quantities. Settleability efficiencies as high as 91% were observed during periods of MaB floc presence and TSS were reduced from an average of 235 ± 29 mg. L^{-1} to 28 ± 2 mg. L^{-1}.

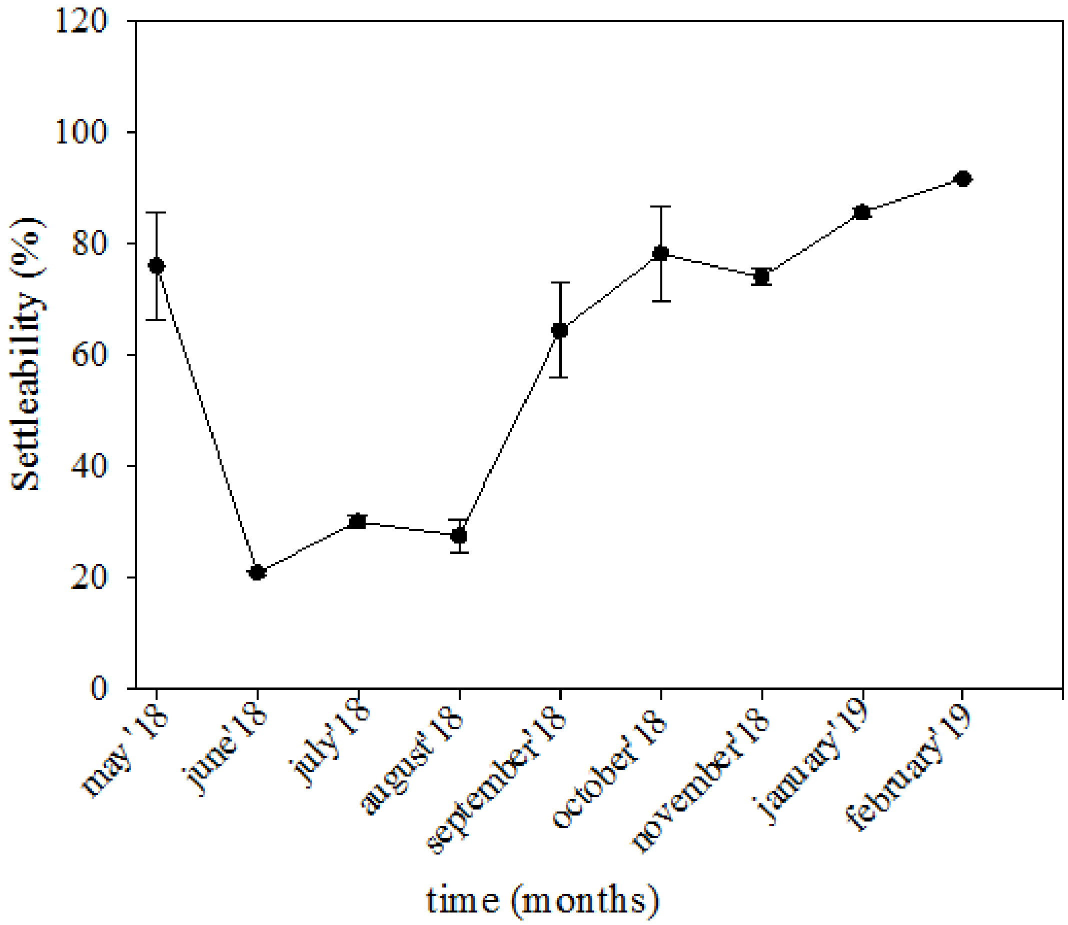


Figure 3:16: Settled biomass % of microalgal bacterial biomass in high rate algae oxidation ponds of an integrated algae pond system treating municipal sewage. Changes in percentage settleability for 1hr Imhoff settleability experiments were carried out. Data presented are the monthly average. TSS measurements were carried out in duplicates.

A noticeable difference in water clarity during settleability was also observed between the period of MaB floc presence and absence (Figure 3.17).

Figure 3:17: Settleability of microalgal bacterial biomass in high rate algae oxidation ponds of an integrated algae pond system treating municipal sewage. 1hr settling time Imhoff cone image during MaB-floc presence and MaB-floc absence.

Flow rate

During the sampling period, variations from the designed flow rate were experienced. Inlet flow rates were routinely low; this was due to the continuous clogging of pipes by sludge at the inlet works. This consequently affected the inflows into the AFP and HRAOP 1 and 2. Inflows into HRAOP were also affected by disruptions/shifts that occurred in the lever that splits the flow from the first settler into HRAOP 2. Figure 3.18 presents the variations experienced in flow rates throughout the sampling period. Flow rates were actively measured from the 17th of July when the continuous flow was resumed. Raw influent flow rate recorded minimum and maximum values of 16 ± 0.01 $kL^{\cdot}\ d^{-1}$ and 73.2 ± 0.06 $kL^{\cdot}\ d^{-1}$ and yielded a mean of 59.9 ± 2.8 $kL\ d^{-1}$. Subsequently the averages for HRAOP 1 inlet (decant from the PFP), ASP 1 weir decant, and the final effluent were 57.2 ± 3.1 $kL^{\cdot}\ d^{-1}$, 54.2 ± 3.6 $kL^{\cdot}\ d^{-1}$ and 30.2 ± 3.1 $kL^{\cdot}\ d^{-1}$, respectively.

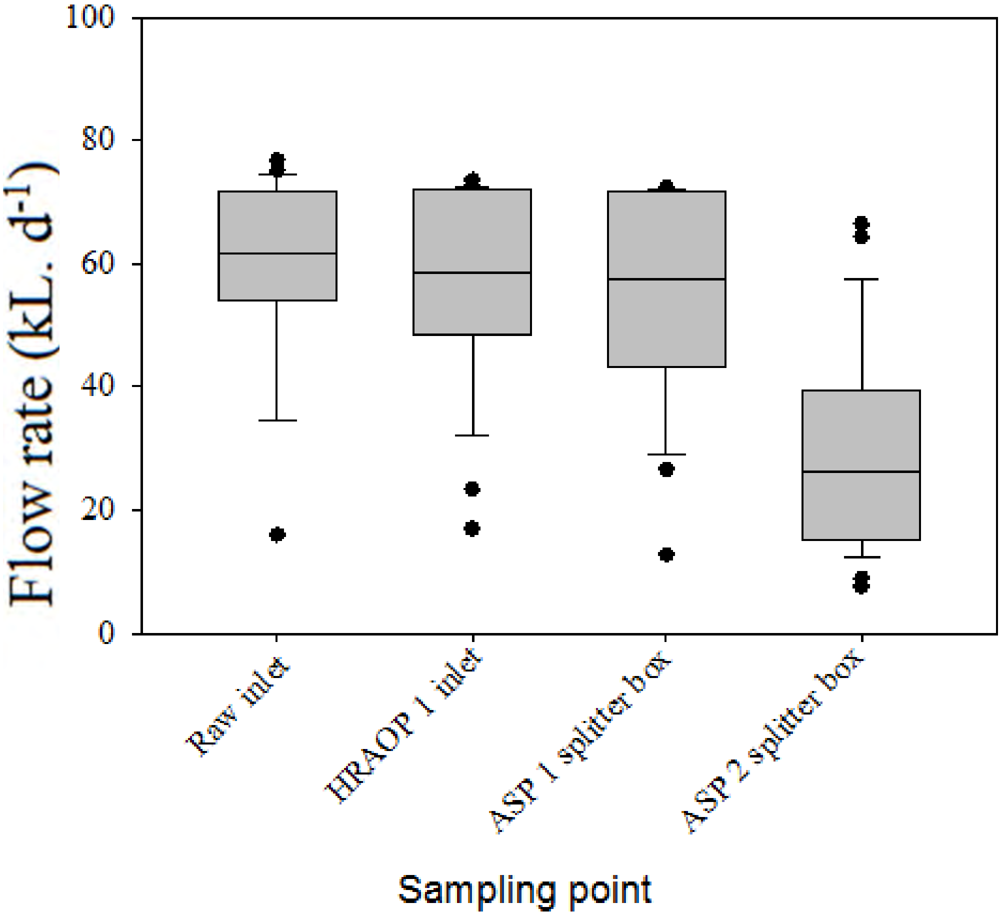


Figure 3:18: Flow rates across integrated algae ponds treatment unit. Data presented are an average of the sampling period. Five replicates were carried out for determinations during each sampling. Data are an average of 43 sampling times.

3.3 Conclusion

Experiments detailed above revealed that the Belmont Valley IAPS still performs as expected of IAPS without a tertiary treatment component producing an effluent suitable for the irrigation of any land or property in volume of no more than 500 kL on any given day. The treatment plant effectively reduced nutrients, faecal coliforms, COD and TSS; however, pH and DO showed a gradual increase from AFP to final effluent. The final effluent displayed consistently high TSS, and COD whilst nutrients and physical parameters were within discharge limits; however, regular settled biomass removal is an effective exercise to control TSS within discharge limits. Faecal coliforms within the final effluent exhibited variations with seasons. The lowest faecal coliform counts were recorded during the hot spring and summer months. The biomass productivity recorded during this study is comparable to previous studies on the Belmont Valley biomass; however, biomass recovery within the ASP was greatly influenced by the presence or absence of MaB-flocs. The microbial species composition did not show variation from previous findings on HRAOP however, microbial species richness during periods of no MaB floc was low. Careful plant monitoring is vital for the effective management of IAPS and troubleshooting exercises are very effective when a normal state has been established because they allow the operator to know when deviations have occurred and how to correct them.

Chapter 4 : Development of a process audit framework

4.1 Introduction

When correctly configured, adequately managed, with appropriate tertiary treatment units, IAPS are capable of producing a quality effluent that meets discharge limits, biomass with valorisable potential and biogas (Cowan et al. 2016). IAPS have also been promoted as low cost, low energy input and sustainable systems with great potential in the 21st century (Oswald 1995). IAPS is a biological system in which a number of processes occur; the system makes use of physical components in order to treat wastewater efficiently. In order to carry out an effective and thorough audit of a wastewater treatment plant the ability of the system to meet regulatory standards of discharge must be assessed, pollution removal capability calculated, biomass characteristics identified and operating protocol monitored. A process audit of IAPS, therefore, will first define the performance efficiency of the plant, include an analysis of final effluent produced according to discharge parameters in order to establish compliance/ non-compliance and assess biomass quantity and quality so that the possible avenues of reuse are determined.

The previous chapter gave insight on the performance efficiency of Belmont Valley IAPS. As expected, the effluent did not always meet discharge regulations for discharge to a water resource and was occasionally characterised by high TSS and COD. Excess TSS, however, could easily be reduced by more efficient management of settled biomass, as shown in the previous Chapter. Even so, biomass produced was approximately 130 kg ha^{-1} d^{-1} and was composed mostly of protein. Plant performance data allowed for the identification of treatment efficiency parameters which could be used to more closely monitor plant management protocol and, establish further control processes and key performance indicators in order to adjust, correct and improve the efficiency of treatment and effluent water quality. In this Chapter, a process audit protocol for IAPS is developed. The water quality and efficiency of IAPS results (Chapter 3) will be used to develop the process audit framework. An analysis of costs incurred for plant maintenance and operation during the period 2009-2017 is also given.

4.2 Results

4.2.1 Audit scope – To define performance efficiency of the plant. To determine if effluent produced meets standards set out in the Water Act. To identify strategies which maintain plant performance efficiency, improve efficiency, and to identify the root cause of plant performance deviances.

4.2.2 Process audit turtle diagram

The process audit turtle diagram was prepared using the ISO 9001 management systems turtle diagram (Figure 2.2) template as a guideline.

The IAPS process begins with the inflow of raw sewage into the AFP; as such other process inputs, therefore, include head of works which is responsible for the screening of effluent and other unit components of the system, i.e. AFP, HRAOPs, ASPs and drying beds. In the IAPS, process equipment used includes the HRAOP paddlewheels, motors and gears which drive the paddlewheels and head of works. During the course of this study, pump repair was carried out by the chief technician, although an electrician had to be called in from time to time. No form of pump monitoring protocol was carried out when the plant was operating normally. However, due to regular breakdown pump replacement is suggested not only to ease the process but to reduce operating costs as will be described later.

Influent goes through a series of ponds, each with a specific role, as described in Figure 4.2 below. Performance efficiency for the Belmont Valley demonstration scale IAPS was previously documented with reductions of 87% in COD, 76% in phosphate, 55% in nitrogen and 99.999% in *E.coli* (Wells et al. 2005). The paddlewheel was constantly monitored for loose screws as part of a daily monitoring routine. Based on results presented in Chapter 3, process outputs include an effluent suitable for irrigation of any land or property at a volume of no more than 500 kL $d^{-1.}$ The analysis of effluent was carried out according to Standard Methods using accepted Merck test kits. A valorisable biomass which is mostly protein is also produced as a by-product of the IAPS process. The most economical use for biomass produced is methane production. Methane rich biogas is also produced in the AFP as a process output. In order for a process to function efficiently and effectively, it must be managed properly and control processes identified and monitored.

Flow rate measurements must be carried out as part of daily plant management and control. Wastewater flow was estimated throughout the sampling period using the bucket and stopwatch method as described by (Spellman 2013). The time it took to fill a 10 L bucket was recorded, and the flow rate was calculated and is presented as kilolitres per day kL d^{-1}. Flow measurements were taken at the inlet point (SB 1), HRAOP 1 inlet (decant from the AFP), ASP 1 weir decant and the final effluent at the second splitter box. Measurements were repeated five times, and the average was recorded. Flow rate measurements allowed for the estimation of the HRT in the PFP and both HRAOPs. When inflow rates deviated from the set flow rates as detailed by (Rose et al. 2007), the flow rate was adjusted accordingly.

Algae settling pond cleaning and weir debris removal for the Belmont Valley IAPS was scheduled for emptying out and cleaning every fortnight to harvest microalgae bacterial biomass and avoid decomposition of settled solids, thereby maintaining effective unit performance (Cowan et al. 2016). Cleaning dates and estimation of the next cleaning date were therefore noted as part of standard operating procedures. Results showed that removal of settled biomass could be used to control final effluent levels below limits of discharge.

Debris accumulation at the ASP weirs was checked occasionally and immediately removed. Delayed removal of settled biomass increased the TSS of final effluent to levels beyond discharge limits, whilst debris in the weir clogged piped and reduced flow rates. Stones restrict the movement of the paddlewheel therefore occasional checking and their removal should be carried out for free and unrestricted movement of paddlewheels.

Factors considered under process control include linear velocity, volute measurements and analysis of outputs**.** Volute measurements were started during the period of no floc formation (June 2018- September 2018). The purpose of this tool was to establish the type of flow i.e linear or turbulent within the HRAOP. A turbulent flow has a negative effect on MaB floc formation hence this exercise must be carried out to ensure that MaB-flocs are sustained. The paddlewheels were stopped for this procedure. The distance from the depression including the width of the paddlewheel was measured and recorded. The width of the paddlewheel was also measured and recorded. The volute was recorded as the difference between these values. This exercise should be carried out when repairs to the paddlewheel are done or when need arises as part of trouble shooting.

Linear velocity of water flow is an estimation of the linear velocity within the HRAOP. It is carried out by measuring the time it takes for a floating plastic table tennis ball to travel around the HRAOP. A table tennis ball is thrown randomly into the pond and the moment it landed on the water surface a timer is started. The time it takes the ball to travel from the front of the paddlewheel around the HRAOP till it returns to the paddle wheel is recorded. Velocity is then calculated according to Equation 2.6 (Chapter 2). This exercise is repeated 15 times for one measurement. The purpose of the exercise is to estimate the rotational speed of the paddlewheel, which should be maintained at 300 mm s^{-1}.

Analysis of outputs should be carried out to determine plant efficiency and to determine whether outputs comply to set standards. During the sampling period, key indicators of plant performance identified included an odour free AFP, species diversity within the HRAOP which corresponded with the presence of highly settleable MaB-flocs, a clear final effluent and algal genera present. The presence of grazers within the HRAOP has been shown to reduce biomass productivity. Presence of MaB-flocs improved settleability within the ASP, reduced final effluent TSS and improved clarity of final effluent. These indicators represent the optimum operating conditions of IAPS and should be checked regularly. Where deviations occur control processes should be checked and adjusted were necessary.

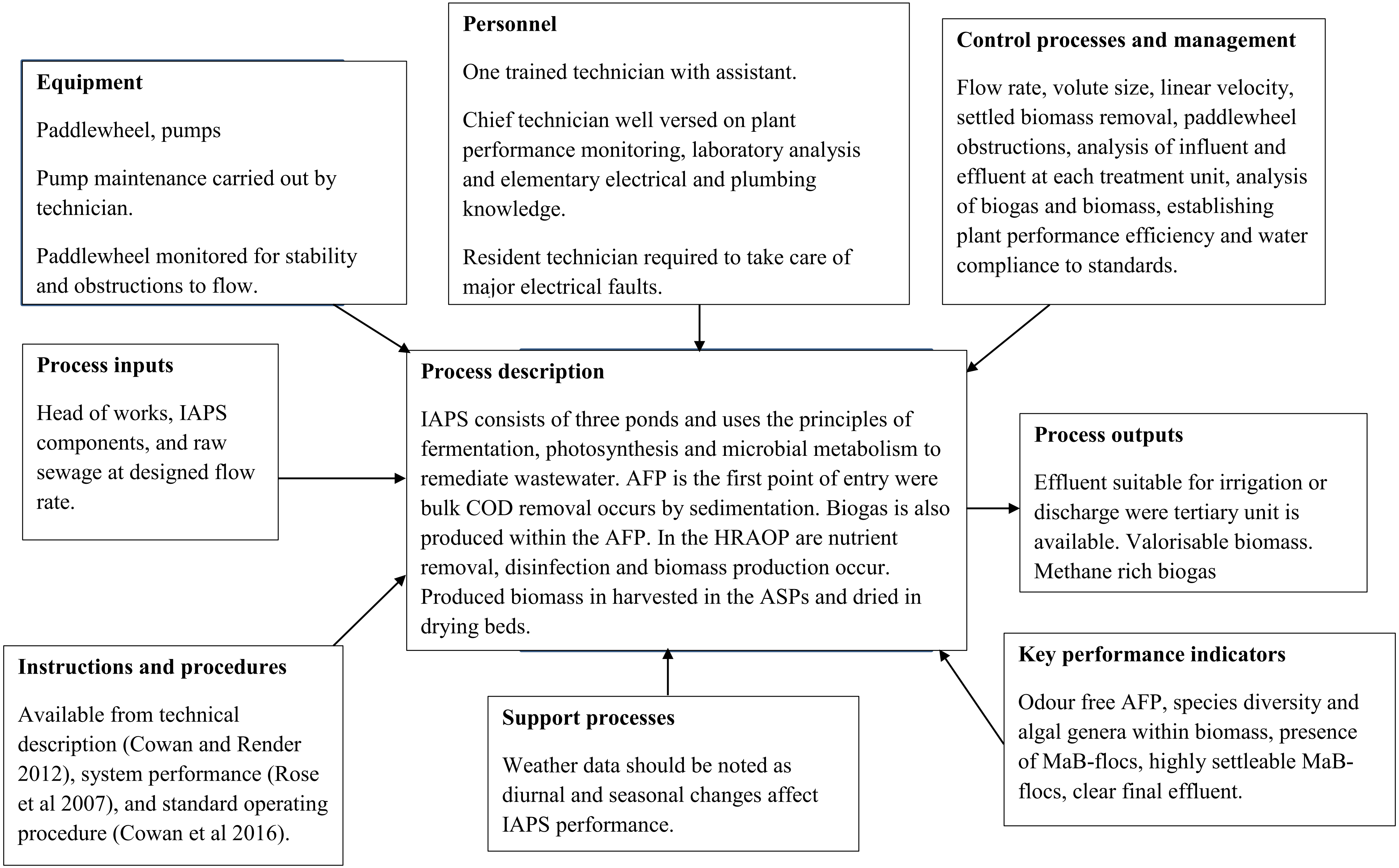


Figure 4:1: Turtle diagram to assist in the process audit of IAPS

4.2.3 Preparation of a process audit flow diagram

The process flow diagram presented below in Figure 4.3 suggests the protocol for auditing IAPS. The process flow chart was created to assess plant performance and verify compliance of outputs to set limits. All outputs of the plant that is water, biomass and biogas produced can be assessed in the following manner. The process flow diagram presented in Russell (2009) was used as a guideline to prepare the process flow for IAPS. The first step during the audit is to check plant design and operational parameters to ensure that the plant is functioning as it was designed; the exercises carried out here include flow rate, linear velocity and volute measurement, ensuring that proper plant management is being followed as described in the turtle diagram is vital from the beginning.

Once these are verified a simple yes/no decision step follows. Where control processes are not being followed they are adjusted and corrected accordingly. If control processes and management steps are being adequately followed an analysis water based on standard wastewater parameters is carried out for influent and all unit operations effluent, this is done to establish unit operations efficiency and check compliance of final effluent to discharge limit. Quantification and assessment of biomass and biogas produced should also be carried out. If everything is functioning according to standards the process audit checklist described below should be filled out. Where variances in performance and output‘s quantity and quality are encountered trouble shooting exercises should be carried out and this will include assessing the key performance indicators, checking process control processes and ensuring proper management. The deviations are then adjusted and the flow diagram followed again.

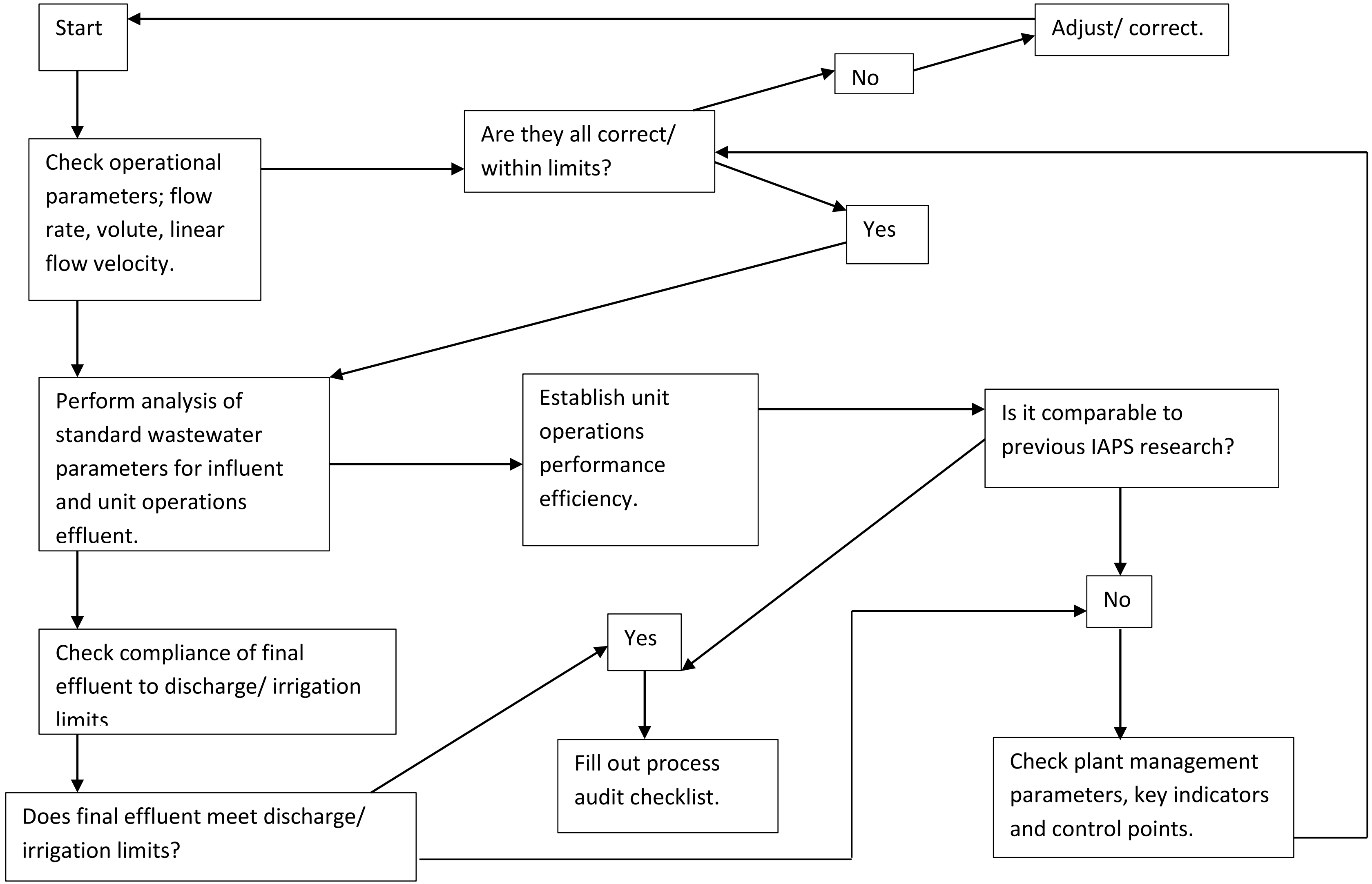


Figure 4:2: Process audit flow diagram for IAPS.

4.3.4 Process audit checklist

The checklist presented below is a handy tool to check that requirements are met and to optimise the process; however, this tool cannot be used alone. As described above, this checklist should be filled after system checks and output analysis have been carried out, meaning components of the turtle diagram should be first identified and the process flow diagram followed. The ISO 9001 management systems checklist template was used as a guideline for the preparation of the audit checklist for IAPS. The checklist questions were prepared to determine whether the process is operating properly and to identify if the system is effectively managed to produce outputs. Also, actions carried out during the process flow diagram are also accounted for and assessed in the checklist. All aspects mentioned in the turtle diagram are included to ensure the proper flow, maintenance and management of the system. The scoring criteria used is as follows:- compliant means the requirement fully meets the standard and there are no variances or deviances, minor non-compliance means there is a slight variance to the set standards while a major non – compliance shows that the process is operating out of scope and steps should be taken to immediately correct this. Opportunities for improvement include any action that can be taken to improve the system process flow and output generation.

Table 4:1: Checklist questions for the process audit of Integrated Algae Pond Systems.

Question	Audit Question	Compliant	OFI	Minor N/C	Major N/C	Audit evidence	Opportunities for improvement
1	Is the plant process defined and documented?						
2	Is the plant process flow identified?						
3	Are process equipment functioning, maintained and fit for use?						
4	Are there indications that process inputs are clear to plant operators?						
5	Is there evidence that plant management activities are known?						
6	Are plant management activities being carried out?						
7	Are there indications that process outputs are accurately defined and understood by all plant operators?						
8	Is plant performance efficiency well defined?						
9	Is plant performance assessed?						
10	Is the final effluent produced defined?						
11	Is there evidence that final effluent is being monitored?						
12	Is the quantity and quality of biomass produced defined?						
13	Are there indications that biomass quantity and quality is being measured?						

14	Is the quantity and quality of biogas produces defined?
15	Are there indications of biogas production within the AFP?
16	Is the quantity and quality of biogas produced being monitored?
17	Have procedures and instructions been established as needed to control the process?
18	Are plant technicians aware of the plant control process procedures and instructions?
19	Are control procedures and instructions being followed?
20	Is the information on plant control processes being documented?
21	Are there steps to follow when variance to plant performance, control processes and key indicators are experienced?
22	Are key performance indicators established and documented?
23	Is there evidence that key performance indicators established are known and identified by operators?
24	Is there a standard operating procedure for the analysis, quantification and identification of outputs?
25	Are sampling points and sampling frequencies defined?
26	Is there evidence that sampling

	points are known by operators?	
27	Is there evidence of management commitment and involvement?	
28	Is there feedback from technicians to management?	
29	How many plant operators are available, is this adequate for the management and maintenance of the plant?	
30	Are plant technician roles defined?	
31	Are operators adequately trained to carry out roles and responsibilities?	
32	Are plant ground maintained?	
33	Is there adequate equipment/tools to carry out plant management duties?	
34	Is the observed process performance consistent the meta data?	
35	Is biomass effectively dried and stored adequately?	
36	Are all actions pertaining to plant management, control processes, water quality, biomass and biogas quantity and quality recorded and well documented?	
37	Where the process is not performing is the evidence of data analysis to determine the root cause?	
38	Where system adjustments have been carried out, have the actions	

	taken been demonstrated as effective?

OFI= opportunities for improvement. N/C = Non compliance

4.2.6 Operational and Maintenance Costs

The overall cost of running the IAPS system over the past 9 years has amounts to a little over half a million ZAR. The maintenance costs of the system have been dominated by pump repairs, and replacements whilst operational costs include electricity consumption costs of the system. Bulk costs come from the electrical charges of the system, which utilises 34 126 kWh.y^{-1}. The total amount spent over the 9-year period towards maintenance alone was R171 859.52 and yielded an average of R19 096 y^{-1}, as shown in Table 4.2· The cost of treating water per person equivalent per year was less than ZAR200. The average cost was R123.87 $PE^{-1}.y^{-1}$ with minimum and maximum values of ZAR79.96 $PE^{-1}.y^{-1}$ and ZAR184.72 $PE^{-1}y^{-1}$ respectively. The cost of treating water per litre per year using IAPS is less than one cent.

Table 4:2: Operation and maintenance costs incurred for Belmont Valley Integrated Algae Pond System (2009-2017)

Year	Operational costs (ZAR. y^{-1})	Maintenance costs (ZAR. y^{-1})	Operation and maintenance costs (ZAR)	
			Per person equivalent (ZAR. $PE^{-1}.y^{-1}$)	Per litre treated (ZAR. $L^{-1}.y^{-1}$)
2009	36 514.82	3 467.34	79.96	0.002
2010	37 538.60	17 575.12	110.23	0.002
2011	38 562.38	16 832.74	110.79	0.002
2012	37 879.86	38 722.87	153.21	0.003
2013	40 951.2	7 697.98	97.30	0.002
2014	42 657.50	4 227.96	93.77	0.002
2015	44 022.54	48 340.66	184.72	0.003
2016	53 236.56	16 788.41	140.05	0.003
2017	54 260.34	18 206.44	144.93	0.003
Average	42 847.09	19 095.50	123.87	0.002

4.3 Conclusion

The process audit framework established involved three techniques, a turtle diagram, a flow diagram and a checklist. When combined, these tools can assist in the effective management of IAPS and improve overall plant performance. The 9 years of operating and maintenance data revealed that the cost of managing IAPS is low despite the continual maintenance carried out for the Belmont Valley IAPS.

Chapter 5 : General Discussion and conclusion

The experiments described in earlier chapters were carried out in order to establish the parameters needed to develop a process audit for integrated algal pond systems. IAPS is a system in which a number of processes (fermentation, photosynthesis, aeration and sedimentation) occur, sewage (input) goes through a series of ponds and is ‗transformed' to water with potential for reuse and/or recycle which is linked to the production of biogas and biomass. The experiments described in this research focus on the water quality aspect of IAPS, and biomass production, however. Outcomes of earlier work on IAPS are taken as the agreed upon/set limit or reference points for the process audit framework.

Results obtained throughout the course of the study showed the reliability and consistency of the system as system performance, final water quality and biomass microbial composition did not differ significantly from previous research findings (Rose et al. 2007; Johnson 2010; Cowan et al. 2016; Jimoh 2017). However, operation and maintenance costs were shown to be alarmingly high.

5.1 IAPS treatment efficiency

The plant performance results during the 9 month sampling are consistent with previous results reported from earlier Belmont Valley IAPS and other IAPS research (Banat et al. 1990; Tadesse et al. 2004; Wells et al. 2005). This is impressive for a plant nearing the end of its design lifespan and affirms the advantage of IAPS when compared to traditional waste stabilisation ponds whose performance efficiency reduces with age (Buchanan 2014). The life expectancy of IAPS is 25 years, and the Belmont Valley research station has been operational for 23 years (Nemadire 2011). Results show that IAPS is effective in reducing the concentrations of standard water parameters with the exception of nitrate and pH which increased throughout the system

IAPS was consistently effective in reducing COD throughout the whole sampling period. In particular, as expected, the leading treatment unit (AFP) recorded the greatest COD reductions and exhibited buffering capacity despite the fluctuations in influent COD concentration. This is basically in agreement with previous research which showed that high levels of organic inflow did not significantly affect the performance of the AFP (Rose et al. 2007). The remaining treatment units, i.e. HRAOP 1 and 2 and ASP 1 and 2, had a combined COD reduction efficiency of about 50%. This is typical in IAPS as HRAOPs have been shown to contribute more to nutrient removal than COD removal (Oswald and Asce 1990). TSS removal in the AFP was consistent during this research whilst removal in the ASP showed great dependence on the availability of MaB floc and species composition. Banat and co-workers reported good settling rates when *Scenedesmus* was dominant, the opposite was observed with the Belmont Valley IAPS. Nitrate concentration increases observed within the system are likely due to nitrification (Rose et al. 2007). Although faecal coliform removal efficiency in this present study was higher than the more recent studies, results obtained are comparable to those attained for the evaluation of the system during the first nine years (Rose *et al.*, 2007; Mambo *et al.*, 2014; Jimoh, 2017). It should also be noted that the pH values during this research were higher in the HRAOPs than previously reported (Jimoh 2017).

5.2 Effluent water quality

As expected, consistently high TSS and COD were recorded in the final effluent. Faecal coliforms were as expected above the general limits for discharge during the autumn and winter months, whilst electrical conductivity, nitrates, phosphates and ammonium were in accordance with the general authorisation limits for discharge. The average TSS concentration was much higher than that experienced in previous years (Table 1.2); this is due to reduced efficiency in the ASP caused by loss of MaB-flocs and reduced species diversity experienced during June 2018 – September 2018. The conditions prevalent in HRAOP support the formation of stable readily settleable MaB-flocs, resulting in high removal efficiencies of biomass by sedimentation in the ASP (Banat et al. 1990). A similar observation was recorded by Sutherland and co-workers (2018), who noted that the absence of microbial flocs resulted in poor settleability efficiencies in algae harvesting ponds at commercial scale. Highly settleable MaB-flocs have been continuously observed during the operation of the Belmont Valley IAPS (Rose et al. 2007; Jimoh and Cowan 2017). It is therefore likely that the absence of MaB-flocs reduced settling efficiency in the ASP resulting in excessively high TSS in its discharge. COD removal has timelessly remained the drawback of IAPS with final effluent COD levels often exceeding the recommended discharge standard of 75 mg. L^{-1}. A recent study attributed elevated COD levels in IAPS to the accumulation of soluble extracellular polymeric substances (EPS) in the HRAOP. Soluble EPS production of 116 ± 4 mg. L^{-1} has been reported for the Belmont Valley HRAOP 2, and the average COD recorded during this period of study was 81.3 ± 6.7 mg. L^{-1}. Although the presence of EPS in the final effluent has undesirable effects to water quality with regards to COD levels, EPS are high value products and their production within the system provides other avenues for IAPS beneficiation (Jimoh et al. 2019). Inefficient faecal coliform removal during the winter and autumn months negatively impacted the average faecal coliform count. This observation is similar to that reported by Rose and co-workers (2007); however, the average faecal coliform count during this research is much higher. Factors which affect faecal coliform removal include pH, DO, temperature, predation, solar irradiance and plant configuration (Davies-Colley et al. 2003; Craggs et al. 2004; Fallowfield et al. 1996; El Hamouri et al. 1994). Consistent with previous evaluations, disinfection showed a strong correlation with pH ($r = 0.65$). pH values above the DWA limit of 9.5 are a common occurrence for HRAOP (El Hamouri et al. 1994; Craggs et al. 2014; Mambo et al. 2014). The results of this research further affirm the need for a tertiary treatment unit to complete the system, thus ensuring that discharge limits are consistently met with regards to COD and faecal coliforms. TSS limits can be controlled by changes in plant operational procedures, although this applies only when MaB-flocs are present as discussed below. Where chlorination is chosen as a further disinfection method, results suggest that it can be omitted or dosing concentrations and or contact time greatly decreased during the summer months.

5.3 Biomass composition and productivity

Algae research has significantly advanced over the years, and the interest in microalgae biomass is due to its vast application to day-to-day life and its ability to be produced on unproductive land at a faster growth rate than other crops (Campbell et al. 2011). IAPS provides a low-cost option in the production of biomass as it is produced as a by-product of wastewater treatment, but the question is, is it enough and what can it be used for? The biomass produced in this system is about 48 tonnes. ha^{-1} year $^{-1}$ but the actual recovered biomass is 35 tonnes. ha^{-1} year $^{-1}$. Despite these losses, the recoverable yield is still within the range of productivities based on experimental measurements reported in the literature, which range from 20-110 tonnes. $ha^{-1}year^{-1}$ (Slegers et al. 2013). This is much lower than the upper limit value of 73 tonnes. $ha^{-1}year^{-1}$ observed for the system (Jimoh and Cowan 2017). This, however, corroborates that seasonal and diurnal changes occur within the system (Jimoh 2017). Of as much importance as quantity, biomass composition determines its most practical and possible uses. Reports suggest that a lipid rich biomass can be trans-esterified into biodiesel, while biomass with a greater proportion of carbohydrate is often fermented to ethanol (Brennan and Owende 2010). The biomass produced in HRAOP 2 consists mostly of carbohydrates and proteins and has a low lipid concentration. The low lipid content makes the conversion to biodiesel economically unsuitable; hence the most economic conversion is through anaerobic digestion to produce methane, as it utilizes both the carbohydrate and protein constituents (Bohutskyi et al. 2018). However, there might be a need for co-digestion to enhance methane production due to the high protein content of the MaB floc biomass. A high protein concentration has a negative impact on methane production; however, co-digestion with a carbon-rich additive has been shown to improve productivity (Yen and Brune 2007).

Gravity sedimentation in ASP is an economic low-cost separation mechanism employed in IAPS and efficiency rates as high as 80% have been recorded (Rose et al. 2007). Hence highly settleable biomass is crucial not only for good water quality but also to ensure high harvesting efficiencies to enable the economic reuse of the biomass. IAPS generate MaB-flocs which are easily separated from the liquid fraction by sedimentation. Stability and formation of MaB floc is affected by cell surface properties of algae and EPS whilst the ability of biomass to settle is affected by the species composition (Gutierrez et al. 2016; Gutzeit et al. 2005). During periods of no MaB floc formation, settleability of biomass was very poor. This resulted in poor water quality with regards to TSS concentration and biomass losses to final effluent of up to 33%.

Although some research reports good settling efficiencies for *Scenedesmus* cultures (Banat et al. 1990; Su et al. 2012; Guo et al. 2013), the results obtained during this research corroborate the observations of Hu and coworkers (2017) who reported a poor settling efficiency of 22.5% for a culture dominated by *Scenedesmus* and *Chlorella*. Reports also show that the presence of bristles on *Scenedesmus* aid in buoyancy and ensure that this microalgae remains in suspension for long time periods when left in a quiescent place, hence making gravity sedimentation in the ASP inefficient (Conway and Trainor 1972; Trainor and Egan 1988). It was of interest to note the inability of the *Scenedesmus* strain to flocculate and how this corresponded with marginally lower EPS concentration within the ponds (results not included) than previously reported by Jimoh (2017), and the absence of predators. This suggests that EPS concentration and predation may be responsible for inducing aggregation of microalgae in this HRAOP. The importance of microbial aggregation has been demonstrated not only in algae-based systems but also in other biological wastewater treatment technologies (Bossier and Verstraete 1996; Zhu et al. 2012; Van Den Hende 2014). The presence of MaB floc greatly improved settleability and increases their recovery within the ASP. Flocculation into aggregates coincided with an increase in microbial diversity within the ponds, thus bringing the question of how microbial communities impact on HRAOP performance.

Enumeration of microorganisms is a common analysis in activated sludge plants to give a measure of plant performance and might be a useful tool which can also be adapted for IAPS plant performance management (Al-Shahwani and Horan 1991; Madoni 1994). The microbial composition was similar to that previously observed for the Belmont Valley and other wastewater treatment HRAOPs (Banat et al. 1990; Johnson 2010; Sutherland et al. 2017). Microalgae are versatile bioresources with broad industrial use. They may be applied in the production of energy and high value products which include pigments, omega-3 fatty acids, antioxidants, and vitamins (Malik et al. 2019). The diversity of microorganisms found in HRAOP opens avenues for the production of biofertiliser. It has also been demonstrated that the type of microalgae used as a substrate for methane production affects the quantity of methane produced (Arcila and Buitron 2016). Research is underway to determine the contribution of the bacterial component to IAPS and their potential uses as a bio-fertiliser.

5.4 Plant management and troubleshooting

Fluctuations in hydraulic retention time did not appear to negatively impact the overall performance of the plant with regards to standard water parameters, thus showing the robustness of the system. However, low inflow rates had a positive impact on settleability whilst inflows above the recommended 37.5 kL d^{-1} entering into the HRAOP 2 reduced the efficiency in the ASP. Algae settling pond cleaning reduced the concentration of suspended solids in the treated effluent to levels acceptable for discharge; however, the time interval adopted as part of plant management protocol should be reduced from two weeks to at most one week to ensure that TSS levels remain compliant. Presence of zooplankton in high population densities had a positive effect on water quality. Daphnia presence ensured that TSS concentration remained within compliance range after one week of pond cleaning and this corroborates previous reports that show that grazers have the potential to reduce algae population densities within a few days (Montemezzani et al. 2016; Day et al. 2017). This provides possible avenues for the use of phytoplankton grazers in polishing IAPS effluent, provided that these grazers select only for the smaller non-settleable algae as grazers able to feed on MaB-flocs would have a marked effect on biomass recovery. Grazers have been shown to select prey based on size (Andersson et al. 1986; Epstein and Shiaris 1992). The disintegration of settled biomass highlighted that regular harvesting and ASP cleaning is essential to avoid loss of biomass to final effluent. This, however, contradicts earlier findings which showed that algae could remain concentrated within the settling ponds for long time periods (Oswald and Asce 1990; Rose et al. 2007).

The trouble shooting exercises administered for this study did not give an insight into the reason for the disappearance of MaB-flocs but rather established the key control parameters that are necessary for the normal performance within the HRAOP and ASP. This provided good data which will be useful in detecting future operational problems. Previous studies do show that formation of MaB-flocs is not always stable (Gutzeit et al. 2005; Sutherland et al. 2018b), and the reasons for this is not very clear. However, it may be suggested that the low nutrient concentrations prevailing during the recycling period caused starvation resulting in EPS being used as a food source thus disintegrating the MaB floc structure (Zhang and Bishop 2003). Observations during this study suggested that the microbial diversity and structure of the MLSS within the HRAOP is a useable measure for plant performance. Microbial diversity and structure of MLSS had a significant effect on final effluent turbidity, but not on COD, nutrient concentrations and physical parameters such as electric conductivity. These findings suggest that like in activated sludge systems the sludge biotic index (SBI) and sludge volume index (SVI) can be adapted as part of plant management strategies (Dick and Vesilind 1969; Madoni 1994).

5.5 Operation and maintenance costs of the Belmont Valley IAPS

Although overall plant performance was not compromised, constant plant maintenance has sacrificed reliability of the system and increased overall long-term costs. It should be noted that in this study, the bulk of the electrical costs accrued in running the IAPS are from a pump which returns the treated effluent back to the municipality. At commercial scale implementation of the IAPS, this pump would not be included; therefore running costs would be reduced. However, the energy input for the head of works has not been included in this study.

For a technology to be considered sustainable, it should be efficient, economic and have low maintenance in the long run (Katuzika et al. 2012). The constant pump failures experienced over the period of this study increased the cost of plant maintenance, suggesting that this technology can be unreliable as during most maintenance jobs the plant had to be shut down and no influent was received. It should be noted that the IAPS installation used in this study is operated as a research facility and not as a service; thus priorities in terms of maintenance and operation would differ from that of a commercial facility. Nevertheless, despite this, the cost of running an IAPS still remains affordable compared to contemporary domestic wastewater treatment technologies which can be more energy intensive and require higher operator skills. Statistics from the Department of Water Affairs (2009) estimated the cost of operation per wastewater treatment facility then at over ZAR 4 million.y^{-1} (Cowan et al. 2016)· Performance efficiency of Belmont Valley IAPS shows that the technology performs as it was designed to perform, but because of lack of upgrades system deployment has been limited.

5.6 Process Audit Framework

The process audit framework suggested in this study includes three tools, a turtle diagram, a flow diagram and a checklist. The turtle diagram helps the auditor gain in depth knowledge of IAPS i.e. how it operates, the personnel involved, the expected outcomes, the key performance indicators, control processes and management protocols. The turtle diagram defines the plant capabilities and its efficiency, which then allows the auditor to follow the flow diagram. The flow diagram focuses on compliance of outputs and the factors that affect the quantity and quality of outputs. Lastly the checklist involves all aspects of the plant from inputs, the process, the outputs, personnel, equipment and management. It focuses on whether the system is functioning according to standards and suggests ways in which the system can be improved.

5.7 Conclusion

Globally there is a need to improve water quality and conserve water. Sound wastewater management through the application of appropriate wastewater treatment technologies could help achieve this goal (Bandao et al. 2013). The wastewater treatment industry is unfortunately largely dominated by grey infrastructure and the potential of nature-based solutions (NBS) has been neglected, overlooked or ignored. Despite the numerous advantages NBS offer over grey infrastructure, funding for the implementation of such technologies has remained low (WWAP 2018). There is a need to improve the uptake of NBS, and this can be achieved by increasing research outputs that fully explore the capabilities of such systems.

Integrated algal pond systems are an efficient wastewater technology capable of not only reclaiming water but also providing an opportunity for resource recovery in the form of biomass and methane rich biogas. It is important to note, however, that this system achieves secondary treatment efficiency; hence there is a requirement for the addition of a tertiary effluent treatment unit to meet the standards for discharge (Rawat et al. 2011). This can be a disadvantage for large cities considering the already massive footprint of the system, but for environments such as the peri-urban space, their large size can be overlooked. Peri-urban spaces are areas of rapid change located at the periphery of metropolitan areas. Traditionally the major land use of these areas was agriculture; however, due to the requirements of neighbouring urban areas, there is a switch to industrial and commercial use (Nkwanyana 2015). The IAPS can be most suitable for implementation in peri-urban spaces where sanitation, water for agriculture and energy needs can be provided by a single installation. The low electricity consumption should be highlighted especially for countries which rely on coal as a source of energy as this indirectly offsets greenhouse gas emissions generated during electricity production. Improvements in biomass recovery and harvesting, which are a weakness of this system, may be addressed by innovations similar to rotating algal biofilms, inclined gravity settlers and sustaining MaB-flocs making sure they do not disintegrate (Gutzeit et al. 2005; Wang et al. 2014; Kesaano 2015).

Although IAPS is a very dynamic system with a number of value adding activities, such as biomass production, nutrient recovery and energy production for example, and thus promoting this technology as only a wastewater treatment technology understates its true value compared to other wastewater treatment technologies. These competing wastewater treatment technologies may appear more attractive to municipalities since they are more conveniently sized, their construction and operational costs are known, and there is a familiarity with the technology within the civil engineering and construction industries; in addition, they are still able to meet effluent discharge limits if properly designed, constructed and operated. Moreover, peri urban spaces are constantly undergoing change which might mean that the system will eventually come across as being obsolete in circumstances where the value of land is linked to urban expansion. Nevertheless, if value can be derived from the exploitation of biomass harvested from the algal ponds in the form of fine chemicals, animal feed and biofertiliser, it gives this technology a strategic competitive advantage over other wastewater treatment technologies. Further development of this concept can lead to the establishment of a biorefinery, in this context a concept where a diverse microbial community is integrated into a single facility producing an array of products off a single resource. While treating a domestic effluent stream, sustainable energy in the form of biogas, bioethanol and biodiesel, can complement the production of biofertiliser or possibly other high value products, all while advancing to a zero-waste process and reducing environmental impacts associated with wastewater treatment.

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Appendices

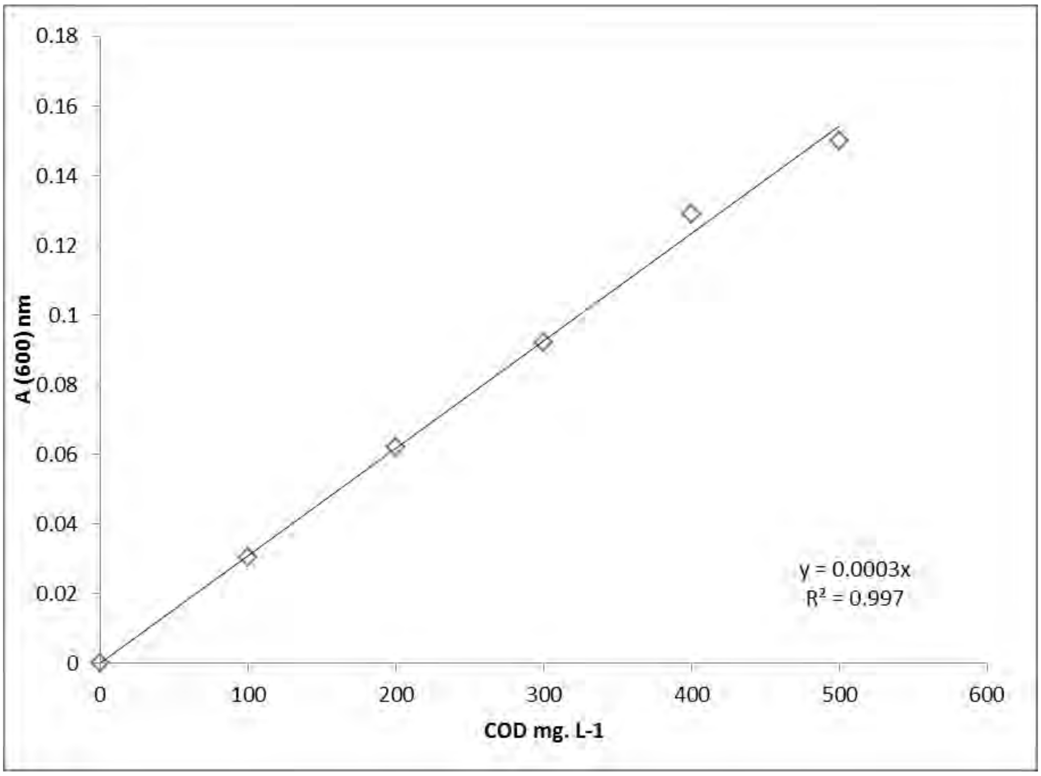

Figure A1: Graph showing the increasing concentrations of COD at a wavelength of 600 nm. The curve was used to interpolate unknown COD concentration in water samples.

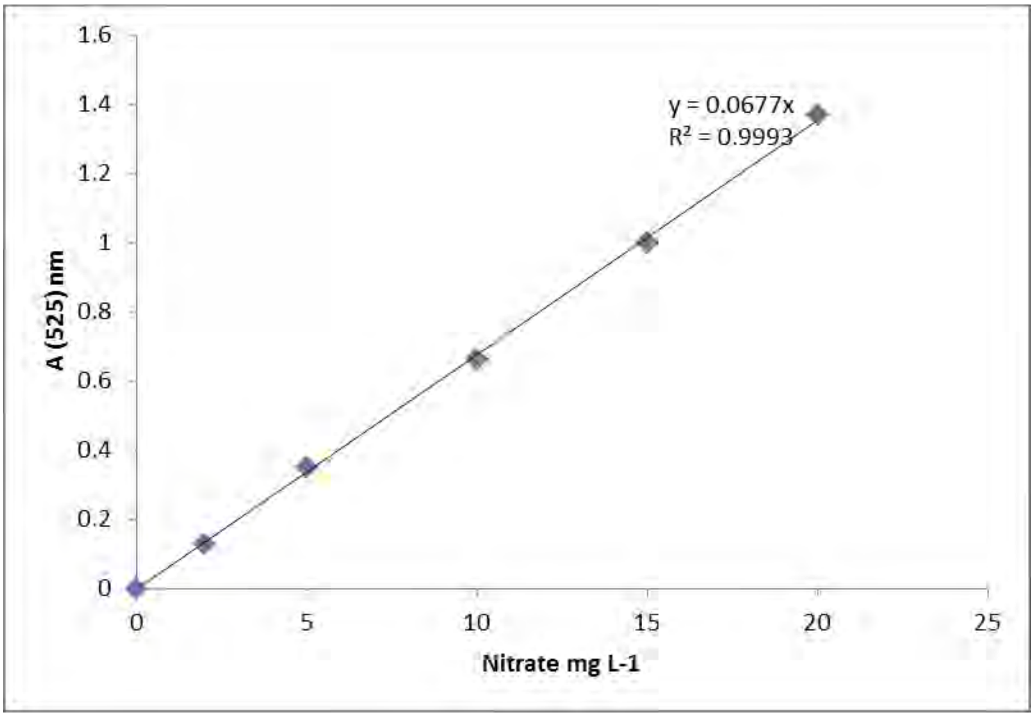

Figure A2: Graph showing the increasing concentrations of Nitrate-N at a wavelength of 525 nm. The curve was used to interpolate unknown Nitrate-N concentration in water samples.

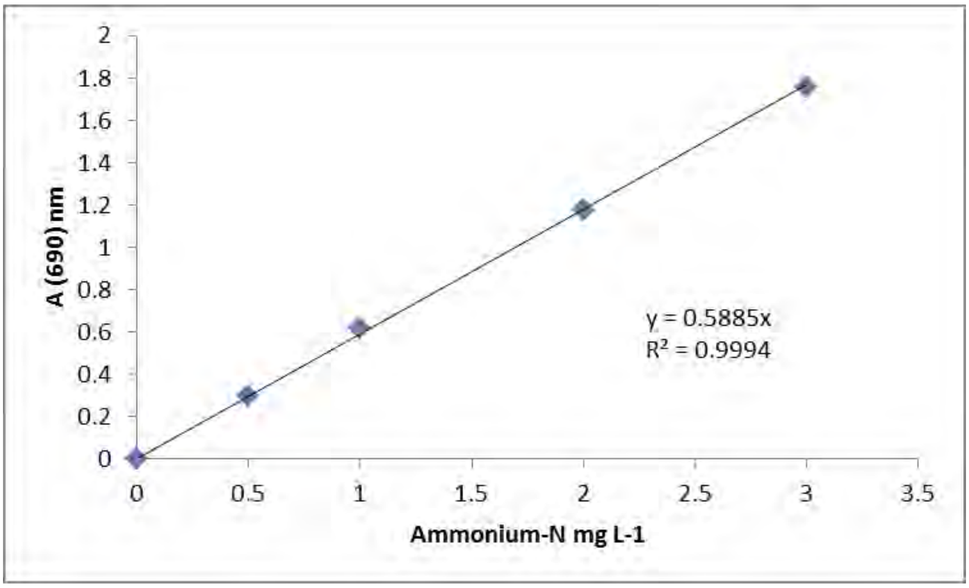

Figure A3: Graph showing the increasing concentrations of Ammonium-N at a wavelength of 690 nm. The curve was used to interpolate unknown Ammonium-N concentration in water samples.

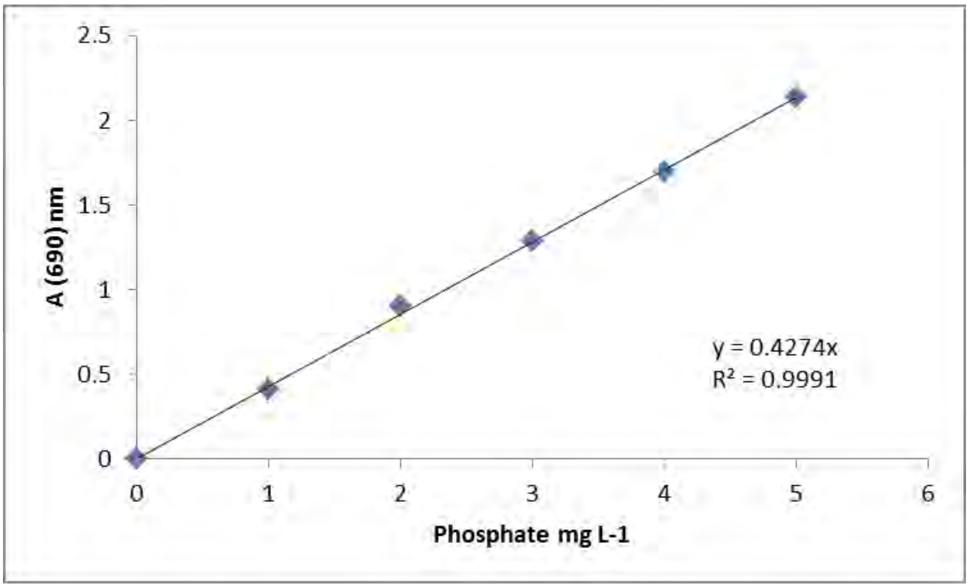

Figure A4: Graph showing the increasing concentrations of Phosphate-P at a wavelength of 690 nm. The curve was used to interpolate unknown Phosphate-P concentration in water samples.

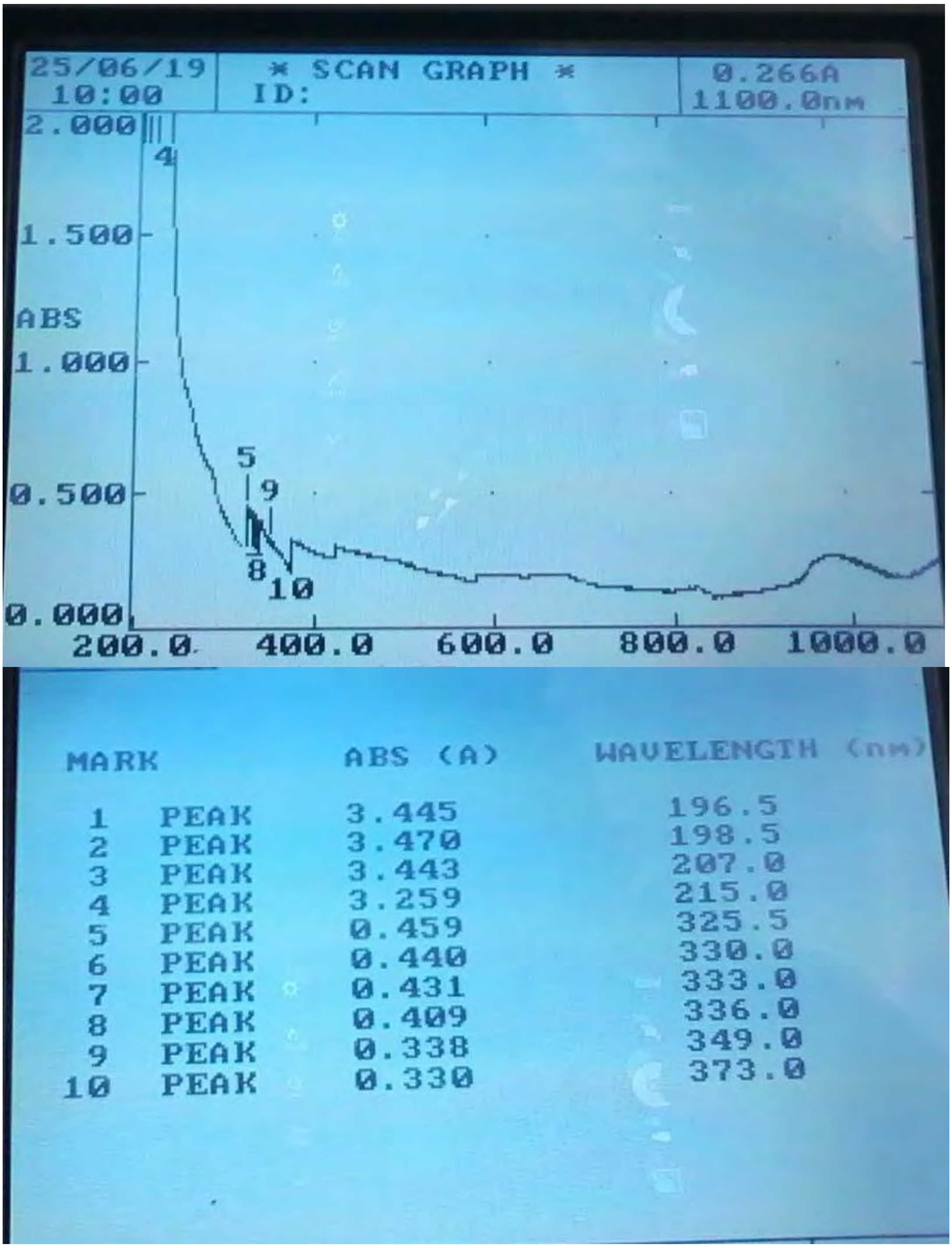

Figure A5: Spectrometric scan of HRAOP 2 water, used to determine possible wavelength for measuring optical density.

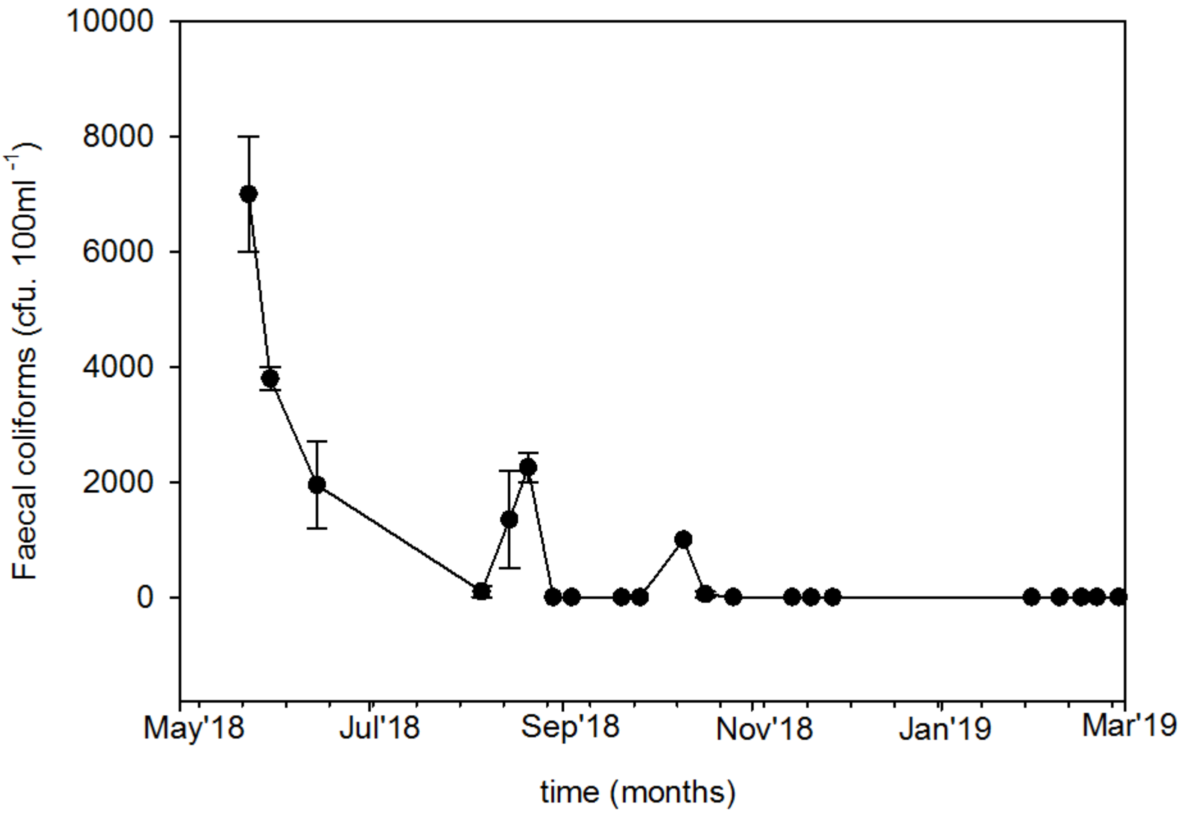

Figure A6: Changes in final effluent faecal coliform counts during sampling period. Bars indicate standard error.