Integrated Algae Pond Systems for the Treatment of Municipal Wastewater

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Abstract

Integrated algae pond systems are a derivation of the Oswald designed advanced integrated wastewater ponding system, and combine the use of anaerobic and aerobic bioprocesses to effect wastewater treatment. Integrated algae pond system technology was introduced to South Africa in 1996 and a pilot plant was designed and commissioned at the Belmont Valley wastewater treatment works in Grahamstown. Previous studies showed that this system delivered a final effluent superior to most pond systems deployed in South Africa but that it was unable to meet the general standard for nutrient removal and effluent discharge. This study was initiated to re-appraise integrated algae pond systems and to assess the potential of the technology as an effective municipal sewage treatment system.

Initially, integrated algae pond system technology for municipal sewage treatment was re-examined to; 1) determine the quality of the effluent over an extended operating period, and 2) to demonstrate that the discharge complies with the general limit values. Water quality after integrated algae pond system treatment was monitored for two periods. Results for the period September 2012 to May 2013 and February 2013 to February 2014 (presented in parentheses) indicated that *ortho*-phosphate, ammonium nitrogen, nitrate/nitrite nitrogen mean values were: 5.3 ± 2 (4.3 ± 1.7) mg.ℓ^{-1}, 2.9 ± 1 (2.6 ± 1.1) mg.ℓ^{-1}, and 12.4 ± 4 (2.3 ± 1.7) mg.ℓ^{-1} respectively. Mean chemical oxygen demand of the final treated water was 72.2 ± 13 (66.6 ± 12.7) mg.ℓ^{-1} and mean total suspended solid concentration was routinely above the general authorization limit at 34.5 ± 13.1 (35 ± 14.2) mg.ℓ^{-1} while faecal coliform counts were higher than expected. Shifts in dominance of the algae biocatalyst were also observed. These changes in species composition, inefficient removal of algae from the water column in the algae settling ponds appeared to be major factors contributing to elevated mean chemical oxygen demand 97.7 ± 15.7 mg.ℓ^{-1} and total suspended solid concentration 35.0 ± 12.3 mg.ℓ^{-1} measured between March 2013-November 2013.

Sequential filtration and coupled chemical and biochemical analyses of water samples abstracted from the outflow demonstrated that residual chemical oxygen demand comprises mostly soluble organic carbon in the form of carbohydrates (49.7 ± 15.9) mg.ℓ^{-1}, proteins (15.3 ± 3.9) mg.ℓ^{-1} and lipids (6.0 ± 0.1) mg.ℓ^{-1} derived presumably from algae biomass. Recalculation of the kinetic parameters coupled with a gap analysis revealed that the Belmont Valley wastewater treatment works integrated algae pond system is operated using an incorrect configuration. Rather, and as indicated in the original process design, water from

high rate algae oxidation pond A (i.e. 37.5 kℓ or 50 % of the hydraulic capacity of the system) should be returned to the integrated algae pond system inlet and not to the Belmont Valley wastewater treatment works to dilute the influent biochemical oxygen demand from 80 $kg.d^{-1}$ to the design specification of 40 $kg.d^{-1}$.

Results from the present study have confirmed that water quality following integrated algae pond system treatment of municipal sewage does not meet the limit values applicable to either irrigation of land or property up to 2 Mℓ on any given day or, discharge of up to 2 Mℓ into a water resource through a pipe, canal, sewer or other conduit. Whereas nutrient removal to within the limit values for both irrigation and discharge to a water resource was achieved, residual chemical oxygen demand and total suspended solid content was persistent and at values above the general authorization limits. It is concluded that further rigorous evaluation of integrated algae pond systems for treatment of municipal sewage requires at least one commercial scale plant, including an inlet works, maturation pond series (or similar), and chlorination, and operated according to design specification. Successful implementation at commercial scale will undoubtedly pave the way for further innovation and expedite transfer of this technology into the South African water and sanitation sector.

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Dad and Taku with my love and appreciation, I dedicate this thesis to you.

"Determine that the thing can and shall be done, and then we shall find the way."

~Abraham Lincoln~

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List of Abbreviations

AD	Anaerobic Digester
AFP	Advanced Facultative Pond
AIWPS	Advanced Integrated Waste-water Pond System
AP	Aerobic Pond
API	American Petroleum Institute
AS	Activated Sludge
ASP	Algae Settling Pond
BF	Biofiltration
BNR	Biological Nutrient Removal
BOD	Biochemical Oxygen Demand
C/F	Coagulation/Filtration
CA	California
CDI	Capacitive Deionization
COD	Chemical Oxygen Demand
CRF	Controlled Rock Filters
CW	Constructed Wetland
DAF	Dissolved Air Flotation
DB	Drying Bed
DO	Dissolved Oxygen
DON	Dissolved Organic Nitrogen
DWS	Department of Water and Sanitation
EBRU	Institute for Environmental Biotechnology, Rhodes University
EC	Electrical Conductivity
HLR	Hydraulic Loading Rate
HRAOP	High Rate Algae Oxidation Pond
HRP	High Rate Pond
HRT	Hydraulic Retention Time
I-HRAOP	Independent High Rate Algae Oxidation Pond
I-PD	In-Pond Digester
IAPS	Integrated Algae Pond System
IPCC	International Panel on Climate Change
MMF	Multi Media Filtration

MP	Maturation Pond
MOB	Methane Oxidizing Bacteria
MPN	Most Probable Number
MPS	Maturation Pond Series
NWMS	National Waste Management Strategy
OP	Oxidation Ponds
PB	Percolation Bed
PCD	Programmed Cell Death
PE	Population Equivalent
PFP	Primary Facultative Pond
PVC	Polyvinyl Chloride
RF	Rock Filter
SB	Splitter Box
SS	Suspended Solids
SSF	Slow Sand Filtration
TDS	Total Dissolved Solids
TF	Trickling Filter
TKN	Total Kjeldahl Nitrogen
TSS	Total Suspended Solids
TTU	Tertiary Treatment Unit
TVS	Total Volatile Solids
TVSS	Total Volatile Suspended Solids
UNEP	United Nations Environment Program
U.S.A	United States of America
UV	Ultra Violet
WRC	Water Research Commission
WSP	Waste Stabilization Pond
WWTW	Wastewater Treatment Works
WWT	Wastewater Treatment

1 Introduction

1.1 Project Background

The primary goal of the National Waste Management Strategy (NWMS) is the achievement of the objectives of the Waste Act (Republic of South Africa *Waste Act* 2008), which are in summary: 1, minimizing pollution, environmental degradation and the consumption of natural resources, 2, implementing the waste hierarchy, 3, balancing the need for ecologically sustainable development with economic and social development, and 4, promoting universal and affordable waste services (NWMS). Framed within the context of the overall goals, approach and regulatory model of the NWMS, the introduction of a new waste water treatment (WWT) technology requires demonstration of proficiency, education, and increased awareness amongst all stakeholders including the public at large, the three spheres of government, and the private sector. South African statutory, para-statutory and non-governmental organizations and citizens share a common concern regarding the national crisis relating to small and medium municipal wastewater treatment works (WWTW), many of which are currently in a state of disrepair and implicated in contributing to disease outbreak and infant mortality (Green Drop Report 2009). A skills shortage, apparent lack of will to address these issues due mainly to the high costs of infrastructure repair and upgrade, and inappropriate technology choices have not helped the situation. While any and all exposure and attention to this problem is real, there is the risk that efforts to mitigate the crisis will sow seeds for a new one through inappropriate or unsustainable technology choices. Further, population growth and migration patterns, financial constraints at local government level, water shortages in many areas, the scarcity and the cost of skilled personnel and the cost of electricity, among others, all challenge this choice. Fortunately there are alternative technologies to the skills-intensive and widely accepted activated sludge (AS) process; these include algae ponding systems (Horjus *et al.* 2010).

The Water Research Commission (WRC) initiated a project in South Africa titled 'Appropriate low-cost sewage treatment using the advanced algae high rate oxidation pond' - which commenced in 1994. The outcome was full technology transfer and the design and construction of an integrated algae pond system (IAPS) demonstration and research facility at the Institute for Environmental Biotechnology, Rhodes University (EBRU) located adjacent to the Belmont Valley WWTW in Grahamstown. The rationale behind the project was to (re)-

design and optimize the IAPS technology in accordance to South African operating conditions while concurrently demonstrating the technology and providing an engineering support base for the development of IAPS process applications in the treatment of different wastewaters for use both within and outside of South Africa (Rose 2002, Rose *et al.* 1996). This pilot scale IAPS continues to operate and receives 75-100 $k\ell.d^{-1}$ municipal wastewater which, after treatment is returned to the Belmont Valley WWTW inlet works. This operating procedure is the outcome of an agreement between EBRU, the WRC, and Makana Municipality based on the following: first, EBRU is not a licensed sanitation service provider; and second, data from a series of investigations to determine nutrient removal efficiency and water quality under a wide variety of conditions revealed that IAPS was inconsistent and unable to meet the South African general authorization limit values for either discharge to a water resource or irrigation (Rose *et al.* 2007, 2002b, 2002a).

The above notwithstanding, IAPS technology is advanced in concept although basic in design and requires minimal skill to operate and maintain. A past appraisal of IAPS as ‗rural', suggested a technology with limited applicability and potential (Laxton 2010). On the contrary, IAPS can be implemented virtually anywhere depending on public needs, awareness and resources. Unlike in developed countries, land limitations are area dependent, with urban areas more restricted than peri-urban areas in South Africa, thus ‗'rural' is an inappropriate designation. Additionally with a growing population of approximately 1.6 % from 2013-2014, according to Statistics South Africa (2015), South Africa may ultimately have no options but to decentralize and possibly privatize much of its WWT. Although decentralization carries with it an element of risk, IAPS is ideally suited and is easily configured to cater to specific needs of industries and institutions, suburbs, cluster houses, holiday resorts, golfing estates and remote villages. Furthermore, although Laxton (2010) positioned IAPS as a coalesce solution to local domestic wastewater problems by depicting the technology as low cost, sustainable, low skill and robust he overlooked one conspicuous advantage which is, that once constructed the technology operates in relative perpetuity.

Reality in South Africa is an understanding that IAPS does not meet the final effluent chemical oxygen demand (COD) and total suspended solid (TSS) concentrations due in part, to suspended algae moving over the weir of the algae settling ponds (ASPs). Despite concerns by the Department of Water and Sanitation (DWS, previously the Department of Water Affairs), the WRC has lauded the technology which it believes can benefit South

Africa in many ways. In an effort to redress prevailing oversight and misconception the WRC proposed and agreed to fund a technology evaluation study at EBRU to derive information on: 1) how IAPS meets final effluent, sludge and sustainability standards; 2) IAPS design analysis; 3) IAPS operation and maintenance; and 4) sustainability benefits. The present thesis forms part of this broader WRC-funded project to specifically address the suitability of IAPS as a technology for municipal sewage treatment in South Africa and does so by re-evaluating in detail the design, operation, and performance of the Belmont Valley WWTW IAPS.

1.2 Wastewater Treatment in South Africa

Wastewater treatment technologies currently deployed in South Africa include waste stabilization ponds (WSP), oxidation ponds (OP), AS, bio-filtration (BF), biological nutrient removal (BNR), constructed wetlands (CW) and more (Oller *et al.* 2011, Tomar *et al.* 2011, Adewumi *et al.* 2010). Brief descriptions of other globally available WWT technologies are presented in Table 1. By 2009 South Africa had approximately 970 municipal WWTW treating an effluent stream of 7.6 $G\ell.d^{-1}$. At the time, the operational cost was estimated to exceed R 3.5 $billion.yr^{-1}$ (Synman 2010, DWS 2009). Distribution of these WWTW according to size shows distinct differences between the nine provinces specifically:

- Gauteng province has a relatively high number of medium (defined as WWTW with a design capacity in the range 2–10 $M\ell.d^{-1}$) and large (defined as WWTW with a design capacity in the range 10–25 $M\ell.d^{-1}$) WWTWs, with fewer micro (defined as WWTW with a design capacity <0.5 $M\ell.d^{-1}$) and small size (defined as WWTW with a design capacity in the range 0.5-2 $M\ell.d^{-1}$) plants;
- Eastern Cape, Northern Cape, Mpumalanga and Limpopo provinces mainly have micro size and small size plants;
- North West, KwaZulu Natal and Free State provinces have a wider spread of WWTW across all the plant size categories; and
- The Western Cape Province has a spread of WWTW sizes similar to the national situation.

Thus, >80 % of municipal WWTW in South Africa treat less than 10 $M\ell.d^{-1}$ and more than 50 % of all WWTW are medium sized or smaller, while the preferred technologies are WSPs and AS at 41 % and 35 % respectively. Due to a paucity of information it is not possible to determine what proportion of the estimated total effluent stream (i.e. 7.6 $G\ell.d^{-1}$) is treated by each technology. Suffice it to say, together with BF at 16 %, the range of WWT technologies commissioned by municipalities in South Africa is particularly narrow but traditionally biological (DWS 2009). The 2012 Green Drop Report evaluated 156 municipalities, where 821 WWT systems were assessed and approximately 73 % of WWTW evaluated ranged in size from micro to medium, as displayed in Figure 1. Together these WWTW had a total design capacity of 6.6 $G\ell.d^{-1}$ with an estimated received flow of 5.3 $G\ell.d^{-1}$ and a spare capacity of 1.3 $G\ell.d^{-1}$, essentially a meagre 20 % design capacity to meet future demand (DWS 2012). The 2012 Green Drop Report further noted that in terms of national distribution, 67 % (i.e. 4.4 $G\ell.d^{-1}$) of the design capacity was confined to urban areas.

Capital investment into new water and sanitation infrastructure for the entire value chain including the refurbishment of existing infrastructure is projected to require an estimated R 670 billion over the next 10 years or R 67 $billion.yr^{-1}$ (DWS 2012). By May 2013, only R 30 $billion.yr^{-1}$ (45 %) was available for the development and rehabilitation of water infrastructure from government sources. Fortunately, that budget has increased significantly in recent years owing to improved government treasury allocations for infrastructure development (DWS 2012). These total allocations were R 10.2 billion for 2013/14, R 12.4 billion for 2014/15, and R 15.5 billion for 2015/16 (DWS 2012).

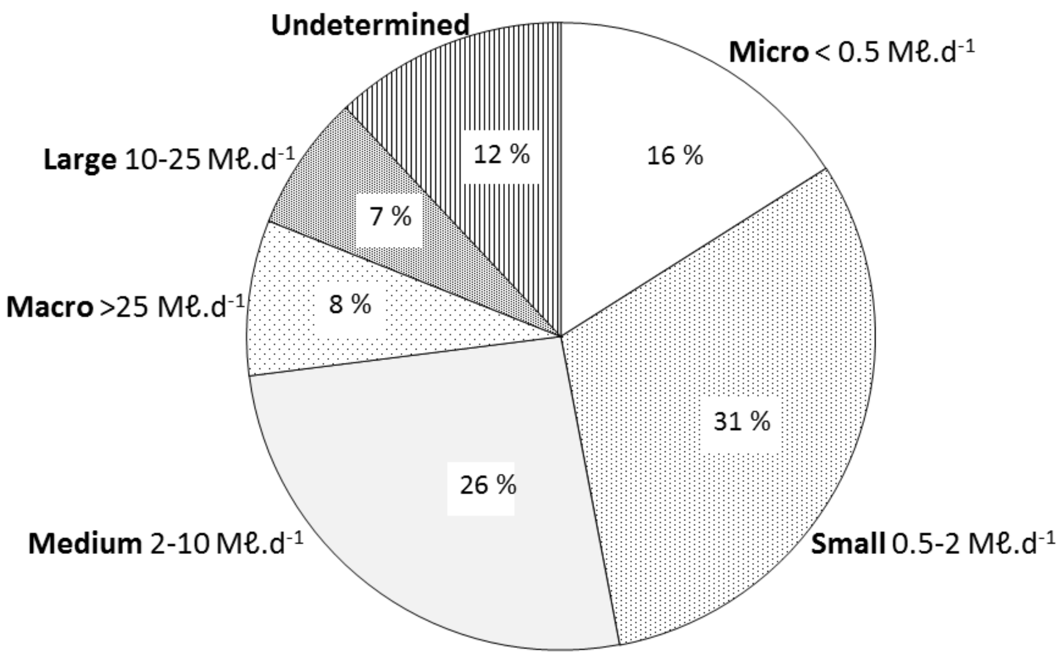


Figure 1 Size distribution of municipal wastewater treatment works in South Africa, adapted from DWS (2012).

Table 1 A brief description of general wastewater treatment techniques

Technology	Brief description	References
Activated sludge system	Air is introduced into primary treated wastewater combined with microorganisms that decrease the organic content of the water stream	(Tammaro *et al.* 2014)
Advanced oxidation ponds	Utilization of ozonation, hydrogen peroxide or UV light to remediate wastewater containing biologically toxic and/or recalcitrant compounds such as aromatics, pesticides, petroleum components and volatile organic compounds.	(Oller *et al.* 2011, Cha *et al.* 1997)
Aerobic granulation	Microbial aggregates that do not congeal under reduced hydrodynamic shear and are able to settle faster than activated sludge flocs are used to treat wastewater	(Wang *et al.* 2007, Zhang *et al.* 2011, Li *et al.* 2011)
Aerated lagoons	Biological oxidation of wastewater through motor driven or submerged diffusers mechanically pumping air into the system.	(Wu and Chen 2011)
Aerobic granular reactor	Use of a microbial biomass that develops on fast settling granules to efficiently remove chemical oxygen demand, nitrogen and phosphorous in a discontinuous system	(Bassin *et al.* 2012, Quan *et al.* 2012)
Anaerobic filter	Use of anaerobic microorganisms attached to filter media under oxygen deficient conditions to remediate wastewater	(Hamdi *et al.* 1991)
American Petroleum Institute (API) oil water separators	Utilization of the specific gravity difference between oil and wastewater to separate copious amounts of oil from wastewater	(Punnaruttanakun *et al.* 2003, Deng *et al.* 2002)
Anaerobic lagoon	Degradation of wastewater generated by animals in an earthen basin containing a water body undergoing remediation by microorganisms that function in oxygen deficient environments	(Wu and Chen 2011, Safley *et al.* 1992)
Belt filter	A slurry is passed through filtering cloths which are in turn passed over belts in a system of rollers generating a filtrate and a cake	(Ebeling *et al.* 2006, Day and Giles 2002)
Biofiltration	Living material captures and biologically degrades pollutants, examples include constructed wetlands, slow sand filtration systems and Riparian zones/forests	(Tomar and Suthar 2011)
Carbon filtering	Activated carbon granules provide a large surface area for the chemical absorption removal of contaminants via filtration	(Tammaro *et al.* 2014)

Capacitive deionization (CDI)	An electrical potential is applied over 2 porous carbon electrodes for the desalination of brackish water. It is considered more energy efficient than reverse osmosis	(Cai *et al.* 2014, Jande and Kim 2014)
Cesspit/Cesspool	A deep cylindrical chamber dug into the earth for the remediation and storage of human excrement	(Public Health 1897)
Clarifier	A sedimentation tank utilized to segregate solids from liquids	(Ravazzini *et al.* 2005, Chatellier and Audic 2000)
Coarse bubble diffusers	Aerate and mix wastewater, they have been found to be more efficient at mixing wastewater than transferring oxygen into the water column	(Ashley *et al.* 1992)
Composting toilet	A dry toilet using minimal quantities of water where human excrement is mixed with material such as sawdust which is able to absorb water, mitigating odours and supporting aerobic microbial degradation processes	(Anand and Apul 2014, Devkota *et al.* 2013, Hill and Baldwin 2012)
Constructed wetland	An artificial marsh built to act as a biofilter removing sediments and contaminants such as heavy metals from wastewater	(Travis *et al.* 2012, Kaseva 2004, Kivaisi 2001)
Dark fermentation	Anaerobic biohydrogen production through the degradation of contaminants found in industrial and domestic wastewater	(Kargi *et al.* 2012, Ozmihci *et al.* 2011)
Dissolved air flotation	Pressurized air is introduced to wastewater. When the pressure is alleviated, the air rises while adhering to solids and this effectively removes wastes from suspension	(Edzwald 1995)
Distillation	The selective separation of wastewater by vaporisation and condensation	(Doušková *et al.* 2010)
Desalination	Removal of salts and minerals (filtration, chemical precipitation etc.) from saline water to generate water suitable for irrigation or following the relevant further treatment drinking water	(Bdour *et al.* 2009, Urkiaga *et al.* 2006)
Electrocoagulation	Application of an electrical pulse in order to change the surface charge of particles in wastewater to causing aggregation and consequent sedimentation thereby remediating the water, this is a form of Electrolysis	(Feng *et al.* 2007, Holt *et al.* 2005)
Electrolysis	Introduction of an electric current to wastewater that results in its separation into its constituents, anions move to the positive anode and cations move to the cathode, these subsequently settle out of suspension and generate a clean effluent	(Cheng *et al.* 2007)

Fine bubble diffusers	Copious amounts of slow rising bubbles provide adequate oxygen transfer for sewage degradation by proliferating aerobic microorganisms	(Colt *et al.* 2010, Ovezea 2009)
Forward osmosis	Application of an osmotic potential through a semi permeable membrane that results in the separation of solutes from wastewater, rendering it clean	(Yangali-Quintanilla *et al.* 2011, Wang *et al.* 2010)
Induced gas flotation	A clarifier that removing suspended matter such as oils and solids through bubble injection	(Painmanakul *et al.* 2010, Moosai and Dawe 2003)
Living machines	Fixed film ecology in a natural tidal wetland which is able to generate reuse quality water	(Lansing and Martin 2006, Brix 1999)
Maceration	Shredding of sewage generating a slurry for easier movement in a wastewater treatment facility	(World Pumps 2011)
Nanotechnology	Exploitation of the unique interactions towards recalcitrant contaminants, currently under extensive research and development	(Qu *et al.* 2013)
Reed beds	Phytoremediation of grey water	(O'Luanaigh *et al.* 2010, Zhao *et al.* 2009, Sun *et al.* 1999)
Reverse Osmosis	Employment of pressure on a selective membrane that allows the solvent to pass through freely but retains solutes and other debris on the pressurized side of the membrane	(Yangali-Quintanilla *et al.* 2011)
Rotating biological contactor	Closely spaced parallel discs with a biofilm layer are introduced to wastewater the microorganisms in the biofilm take up the nutrients while degrading any organic compounds in the wastewater	(Teixeira *et al.* 2001, Buchanan *et al.* 1994)
Sand filters	Utilized for tertiary treatment and generally requiring the injection of flocculating chemicals, they are particularly effective for the removal of pathogens and the generation of high quality water	(Katukiza *et al.* 2012)
Septic tank	An anaerobic digester that may be directly linked to a household. It results in the degradation of organic compounds and the remediation of wastewater which then seeps into the environment	(Withers *et al.* 2011, Moussavi *et al.* 2010)
Sequencing batch reactor (Aerobic or	Bacteria are used to remediate wastewater, oxygen is pumped into the first reactor aiding in the complete aerobic breakdown of the components of the wastewater. The effluent generated is then channelled into a second reactor where any suspended solids are allowed to settle out	(Bassin *et al.* 2012, Liu *et al.* 2005, Rodrigues *et al.* 2003)

anaerobic)	of suspension	
Submerged aerated filters	Employment of an upflow fixed biofilm reactor with a coarse medium that does not require backwashing	(Khoshfetrat *et al.* 2011)
Ultrafiltration	Use of hydrostatic pressure through a semi permeable membrane that results in the separation of solutes from the solvent. It is one of a variety of membrane technologies which vary depending on size of compound being filtered i.e. nanofiltration, microfiltration and gas separation	(Cha *et al.*1997, Hamza *et al.* 1997, Cherkasov *et al.* 1995)
Upflow anaerobic digester	Employment of gravity within an anaerobic digester to settle organic solids out of suspension for degradation and retention in the reactor, while the clean effluent is channelled out of the reactor	(Parawira *et al.* 2005, Hansen *et al.* 1992)
Wet oxidation/ Zimpro Wet Air Oxidation	Superheated air is used to oxidize components in wastewater for degradation by conventional treatment systems	(Sun *et al.* 2008, Lei *et al.* 1998, Cha *et al.* 1997)

DWS (2009) reported that 16 % of WWTW were hydraulically overloaded, 22 % organically and nutrient overloaded, and <35 % were hindered by elevated loads and high influent flows (Synman and van Niekerk 2011, DWS 2009). Furthermore, >50 % of WWTW were reported to have experienced difficulties with flow balancing, were incapable of remediating wastewater to a secondary quality due to excessive sludge build up in maturation ponds (MP), while chlorination facilities were routinely malfunctioning (Synman 2010, DWS 2009). Synman and van Niekerk (2011) reported that 30 % of plants required urgent intervention to prevent the outbreak of waterborne diseases. Moreover, Synman and van Niekerk (2011) stated that if the regulations were to be strictly enforced; only 4 % of the WWTW surveyed would comply with discharge standards. One possible solution to this would be to utilize IAPS and/or components of IAPS technology such as high rate algae oxidation ponds (HRAOPs) to increase the capacity of existing but overloaded and under- performing WWTW to reduce sludge build up by intercepting and treating culprit flows. Indeed, preliminary studies showed that incorporation of independent-HRAOPs (I-HRAOPs) could potentially be used in a remedial capacity to mitigate overloading and under performance of traditional WWTW (Rose *et al*. 2007).

1.3 Integrated Algae Pond Systems

An IAPS is essentially an adaptation of the Advanced Integrated Wastewater Pond System (AIWPS) and like the AIWPS is a passive process that uses gravity, solar energy and biological activity to treat wastewater (Downing *et al.* 2002, Oswald 1995). AIWPS exploits the natural functionality of anaerobic, facultative, and aerobic microorganisms resident in distinct locales within the pond system (Craggs *et al.* 1996, Oswald 1995) which comprises an advanced facultative pond (AFP) containing an in-pond digester (I-PD), HRAOPs, ASPs and an MP series. Tertiary treated water may be reclaimed following filtration and ultra violet (UV) light sterilization as depicted in Figure 2. Table 2 presents data for water quality of the final effluent from AIWPS plants deployed at Richmond, St Helena, Hollister, and Delhi, California, U.S.A. treating municipal sewage. Aside from the plant at Hollister, the other three deployments included a tertiary treatment process positioned after the ASP, despite this, all the systems removed >95 % influent biochemical oxygen demand (BOD). Additionally the TSS content in the wastewater entering the Richmond and Delhi plants was reduced by >90 % while total *ortho*-phosphate and ammonia-N in water from the St Helena and Richmond installations was <10 mg.ℓ^{-1} and <1 mg.ℓ^{-1} respectively. The system at St Helena reduced influent COD by 93 %, rendering an outflow COD concentration of 32 mg.ℓ$^{-1}$ (Ertas and Ponce 2012). These consolidated data derived for AIWPS operating in California appear to support the view that this technology has the potential, when implemented in its entirety, to generate a treated water of a tertiary quality. In contrast to these North American examples, the pilot IAPS at the Belmont Valley WWTW in Grahamstown was implemented without a dedicated tertiary treatment component namely an MP and in this respect resembles most closely the AIWPS Secondary Process depicted in Figure 2. Consequently, and at best, the final effluent generated by the Belmont Valley IAPS can only be described as a 'secondary treated' water which may justify cautious conclusions drawn following extensive analyses of system performance and tertiary treatment operation (Rose *et al.* 2007).

Table 2 Nutrient removal and performance data from commercial scale AIWPS in California, U.S.A. (Ertas and Ponce 2012).

Parameter	Richmond[A]	St Helena[B]	Hollister[C]	Delhi[D]
		mg.ℓ$^{-1}$ (% removal)		
BOD	2 (99)	7 (97)	7 (96)	4 (98)
COD	n.d.	32 (93)	n.d.	n.d.
TSS	2 (99)	n.d.	n.d.	10 (92)
NH_3-N	0.4 (99)	n.d.	n.d.	n.d.
NO_3^{2-}-N	n.d.	n.d.	n.d.	n.d.
Total P	0.5 (92)	5 (90)	n.d.	n.d.
Soluble P	0.2 (96)	n.d.	n.d.	n.d.

[A]ASP followed by Dissolved Air Flotation and sand filter
[B]ASP followed by MP
[C]ASP only
[D]ASP followed by MP and Percolation Beds
n.d.=not determined

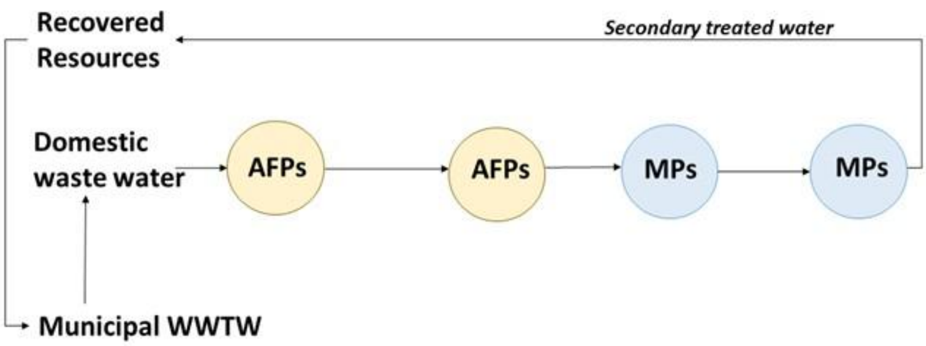
It is well established that the majority of biological remediation of wastewater by AIWPS/IAPS takes place in the AFP containing the anaerobic I-PD (Green *et al.* 1996, Oswald *et al.* 1991). The resultant effluent is then polished in 2 HRAOPs which are paddlewheel mixed raceways usually operated in series (Boshoff *et al.* 2004, Van Hille *et al.* 1999). Approximately 80 % of the algae biomass in HRAOPs following floc formation is removed by gravity in ASPs (Van Hille *et al.* 1999). The effluent is alkaline ranging between pH 9.5 to 10 which eliminates pathogens including *E. coli* (Chan *et al.* 2009, Grönlund *et al.* 2004), while solar-derived UV light also contributes to significant disinfection within HRAOPs (Green *et al.* 1996). The pilot scale IAPS, in line with other global deployments (Table 3), was intended to demonstrate and evaluate the performance of the system for use as a 'green' technology to address issues such as climate change and sustainable development

(Rose 2002, Rose *et al.* 2002b). Costs associated with construction and operation (including maintenance) of the IAPS were, at the time, viewed as highly competitive (Rose *et al.* 2002b), job creation was evidently possible (Harun *et al.* 2010, Rose *et al.* 2002a), while improved access to clean water was and still is understood to stimulate social and economic development (Adewumi *et al.* 2010, Rose 2002, Oswald 1995). An extended study evaluating the operation and performance of this IAPS as a full WWT system for South Africa was published in 2007 and revealed the following;

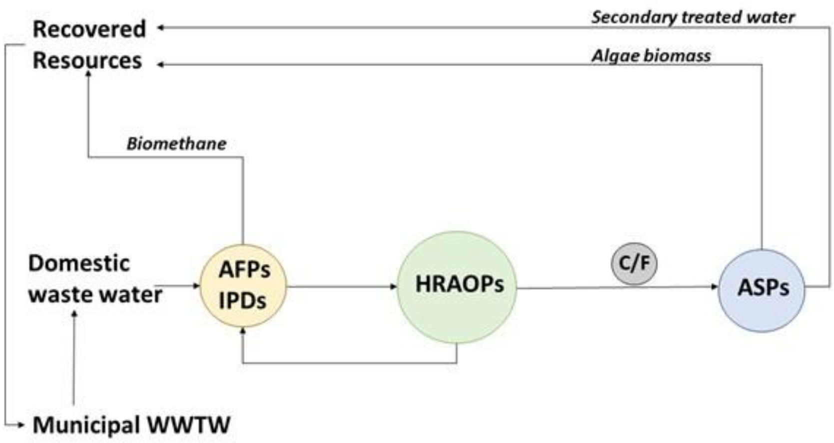
- The system did not achieve the 75 mg.ℓ^{-1} discharge standard for COD,
- Although a reduction in *ortho*-phosphate was observed, it was not within the 10 mg.ℓ^{-1} required for discharge,
- Residual ammonia-N levels exceeded the 6 mg.ℓ^{-1} irrigation standard,
- Nitrate-N removal was at best erratic and at times, nitrate-N concentrations increased (Rose *et al.* 2007).

These findings offer an explanation for the reluctance to implement IAPS as a WWT technology in South Africa, possibly due to the apparent inability of the system to yield a treated water of a quality compliant for either irrigation or discharge to the environment. The general authorizations (Republic of South Africa *National Water Act* 2013) state that in order for an effluent to be discharged to the environment, it must comply with wastewater general limit values and have a COD value <75 mg.ℓ^{-1} (after removal of algae). Metadata from the operation of the Belmont Valley WWTP IAPS for the period 1999-2006 clearly shows a treated water with COD consistently >75 mg.ℓ^{-1} (Rose *et al.* 2007, Wells 2005, Rose *et al.* 2002a).

Nevertheless, commercial AIWPS operating in California demonstrate the potential of the technology to treat domestic wastewater, in particular after implemention of an appropriate tertiary treatment process (Ertas and Ponce 2012). First generation AIWPS were designed to remediate domestic wastewater without the recovery of methane and biomass (Green *et al.* 1995). By comparison, second and third generation AIWPS were designed to be more self-sustaining, to operate in relative perpetuity and to allow for the recovery of methane, reclamation and reuse of water, and in some cases for harvest of algae biomass (Green *et al.* 1996). As shown in Figure 2, the original AIWPS designs always included a polishing or tertiary treatment step comprising of either an MP or similar which would allow the final treated water to meet the specifications for discharge as required except for the presence of total coliforms, which requires additional disinfection (e.g. chlorination, ozonation, UV light disinfection, etc.). In fact, a report on the operation of hectare-scale HRAOPs for enhanced WWT strongly advocated that additional treatment of the algae harvester effluent (i.e. outflow from ASP) requires polishing to meet specific discharge standards (Craggs *et al.* 2012). Craggs *et al.* (2012) recommended the inclusion of one or a combination of MPs and UV light treatments by storage prior to discharge or, rock filtration of the MP effluent, or direct UV light treatment if land was insufficient. Depending on available funds, membrane filtration could also be used to achieve a high quality final effluent for reuse. However without a final polishing step, and as demonstrated in other studies (Craggs *et al.* 2012), the COD of treated water remains elevated resulting in the possibility that if discharged, water from an IAPS will be detrimental to any receiving water bodies (Park and Craggs 2011).



AIWPS Type II Process Schematic



AIWPS Advanced Secondary Process Schematic

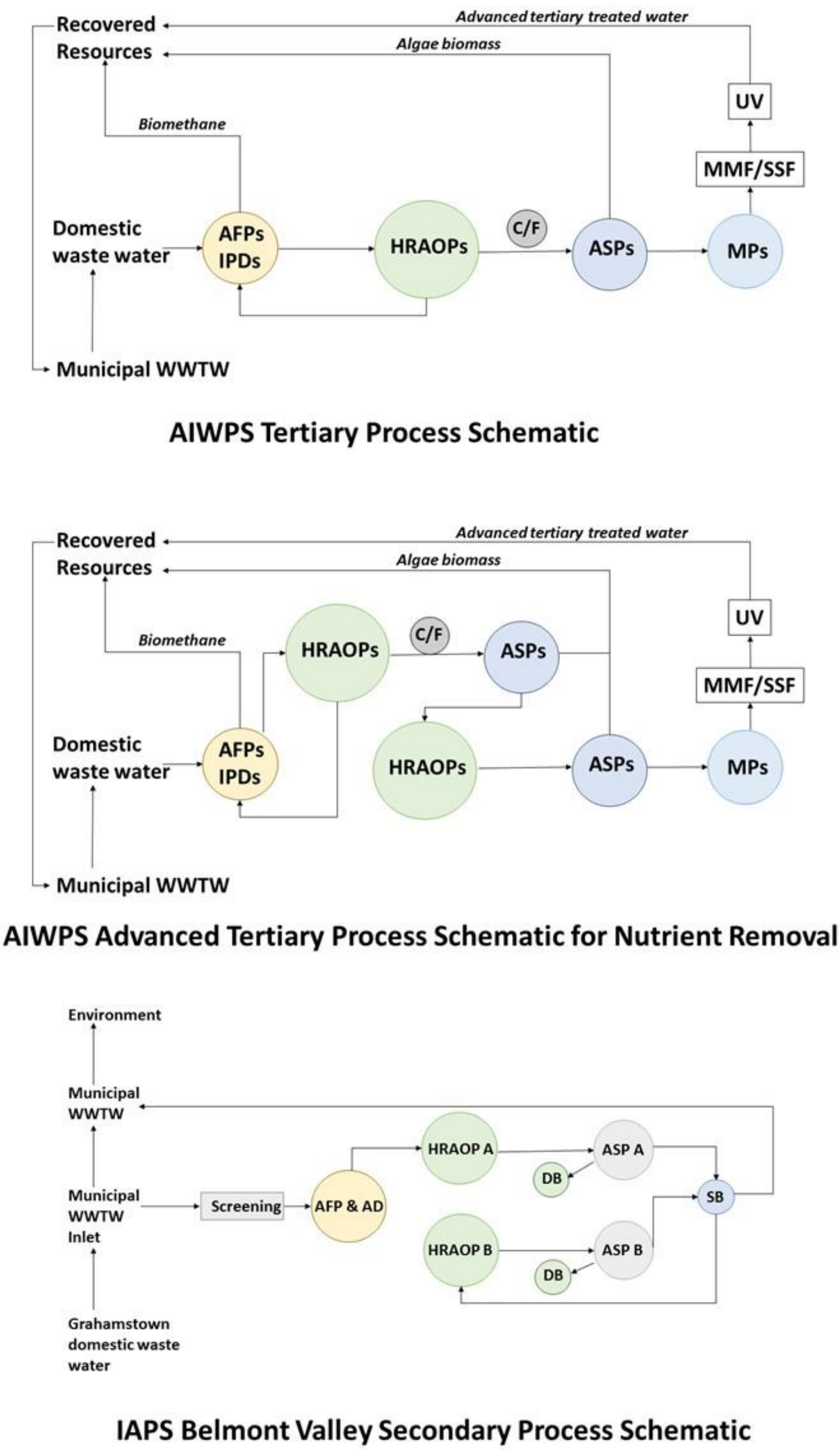


Figure 2 Schematics illustrating the process flow for various AIWPS designs based on technology developed by Oswald to recover nutrients, energy and water from influent wastewater. Nutrients - harvested algae from the algae settling ponds can be dried and sold as animal feed or fertilizer; Energy - near quality pipeline biogas can be captured, flared, or utilized to offset energy consumption from the WWTW. Water - the WWTW final effluent can be reclaimed for beneficial reuse. AFP=Advanced Facultative Pond; IPD=In-Pond Digester; HRP=High Rate Pond; C/F=Coagulation/Flocculation; ASP=Algae Settling Pond; MP=Maturation Pond; PB= Percolation Bed; MMF=Multimedia Filtration; UV= Ultraviolet Light Disinfection (Internet 1, 2013, Green *et al.* 1996).

Table 3 Global IAPS applications for domestic wastewater remediation and environmental discharge.

Country	Climate	Comissioned/Evaluated	References
Brazil	Tropical-Temperate	1983	(Kawai *et al.* 1984)
China	Monsoon	2002	(Chen *et al.* 2003)
Egypt	Desert	1990	(El-Gohary *et al.* 1991)
France	Temperate	1997	(Bahlaoui *et al.* 1997)
Germany	Temperate	1986	(Grobbelaar *et al.* 1988)
India	Tropical	1986	(Mahadevaswamy and Venkatamaran 1986)
Kuwait	Desert	1990	(Al-Shayji *et al.* 1994)
Jordan	Desert	1985	(Al-Salem 1987)
Morocco	Mediterranean	2003	(El Hamouri *et al.* 2003)
The Netherlands	Moderate maritime	1988	(Kroon *et al.* 1989)
New Zealand	Temperate	2007	(Craggs *et al.* 2004)
New Zealand	Temperate	2010	(Broekhuizen *et al.* 2012)
Spain	Mediterranean	2006	(Garcia *et al.* 2006)
Sweden	Cold	2003	(Gröndlund *et al.* 2004)
United States of America	Mediterranean	1959	(Downing *et al.* 2002)

1.4 IAPS Components and Function

Advanced Integrated Wastewater Pond Systems and their derivation IAPS are aesthetically pleasing water bodies composed of four pond types each carrying out distinct functions for the remediation of domestic wastewater. Examples of AIWPS WWTWs in California, U.S.A., specifically in the cities of Delhi, Hilmar, and St Helena are shown in Figure 3. Domestic wastewater is initially directed into an AFP. The AFP is a combined anaerobic I-PD and facultative pond. Wastewater is first channeled into the I-PD and then enters the facultative pond. The facultative pond operates as a buffer system where algae in the aerobic layer oxygenate the water while assimilating nutrients. Algae aggregates and debris settling from the surface of the pond are anaerobically degraded at the base of the system. Water from this pond decants into HRAOPs operating in series. These ponds are fully oxygenated through paddlewheel mixing which enhances algae photosynthetic activity. Water from the HRAOPs passes via ASPs to MPs where solar disinfection and pathogen removal occur (Ertas and Ponce 2012, Downing *et al.* 2002, Green *et al.* 1996, Craggs *et al.* 1996b, Nurdogan *et al.* 1996, Oswald *et al.* 1957).

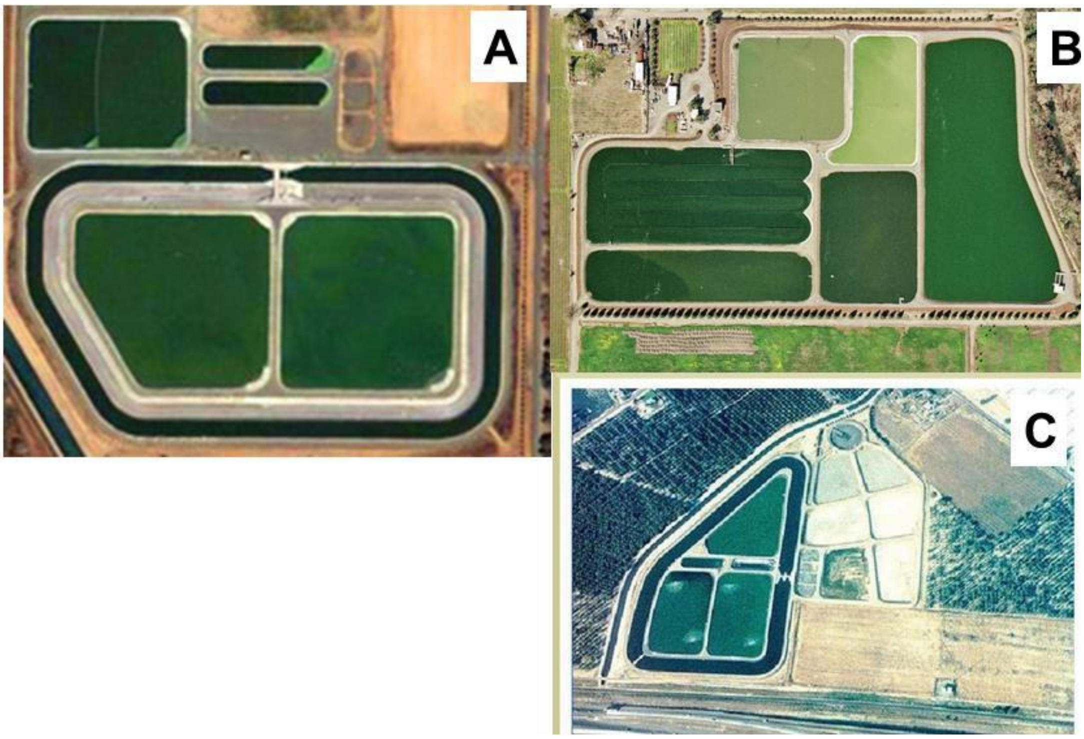


Figure 3 Aerial photographs of the most established examples of the AIWPS technology A: Hilmar, California has a capacity of 2.43 Mℓ.d^{-1}, B: St Helena, California (capacity undisclosed) and C: Delhi, California has a capacity of 3.18 Mℓ.d^{-1} (Ertas and Ponce 2012).

Advanced Facultative Pond (In-Pond Digester and Primary Facultative Pond) – The AFP receives raw wastewater directly into the I-PD while secondary facultative ponds treat settled wastewater (Mashauri and Kayombo 2002, Nelson *et al*. 2004). Very little information is available on the biology and function of the IAPS I-PD. Preliminary data derived from the Belmont Valley WWTW IAPS has indicated the potential to produce a biogas stream comprising more than 80 % methane (Tijjani 2011). Since a value of 70 % methane is traditionally regarded as good (Dugan *et al.* 1972), all indications are that an above average biogas stream can be routinely obtained from this system. The only reliable data on methane production by the I-PD of an IAPS was recorded from a plant in Richmond, California (Green *et al*. 1995). As indicated in Table 4, this equates to a theoretical maximum of approximately 9.6 kℓ.biogas.d^{-1} for a 100-household system (i.e. ~500 PE) similar to that currently in operation at EBRU.

Table 4 Methane production rates from IAPS in-pond digesters in advanced facultative ponds.

Production rate	Richmond, CA	Theoretical Maximum
kℓCH_4.$kg^{-1}$$BOD_5$	0.15	0.53
kℓCH_4.$kg^{-1}$$BOD_{ULT}$	0.24	0.85
kℓCH_4.d^{-1} (400 PE)	7.68	27.33
kℓCH_4.d^{-1} (500 PE)	9.60	34.03
kℓCH_4.d^{-1} (600 PE)	11.52	40.84
If biogas is 86 % CH_4, then total kℓbiogas.d^{-1} (500 PE)	11.16	39.57

Biogas (methane equivalent) production at or near maximum and with calorific value of 4.9×10^7 J.m^3 could theoretically give a total daily energy stream of 1.7×10^9 J.m^3 from a 500 PE system similar to the EBRU pilot plant. Unfortunately, these systems do not operate near the theoretical maximum and recent kinetic studies at EBRU have revealed a biogas production rate with methane and energy content as indicated in Table 5 (Tijjani 2011).

Table 5 Mean biogas and methane yield from the EBRU (500 PE) IAPS in-pond anaerobic digester.

Biogas (kℓ.d^{-1})	CH_4 (kℓ.d^{-1})	Energy Yield J.yr^{-1}
7.54 ± 2.36	5.12 ± 1.60	9.4×10^{10}

Detailed studies on the co-digestion of algae biomass and domestic wastewater showed that whole, untreated algae biomass increased methane output by 200 % whereas further fracturing of the biomass either by sonication alone or freeze-thaw followed by sonication increased methane yields by 500 % and 650 % respectively (Tijjani 2011). This indicates that were algae biomass to be co-digested on site the combined methane output from the IAPS I-

PD and the digestion of algae biomass can be increased substantially to yield an energy stream far in excess of an already impressive 9.4×10^{10} $J.yr^{-1}$.

Facultative ponds are typically 0.9-2.4 m in depth, with a retention time up to 50 d (Nelson *et al*. 2004). These ponds consist of an aerobic, anoxic and anaerobic layer (Charlton 1997). Odour control and removal of nutrients occur in this pond through the mutualistic relationship that exists between aerobic and anaerobic microorganisms (Muga and Mihelcic 2008). Algae found in the aerobic layer of the pond release oxygen into the system, aerobic microorganisms utilize this oxygen to break down organic compounds and in the process, generate CO_2, which is the substrate for algae photosynthesis (Sutherland *et al*. 2015, Mara 2005, Cole 1982) and the biochemistry involved is shown schematically as follows:

Photosynthesis:	$CO_2 + H_2O \leftrightarrow CH_2O + O_2$
Aerobic oxidation:	$CH_2O + O_2 \leftrightarrow CO_2 + H_2O$
Organic acid formation:	$2CH_2O \leftrightarrow CH_3COOH$
Methanogenesis:	$CH_3COOH \leftrightarrow CH_4 + CO_2$
	$CO_2 + 4H_2 \leftrightarrow CH_4 + 2H_2$
Heterotrophic nitrification:	Fixed N $\leftrightarrow NO_3^-$
De-nitrification:	$2NO_3^- + 3CH_2O \leftrightarrow N_2 + 3CO_2 + 3H_2O$

Oxygen does not penetrate the entire depth of the system (Nelson *et al.* 2004). Turbidity due to suspended matter ensures light also does not penetrate to the base of the system (Kayombo *et al.* 2002), while the activity of aerobic microorganisms ensures that oxygen is depleted in the top layer of the pond (Charlton 1997). Thus, there is temperature, oxygen, light, CO_2, as well as pH stratification within facultative ponds (Nelson *et al.* 2004, Kayombo *et al.* 2000, Charlton 1997). A thin sludge layer develops at the base of this pond where anaerobic conditions dominate; nutrients produced by anaerobic digestion dissolve in the water, are re-suspended and utilized by microorganisms in the aerobic layer of the pond (Green *et al.* 1995).

High Rate Algae Oxidation Ponds – There are 3 different geometries of algae raceway ponds namely, circular, sloping and the traditional oblong shape. Circular ponds are agitated using a rotating arm, while gravity and a pump are used in the sloping variety and the oblong shape is mixed using a paddlewheel (Hadiyanto *et al*. 2013). High rate algae oxidation ponds of the oblong variety were initially developed for combined WWT and nutrient recovery (Golueke *et al*. 1959, Golueke *et al.* 1957, Oswald *et al.* 1957). Later, bespoke HRAOPs were developed and utilised for the mass cultivation of algae (Sutherland *et al*. 2015, Sutherland *et* *al.* 2014a, Park and Craggs 2011). In particular, commercial production systems for *Athrospira platensis* (Rose and Dunn 2013, Dunn and Rose 2013, Borowitzka 1999), *Dunaliella salina* (Moheimani *et al.* 2006, Borowitzka 1999), and *Haematoccocus pluvialis* (Harun *et al.* 2010, Borowitzka 1999) were established, many of these continue to successfully produce algae biomass. Jiminez-Perez *et al.* (2004) found that algae tend to dominate in wastewater from whence they originate due to progressive acclimation. Thus, desired species control, when cultured in wastewater under continuous flow, in HRAOPs may prove difficult when invasive strains outcompete resident algae.

The environment in the raceway tends to select for robust algae species like *Chlorella* sp. and *Scenedesmus* sp. that are able to thrive under diurnally varying conditions for sunlight, temperature, pH and dissolved oxygen (DO) content (Zhou *et al.* 2014). As colonial algae like *Scenedesmus* sp. are prone to settling, turbulent mixing generated by paddlewheel propulsion needs to overtake algae sinking losses (Sutherland *et al.* 2015, Sutherland *et al.* 2014b, Hadiyanto *et al.* 2013). Settled algae discolour and are not photosynthetically active therefore turbulent mixing ensures retained productivity, reduced algae settling, minimal nutrient stratification and restricted physicochemical gradients (Sutherland *et al.* 2014b, Hadiyanto *et al.* 2013). Hydrodynamic stress, depending on the algae strain, may however negatively impact sheer sensitive cultures, though Hadiyanto *et al.* (2013) report decreased photoinhibition and improved gas exchange when flow velocity increases. Hadiyanto *et al.* (2013) suggest the use of flow deflectors which increase paddlewheel mixing power while impeding the settling of cells. Sutherland *et al.* (2014a) demonstrated that by increasing the paddlewheel mixing speed, biomass yields increased concomitantly with nutrient removal efficiencies. Rogers *et al.* (2014) upon evaluation of an 8 000 m^2 paddlewheel mixed system for the production of an algae biomass using wastewater found that sourcing the water, purchasing CO_2 and other nutrient could prove cost prohibitive, depending on the location of the facility (Rogers *et al*. 2014).

Choi and Lee (2015) state that when the N:P ratio is 31:40, *Scenedesmus* sp. yield from wastewater ranges between 400-780 $mg.\ell^{-1}.d^{-1}$ when supplemented with CO_2. Furthermore, Xu *et al*. (2013) found high algae growth rates and optimal light exposure were vital for algae activity in these ponds. Matamoros *et al.* (2014) also found that water temperature, exposure to light, and algae settling, impacted the ability of the algae to remediate emerging micro-contaminants in waste water, however >90 % removal efficiencies for substances like caffeine, ibuprofen and acetaminophen were maintained throughout that particular study. Previously however, Waller *et al.* (2012) raised concerns regarding the susceptibility of raceways to diurnal and seasonal temperature variations. Biomass productivity is detrimentally impacted by a low water temperature, low nutrient concentration, reduced exposure to light, algae settling and an elevated oxygen concentration (Sutherland *et al.* 2014a, Waller *et al.* 2012). However, this nutrient rich algae biomass, generated in HRAOPs from wastewater remediation (Craggs *et al.* 2012) is more cost efficient to produce when due consideration is given to skills, energy and cost intensive photobiorectors deployed for the production of high value algae products (Yadala and Cremaschi 2014).

Algae present in HRAOPs can either be uni and/or multicellular, prokaryotic or eukaryotic (Incropera and Thomas 1978) and in wastewater remediation, utilize nutrients from the surrounding system directly cleaning the wastewater (de-Bashan *et al.* 2004, de la Noue and de Pauw 1988). Symbioses between bacteria and algae bring about an ecological pattern different from pure culture behaviour through the exploitation of their normal functionality (Craggs *et al.* 2004), resulting in the remediation of wastewater (Kayombo *et al.* 2002). This cycle is specific to aerobic ponds (Oswald *et al.* 1955). Oswald *et al.* (1957) reported that HRAOPs yield biomass of 0.2 $kg.m^{-2}.month^{-1}$ and 1.2 $kg.m^{-2}.month^{-1}$ which amounts to approximately 7.4 $kg.m^{-2}.yr^{-1}$. Rose and Dunn (2013) were able to measure 1.6 $kg.m^{-2}.yr^{-1}$ ash free dry weight from *Arthrospira* sp. However, yield varies due to fluctuations in temperature and light intensity (Tadesse *et al.* 2004, Oswald *et al.* 1955). As illustrated in Figure 4, algae photosynthesis supplies aerobic bacteria in the ponds with oxygen while bacterial oxidation produces CO_2, nitrate-N and *ortho*-phosphate which are assimilated by algae into carbohydrates, lipids, proteins and various other organic compounds (Sutherland *et al*. 2015, Craggs *et al.* 2012, Green *et al.* 1996, Oswald *et al*. 1957).

Various pre-occurring microbial processes are utilized in both biological and conventional WWT systems to remediate polluted streams (Hesnawi *et al.* 2014, Marlow *et al.* 2013). Ammonification, a biologically catalysed process arises from the breakdown of proteins and amino acids releasing ammonia-N (Katipoglu-Yazan *et al.* 2012). Ammonia-N is then stored in the environment as more stable ammonium. Ammonium is vital for all plant and animal nutrition (Chen *et al.* 2009, Hotta and Funamizu 2008). Ammonification is responsible for a majority of organic N contributions in wastewater (Hotta and Funamizu 2008). Nitrification carried out by a small group of specialized autotrophic microorganisms (Katipoglu-Yazan *et* *al.* 2012), in wastewater sequentially converts ammonia-N and/or ammonium to nitrite-N. Further oxidation of nitrite-N generates nitrate-N (Galanopoulos *et al.* 2013, Katipoglu-Yazan *et al.* 2012). Nitrate-N generated by this process contributes to eutrophication if retained in water discharged to the environment (Hotta and Funamizu 2008). Denitrification, carried out by microorganisms like *Pseudomonas* sp., *Paracoccus denitrificans* and *Thiobacillus denitrificans* reduces the oxidized forms of N namely nitrate-N, nitrite-N, nitric oxide and nitrous oxide to N_2 gas (Lu *et al.* 2014, Rodriguez-Caballero *et al*. 2014), unfortunately depending on the substrate denitrification also generates nitrous oxide which is implicated in global warming (Quan *et al.* 2012). At present denitrification is harnessed in WWT to prevent eutrophication, Quan *et al.* (2012) report that while many microorganisms are capable of carrying out heterotrophic facultative aerobic denitrification the process offers no conspicuous energetic advantage.

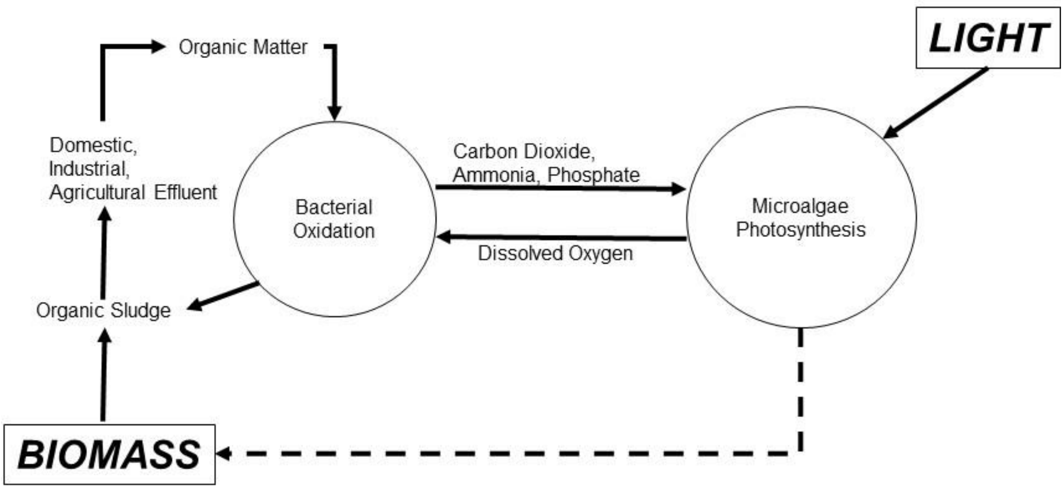


Figure 4 Interaction between bacterial oxidation and algae photosynthesis in high rate oxidation ponds (Sutherland *et al.* 2015, Munoz and Guieysse 2006, Oswald *et al.* 1955).

Algae cell access to light and algae driven nutrient removal can be indirectly be measured as a consequence of an increase in algae biomass (Sutherland *et al.* 2015, Craggs *et al*. 2004, Oswald *et al.* 1955). In HRAOPs algae consortia are more efficient at nutrient removal than monoculture (Zhou *et al.* 2014). To ensure best possible photosynthetic rates, aerobic ponds are traditionally no more than 0.30-0.45 m deep, are continuously mixed and driven by a paddlewheel (Rogers *et al*. 2014, Oswald *et al*. 1953b, Oswald *et al*. 1953a). However, Sutherland *et al*. (2014b) recently demonstrated that a depth of 0.50 m was also sufficient for algae productivity. As a consequence of enhanced productivity, aerobic raceway ponds not only have high BOD removal capacity but also produce algae in an economically viable manner (Craggs *et al.* 2012, Green *et al.* 1996).

Elevated concentrations of N and P have been implicated in eutrophication (de la Noue and de Pauw 1988). Eutrophication negatively impacts an area when an algae bloom occurs due to an influx of nutrients caused by anthropogenic activities, leading to anaerobic conditions in the affected ecosystem (Withers *et al*. 2011, Benemann *et al*. 2003). Controlled eutrophication is allowed to occur in HRAOPs (Oswald *et al.* 1957). Algae growth and productivity is stimulated by the products of bacterial respiration as shown in Figure 4 and by the assimilation of auxiliary nutrients including N and P released by the aerobic breakdown of organic compounds. It has been established that 80 % of the DO in WWT ponds is generated through algae photosynthetic activity (Shilton and Guieysse 2010, García *et al.* 2006, Shilton *et al.* 2006, Craggs *et al.* 2004). In addition, algae photosynthetic activity contributes to elevated pH of the growth medium (Rogers *et al*. 2014, Sutherland *et al*. 2014a). A high pH environment aggravates volatilization of ammonia from the surface of these ponds (Appels *et al*. 2008). Ammonia emissions can exceed 90 % under these conditions (Chen *et al*. 2012, Appels *et al*. 2008, Rose *et al.* 1996). Phosphorous however settles out of suspension or is assimilated as a constituent of the microbial biomass (Rose *et al*. 1996, Nurdogan and Oswald 1995). Bacteria in the ponds can assimilate P but are not responsible for the majority of P sedimentation (Nurdogan and Oswald 1995). The greatest proportion of P is accumulated in the algae biomass (Rose *et al*. 1996, Nurdogan and Oswald 1995).

Mixing, or turbulent flow, is essential to maintain optimum conditions for maximum production of algae (Chaumont, 1993). Mixing can be achieved using air lifts, injectors, propellers, pumps, gravity flow and paddlewheels (Yadala and Cremaschi 2014, Chaumont 1993). Apart from preventing thermal and oxygen stratification, the gentle paddlewheel mixing maintains the surface velocity required to keep algae and algae flocs in suspension near the surface within the sunlight penetration zone. Mixing also allows the larger bacterial flocs to use the photosynthetic oxygen and to move more slowly along the lower section of the HRAOP. A linear velocity of 0.50 $m.s^{-1}$ appears to be sufficient to achieve de-stratification and to hold the algae in suspension and inhibit settling (Rose *et al.* 2002a). However, a 0.50 $m.s^{-1}$ channel velocity is very difficult to maintain due to frictional losses, especially in the bends. Thus, a velocity of 0.20-0.30 $m.s^{-1}$ is deemed adequate for algae growth and accumulation of biomass (Wells 2005, Rose *et al*. 2002a). Flow simulations carried out by Hadiyanto *et al.* (2013) increased the rotation speed of the paddlewheel, to enhance turbulent flow. Hadiyanto *et al*. (2013) found that by doubling the velocity of the water stream they were able to increase the productivity of the HRAOP by 50 %. Unfortunately, greater velocity requires substantially more energy which might compromise the cost effectiveness of using HRAOPs for WWT.

UV light disinfection occurring in HRAOPs during the day results in an effluent that is free of pathogens, while nocturnally, an elevated pH and hyperoxygenation facilitate disinfection (Park *et al.* 2011a, Oswald *et al.* 1957, Oswald *et al.* 1955). Oxygen is also needed to breakdown residual BOD in order to prevent anaerobicity in any receiving water body (Park *et al.* 2011b). Park *et al*. (2011b) demonstrated that domestic wastewater contains only half the C required by algae for N assimilation. Typically a C:N ratio of 3:1 is present in domestic wastewater whereas in algae biomass this C:N ratio is 6:1 (Park *et al.* 2011b). Thus, it has been suggested that a supplementary C source might be needed to completely remove N contaminants and increase biomass yield (Craggs *et al.* 2012). This C limitation is further aggravated by irradiance-induced changes in pH, causing C loss from the system (Sutherland *et al.* 2015). Essentially, an increase in light intensity causes the water temperature to rise which decreases the solubility of CO_2 causing its decline in concentration in the system. This increase in water temperature also boosts algae productivity and the increased uptake of CO_2 from the water by algae which results in CO_2 depletion (Sutherland *et al.* 2015, Kayombo *et al*. 2003, Mashauri and Kayombo 2002). Photosynthetic C assimilation by algae elevates the concentration of hydroxyl ions in the water column which consequently increases the alkalinity (Shilton 2006, Tadesse *et al.* 2004). This occurs when carbonates are removed by algae and hydroxyl ions are released (Craggs *et al*. 2004, Tadesse *et al.* 2004) as follows:

$$2HCO_3^- \leftrightarrow CO_3^{2-} + H_2O \leftrightarrow CO_2$$

$$CO_3^{2-} + H_2O \leftrightarrow 2OH^- + CO_2$$

This increase in pH also promotes algae aggregation and adsorption of inorganic phosphate (Shilton 2006, Tadesse *et al.* 2004, Oswald *et al*. 1953). Under conditions of high light intensity the pH can increase to >10, which is lethal to faecal bacteria (Green *et al.* 1995, Pearson *et al.* 1995). Thus, high DO coupled with high UV light, and the presence of dissolved humic substances act as the major disinfectants against faecal bacteria (Curtis *et al.* 1992).

Algae Settling Ponds – Algae from HRAOPs form flocs and settle readily in ASPs. Floc formation is triggered by elevated pH (Sutherland *et al*. 2014a, Green *et al*. 1996). Oswald *et al*. (1990) noted that algae agitated in paddlewheel mixed environments were generally colonial forms that tended to settle out of suspension more easily. Following the treatment of such algae dense effluents by dissolved air flotation (DAF) or removal by sedimentation, it was found that the BOD was reduced to <20 mg.ℓ^{-1} (Ertas and Ponce 2012, Green *et al.* 1996). Advanced settling ponds are designed to ensure maximum settling of algae while limiting further algae growth. A 1-2 d residence is generally sufficient to remove 50-80 % of algae via sedimentation (Nurdogan and Oswald 1996). Al-Shayji *et al.* (1994) reported that algae floc formation and settling were sensitive to climate and operational parameters (e.g. sunshine, hydraulic conditions in the ASP, variations of the flow rate, and pond depth) and in the case of fluctuating conditions, 20-30 % of the biomass can be lost – presumably through the onset of cell death. Green *et al.* (1996) proposed that a hydraulic retention of 0.50 d was sufficient to encourage settling and to render a 2-4 % solids slurry (Green *et al.* 1996).

Tertiary Treatment – Water emanating from the terminal ASP component of IAPS is of a secondary quality. This water requires tertiary treatment to render it compliant with South African regulations for irrigation and/or discharge into the environment. Tertiary treatment entails further removal of N (>70 %), P (>80 %), BOD (95 %) and COD (85 %). Concentrations of faecal coliforms also need to be reduced to levels less than 1000 cfu.100mℓ^{-1} (DWS 2013, Hamoda *et al*. 2004, Green *et al*. 1996). Examples of tertiary treatment systems are described below;

Maturation Ponds – Among the various tertiary treatment processes available MPs are considered adequate for additional polishing (Mara 2005). Alkalinity, low temperatures, elevated DO and UV light radiation native to these systems are exploited for disinfection (Von Sperling and De Lemos Chernicharo 2005). These systems generally operate in series where 2 or 3 ponds, with a 12 d average retention provide sufficient pathogen removal. Maturation ponds have high algae diversity and are also fully oxygenated throughout the day (Mara 2005). An alkaline water characteristic to MPs, ranging in pH 9.5-10 destroys pathogens including *E. coli* (Chan *et al.* 2009, Von Sperling 2007, Grönlund *et al.* 2004), while UV radiation contributes to significant disinfection within the system (Green *et al.* 1996). This high pH also enhances N removal both via biomass assimilation and volatilisation (Kayombo *et al*. 2005). In fact, ammonia-N removal in MPs exceeds that of other tertiary treatment processes like constructed wetlands. A comparative assessment demonstrated that MP series were second only to aerated rock filtration (Johnson *et al.* 2007). Craggs *et al.* (2012) recommend the inclusion of one or a combination of MPs and UV light treatments by storage prior to discharge, or rock filtration of the MP effluent, or direct UV light treatment if insufficient land is available otherwise, if funds are permitting, membrane filtration can be used to achieve a high quality final effluent for reuse.

Dissolved Air Flotation – this is a tertiary treatment process that utilizes microbubbles to remove suspended solids like algae (Kim *et al*. 2015). Kim *et al*. (2015) concede that the efficiency of the DAF process depends on factors inclusive of quantities of bubbles, bubble size, hydraulic loading and the depth of the bubble layer. It is a cost effective alternative to conventional sedimentation clarification processes. Concerning reliability, it is however, a skills and energy intensive process and can become cost prohibitive in the long term given a rural setting (Shilton 2006, Mara 2005). Chlorination, commonly utilized to supplement treatment of secondary wastewater to a tertiary quality, can result in the rupture of algae cells concomittantly causing increases in extra and intracellular organic C compounds in the WWTW outflow (Jeong *et al.* 2015).

Slow Sand Filtration – SSF is a long-standing, simple, inexpensive, low energy, malleable process that relies on physical and biological activity (Aslan and Cakici 2007). When compared with ion exchange and reverse osmosis, though these technologies are faster and more effective, they are however susceptible to membrane fouling (reverse osmosis), have poor selectivity for nitrate-N, generate a secondary waste requiring disposal, require a concentration threshold and are extremely cost and skills prohibitive when given a resource strained setting (Casas and Bester 2015, Aslan and Cakici 2007). Fortunately SSF requires no chemical dosing to generate water of a high quality. A filter bed may be constructed utilizing a material with a high surface area that can be colonized by microorganisms such as *Pseudomonas* sp. and *Trichoderma* sp. This media also forms a filter preventing the passage of suspended particles such as spores and algae debris. The slow flow rates of effluent through this bed result in the efficient remediation of wastewater as colonizing antagonistic microorganisms treat the water while the filter bed traps most debris (Casas and Bester 2015). Hamoda *et al*. (2004) reported 95 % suspended and volatile suspended solid removal and the elimination of a combined 99 % BOD and COD from a secondary treated effluent.

Rock Filtration – Rock filters/controlled rock filters (CRF) can be used to remove algae from effluents as they consist of submerged beds of rocks ranging in size from approximately 0.08-0.20 m in diameter. Vertical flow rock filters reportedly produce the best performance (Middlebrooks 1995). Short *et al.* (2007) were able to enhance the water quality from an open pond system using controlled rock filtration for reuse.

Quaternary and/or quinary treatment are not typically components of AIWPS/IAPS WWT systems and will therefore not be discussed here.

1.5 IAPS as a Technology for Municipal Sewage Treatment: The Challenge

Commercial scale installation of IAPS as a municipal sewage treatment process is controversial due to the perceived poor performance of the technology (Nemadire 2011, Laxton 2010). Dissection of this perception is regularly hampered by the availability of foreign, efficient and established package plants, such as AS and TF often accompanied by foreign expertise (Nemadire 2011, Rose *et al.* 2002a). Yet these options may prove inappropriate in the long term, where developing countries lack the necessary infrastructure, expertise and financial resources required to fully exploit these technologies (Nemadire 2011). These conditions are however uniquely suited to the implementation of passive technologies like IAPS. This process takes advantage of naturally occurring warm temperatures, sunlight and gravity and has the potential to generate an algae biomass for fertilizer and methane gas for fuel (Tijjani 2011, Rose *et al.* 2002a, Green *et al.* 1996). Thus it becomes imperative to address the concerns surrounding the deployment of IAPS. Findings by Rose *et al*. (2007) on the water quality from the ASP suggest a WWT technology unable to consistently produce an effluent with $\leq$75 mg.ℓ^{-1} COD, $\leq$10 mg.ℓ^{-1} nitrate-N, $\leq$6 mg.ℓ^{-1} ammonia-N, $\leq$25 mg.ℓ^{-1} TSS and $\leq$1 000 cfu.100mℓ^{-1} faecal coliforms, warranting further investigation to determine the cause of this poor performance. Meiring and Oellerman (1995) previously suggested that the elevated COD and TSS of the treated water from IAPS was a consequence of inadequate design and operation. An elevated COD in the outflow can potentially cause anaerobicity in receiving water bodies (Park and Craggs 2011); yet to circumvent this Green *et al.* (1995) reported that the water containing algae from the ASPs could be used directly for landscape irrigation and horticulture.

Craggs *et al.* (2012) later advocated the need for additional treatment of the outflow from ASPs to meet specific environmental discharge standards. Jeong *et al.* (2015) argue against chlorination to deactivate the algae biomass in this outflow as it causes algae cell rupture releasing both intra and extracellular polysacharrides. Not only do these polysaccharides contribute to increased dissolved organic C content, they also form chlorinated compounds which may be toxic. Chlorinated nitrogenous disinfection by-products are 140 times more toxic than carbonaceous disinfection by-products (Jeong *et al.* 2015). Thus it becomes beneficial to minimize the use of chlorination to disinfect this effluent opting rather for UV disinfection (Green *et al.* 1995). Xu *et al*. (2014) and Craggs *et al*. (2012) maintain that the raceway component of an IAPS can be utilized effectively to polish wastewater and to generate a disinfected algae biomass. Algae harvest however remains the problem, though microfiltration is an option (Nemadire 2011, Craggs *et al.* 2005, Chaumont 1993, de la Noue *et al.* 1992). Green *et al.* (1995) found that a 10-15 d retention in an MP series resulted in sufficient faecal coliform disinfection for discharge into constructed wetlands (Green *et al.* 1995). Green *et al.* (1995) also proposed the use of DAF followed by filtration then UV light disinfection which would result in a water quality from IAPS suitable for unrestricted use.

Pond systems, specifically the traditionally deployed WSPs though passive, require more land than IAPS. Further unlike the IAPS, WSPs do not offer the option to harvest by-products such as methane for heating and cooking or fertilizer for agriculture (Laxton 2010, Rose *et al.* 2007). Thus far, components of IAPS such as HRAOPs in use in Ashton, Western Cape treat 4 $M\ell.d^{-1}$ effluent from fruit canning while the I-PD is used to treat 1.6 $M\ell.d^{-1}$ effluent from tomato canning in Musina, Limpopo (Laxton 2010). These installations demonstrate the versatility of the components of IAPS. In addition, several South African-based projects have been initiated for deployment of the IAPS technology to treat municipal sewage, inclusive of a UNEP WioLap sponsored IAPS (1 $M\ell.d^{-1}$, Bushman‘s River, Nlambe Municipality), two Partners-for-Water sponsored IAPS (2 $M\ell.d^{-1}$ Grahamstown, Makana Municipality; 1.5 $M\ell.d^{-1}$, Alice, Amathole Municipality), and the conversion of a WSP system to an IAPS (2-3 $M\ell.d^{-1}$, Bedford, Amathole Municipality). Each project proceeded through the design stage but failed at implementation. Nemadire (2011) attributed the culmination of each to: failure by IAPS to meet South African authorization limits, scarcity of required skills, availability of other technologies (e.g. AS), and delay due to conflicts with stakeholders. Laxton (2010) pointed out a vital factor required for the successful deployment of this type of technology; a multidisciplinary understanding of the technology to comprehensively address socio-economic and environmental prospects to fully benefit from IAPS.

At present many WWTW in South Africa are in disrepair, operate below capacity and intermittently fail while demands on service delivery continue to increase (DWS 2012).

Wastewater treatment can no longer be a linear practice ensuring disease mitigation and eliminating environmental pollution (Rose *et al*. 2002a, Oswald *et al*. 1990). Rather employment, energy and product generation need to ensure valorization of WWT practices (Mambo *et al.* 2014a). In addition, though capital costs should be given their due consideration, long term running and maintenance costs of such facilities are equally if not more important (Nemadire 2011, Laxton 2010). It becomes crucial when a facility malfunctions and foreign expertise is required to resolve technical issues (Nemadire 2011).

Nemadire (2011) states IAPS are 50-70 % more cost competitive to construct, set up and run than AS, TF, and other conventional WWT technologies (Nemadire 2011, Laxton 2010). It is also a simple technology requiring very low skill levels to operate (Rose *et al.* 2007). The IAPS is a robust technology that is not plagued by electrical issues as with an AS and extended aeration. Construction of IAPS is such that the land can be reclaimed in future for a different purpose, therefore the construction of such a system is not environmentally detrimental (Rose *et al*. 2002a). Minimal technical equipment such as a pump and electrical batteries for the paddlewheels are required for IAPS functionality (Van Hille *et al.* 1999). Thus when governments have to bear the capital investment of constructing an IAPS as well as maintenance, costs associated with IAPS are highly competitive (Rose *et al.* 2002a). Further economically low level jobs are created (Harun 2010). It has also been observed that improved access to clean water simulates social and economic development (Adewumi *et al*. 2011, Oswald 1995). The effluent generated by IAPS can be utilized in brick making, chicken farming, food gardening and as a fertilizer (Rose *et al.* 2002a, Green *et al.* 1995). These passive systems are able to withstand varying shock loads while when correctly configured are also capable of consistently generating the desired effluent quality (Muga and Mihelcic 2008, Oswald *et al.* 1994). As minimal skill levels are required for operation and maintenance of a technology like IAPS, overall it will cost less over an extended period of time to run (Mwabi *et al.* 2011, Sivakumar *et al.* 2012, Flores-Alsina *et al.* 2011). However, despite the vast potential benefits of this technology, the poor performance of the Belmont Valley pilot IAPS needs to be assessed and addressed.

2 Aims

The challenges outlined above coupled with the dire state of water and sanitation infrastructure and service delivery in South Africa demands that closer scrutiny be given to all available WWT technology solutions. Thus, the present study has as its major objective, a re-evaluation of IAPS as a technology for municipal sewage treatment. This investigation forms part of a broader WRC-funded project that seeks to provide answers to questions about IAPS posed by authorities prior to implementation of this WWT technology. Within this framework the aims of the present study were to:

- Determine the state of the art of IAPS as a WWT technology and to provide a retrospective after re-evaluation of data from earlier studies on the pilot-scale IAPS located at the Belmont Valley WWTW;
- Re-evaluate the design and operating configuration of the Belmont Valley IAPS and determine the parameters needed to ensure system performance and sewage treatment efficiency to produce a water of a quality suitable for either irrigation or discharge to a water resource in terms of the general authorisation limit values;
- Monitor water quality from the Belmont Valley IAPS over an extended period and attempt to characterize residual COD and/or TSS in the final effluent;
- Identify and make recommendations for best practice to ensure nutrient removal efficiency and performance of IAPS in terms of compliance and standards.

3 Results and Discussion

Details of experimental approach and set-up, methodology, and results from the present investigation have been compiled into a series of manuscripts and these are presented in the, Appendix (Chapter 6). A summary of the purpose of each manuscript and the major findings follows.

3.1 The Belmont Valley integrated algae pond system in retrospect (Appendix 6.1)

This paper, published in Water SA (Mambo *et al*. 2014a), presents an overview of AIWPS-IAPS as a municipal sewage treatment technology, a retrospective on its introduction into the South African water sector, and summarizes current knowledge about this bioprocess technology. In brief, IAPS as a municipal sewage treatment technology was re-examined in relation to design and operation, the underpinning biochemistry of nutrient removal by algae was described, and a retrospective was provided on the demonstration system at the Belmont Valley WWTW. In addition to presenting details of the process flow, several shortcomings and/or oversights were highlighted and, in particular, the need for an appropriate tertiary treatment component. However, despite the use of IAPS for sewage treatment in many countries, this technology is still viewed with some skepticism. Thus, a major purpose of this overview was to provide a synthesis of available information on IAPS and an appraisal of its use for municipal sewage treatment. It was determined that in the absence of a tertiary treatment process the IAPS was operating in line with the Secondary AIWPS model, designed to produce water of a secondary quality. Therefore the water quality leaving the system would be unsuitable for either environmental discharge or irrigation. Thus it became imperative to generate a comprehensive dataset evaluating this bioprocess technology. This information could improve the understanding of the technology and provide authorities with the confidence needed when deciding on the implementation of an appropriate algae based WWT technology.

3.2 Integrated Algae Pond Systems for Municipal Sewage Treatment: A Gap Analysis (Appendix 6.2)

This manuscript describes a gap analysis of IAPS as a WWT technology and specifically addresses the gap that exists between best practice and current operations using the Belmont Valley WWTW IAPS as a case study. This pilot was implemented to demonstrate operation of IAPS technology under South African conditions. Unfortunately, extended studies revealed an outflow water quality routinely unsuitable for either irrigation or discharge to a water resource. Thus, configuration and operation of the Belmont Valley WWTW IAPS were re-evaluated in an effort to identify the underlying reasons for poor water treatment capability. Component parts of the IAPS and overall WWT efficacy were assessed. Both metadata and real time data derived from the pilot IAPS at the Belmont Valley WWTW were used to supplement data for the gap analysis. The study found that though the pilot was originally designed and configured to receive an influent BOD loading of 40 $kg.d^{-1}$ it was in fact receiving a calculated 80 $kg.d^{-1}$, double the original design specification. Also, water from ASP A was not returned to the IAPS inlet but rather channelled to the municipal WWTW inlet and thus not available to potentially dilute the 80 $kg.d^{-1}$ influent BOD to the prescribed 40 $kg.d^{-1}$. Furthermore, this IAPS lacks an inlet works, tertiary treatment and disinfection components, resulting in partial WWT which would contribute to a poor final effluent quality. Both the organic load into the system and the harvesting of biomass appear to be compromised and directly contribute to elevated TSS and COD of the treated water. Population dynamics of the algae/bacterial biocatalyst within HRAOPs may also contribute to reduced nutrient abstraction particularly when diatoms dominate and out-compete algae. Efforts to beneficiate products of IAPS WWT by downstream value adding remain unrealized. Indeed, the products which include treated water for recycle and reuse, biogas for electricity/heating, and biomass have for most part not been accurately quantified. Lastly, the gap analysis revealed surprisingly limited and only rudimentary information relating to the costs of construction, implementation and operation of IAPS as a sewage treatment technology. It was concluded that an incorrect operating procedure had contributed to poor water quality data and a negative perception of IAPS for municipal sewage treatment in South Africa. In addition, insufficient information about this technology and its apparent benefits is available to facilitate decision making by the water and sanitation sector.

3.3 Operation of an integrated algae pond system for the treatment of municipal sewage: a South African case study (Appendix 6.3)

In this paper, published in Water Science and Technology (Mambo *et al.* 2014b), data on the operation of the Belmont Valley WWTW experimental IAPS treating municipal sewage, the quality of the treated water, and the contribution of various tertiary treatment processes used to polish and enhance water quality prior to discharge is presented. In summary, spectrophotometric assays indicated that the treated water from this IAPS was compliant with the discharge limits for phosphate-P, ammonium-N and nitrate/nitrite-N, and mean values were: 5.3, 2.9 and 12.4 mg.ℓ^{-1}, respectively. Chemical oxygen demand however, fluctuated significantly and was dependent on full function of the IAPS. Mean COD of the final treated water was 72.2 mg.ℓ^{-1}. Although these results suggest that the treated water discharged from this IAPS operating under South African conditions meets the standard for discharge, mean TSS was confirmed as being routinely above the limit at 34.5 ± 13 mg.ℓ^{-1} and faecal coliforms were higher than expected.

3.4 Chemical Oxygen Demand and Total Suspended Solids as Limiting Factors in Integrated Algae Pond Systems (Appendix 6.4)

In this paper attention is focused primarily on the COD and TSS content of the final outflow after treatment of municipal sewage using the Belmont Valley WWTW IAPS. In summary, measurements from March 2013-November 2013, of COD (97.7 ± 15.7 mg.ℓ^{-1}) and TSS (35.0 ± 12.3 mg.ℓ^{-1}) rendered the effluent non-compliant with the general authorization limit values for either irrigation or discharge to a water resource. Sequential microfiltration revealed that a substantial portion of the COD was <0.22 µm, suggesting a COD comprising largely of dissolved organic C rather than particulate organic C. In addition to the expected nutrients nitrate/nitrite-N, ammonium-N, and *ortho*-phosphate a range of compounds typically associated with dissolved organic matter (C) were detected. These organics included carbohydrates, reducing sugars, proteins, α amino-N, and humics indicative of activity in a microbial loop which presumably involves programmed algae cell death occuring at night (i.e. dark induced) and most likely in the HRAOPs and ASPs. A TSS concentration consistently >25 mg.ℓ^{-1} attributed to algae and cell debris, also reinforces the need to efficiently polish effluent from ASPs using an appropriate TTU positioned appropriately prior to either irrigation or environmental discharge.

3.5 Design configuration and process flow of IAPS for municipal sewage treatment

The IAPS pilot located at EBRU adjacent to the Belmont Valley WWTW (33° 19′ 07″ S, 26° 33′ 25″ E) was evaluated in terms of design configuration and process flow. Figure 5 presents an aerial photograph of the plant. The facility treats 75 kℓ.d^{-1} municipal sewage and details of the operating configuration and process flow are shown in Figure 6. The system comprises an AFP containing an I-PD (840 m^2) and a primary facultative pond with a surface area of 225 m^2, two HRAOPs (500 m^2) and two ASPs. This facility differs from AIWPS convention, specifically the Richmond, Delhi and St Helena examples in not having as component parts either a TTU to polish the outflow from the ASPs or a dedicated inlet works. Inlet works are vital for screening and grit removal.

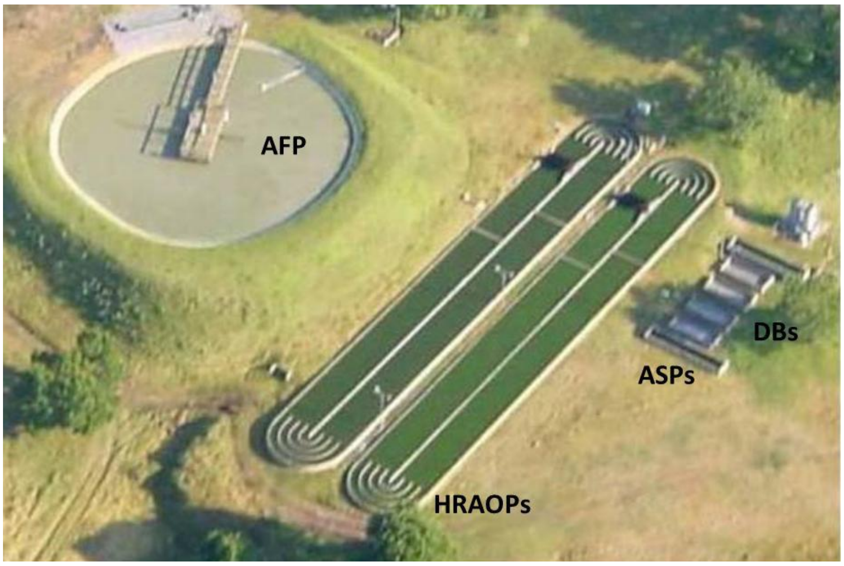


Figure 5 An aerial photograph of the IAPS pilot operating at the Belmont Valley WWTP treating 75 kℓ.d^{-1} municipal sewage. The pilot is composed of an AFP: Advanced Facultative Pond, which is a combination of an I-PD and a primary facultative pond, 2 HRAOPs: High Rate Algae Oxidation Ponds, 2 ASPs: Advanced Settling Ponds and 2 DBs: Drying Beds. Note the absence of an MP required for post treatment.

Currently municipal sewage enters the I-PD, where suspended and dissolved solids are anaerobically degraded. The upward flow of 1–1.5 m.d^{-1}, ensures water retention in the I-PD for 3 d and 20 d in the primary facultative pond. Effluent then gravitates from the facultative pond into the first of 2 HRAOPs. Paddlewheel mixing, or turbulent flow, is essential to maintain optimum conditions for maximum algae productivity in the HRAOPs. Typically, this linear flow achieved using paddlewheels and required to prevent stratification, is powered by a small electrical motor (250 J.s^{-1}). Water undergoes nutrient scrubbing for 2 d in the first HRAOP, and is then channeled to the first ASP for half a day. Due to the configuration of this pilot and in accordance with original design parameters (Rose *et al.* 2002), only 50% of the partially treated water from the first ASP gravitates to the second HRAOP where it is detained for 4 d before release to the second ASP, where the bulk of suspended algae biomass is removed by settling. All effluent, treated and partially treated, is returned to the municipal WWTW inlet works.

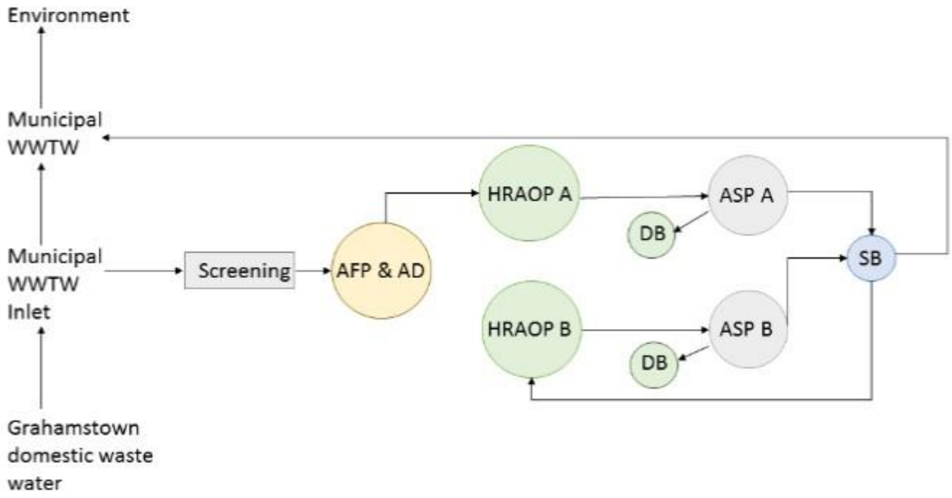


Figure 6 Schematic of the flow of domestic wastewater through the Belmont Valley IAPS. Ideally the system would have an inlet works and screening facility prior to the IAPS. From the WWTW: Wastewater treatment works, water enters the AFP: Advanced facultative pond, AD: Anaerobic digester also I-PD: In Pond Digester, and decants into the HRAOP: High rate algae oxidation pond, and if released into an ASP: Advanced settling pond, where from the SB: Splitter box, water is directed to the municipal WWTW while settled algae are placed on the DB: Drying bed.

Inlet works which may provide coarse screen, rag catcher, grit channel and flow meter services to WWTW are absent from the pilot. Not only would these services prevent damage to the facility, a flow meter would ensure regulation of the influent velocity and quantity into the plant. In future an inlet works must be included in the commercial design and implementation of IAPS technology. Inlet works are also recommended in the proposed Department of Public Works Design Guidelines for Small WWTW (2012).

Screening prevents blockages by removing larger floating and recalcitrant organic solids, which do not decompose. While grit channels remove heavy inorganic matter like grit, sand, gravel, road scrapings and ash. These particles do not decay but can damage pumps making sludge digestion difficult (Horan 1993). Screening at the inlet works removes 60 % large solid waste while BOD is also substantially reduced (Tchobanoglous *et al*. 2003). If overlooked in process design, the absence of an inlet works can result in the poorly treated water (Qi and Chang 2012, Tchobanoglous *et al.* 2003, Horan 1993). In particular, the effluent leaving the system may contain elevated BOD and COD concentrations. Inlet works carrying out preliminary WWT manually screen wastewater while maintaining the influent velocity at >0.5 m.s^{-1} to prevent natural settling (Manickum and John 2014, Athanasoulia *et al.* 2014, Hesnawi *et al.* 2014). A well designed and operated grit removal chamber allows settleable solids to remain in suspension and these become C substrates for biological processes remediating wastewater downstream. Even suspension is attained at a velocity >0.3 m.s^{-1} (Tchobanoglous *et al.* 2003). In order to test the kinetic parameters as reported for the Belmont Valley IAPS (Rose *et al.* 2002a), the design criteria were re-evaluated using metadata obtained by analyzing the raw sewage influent entering the system (Rose *et al.* 2007) and the results are shown in Table 6.

Table 6 Effect of loading rate on influent flow as determined from the model developed to determine the kinetic parameters for IAPS implementation. Numbers in gray were derived experimentally and used to test the model using the documented design parameters reported by Rose *et al.* (2002a)

Parameter	Design	Actual
Volume (kℓ.d^{-1})		
PE equiv	500	500
Vol per PE.d^{-1}	0.15	0.15
Flow	75	75
Strength (mg.ℓ$^{-1}$)		
BOD	800	1080
COD	1000	1800
TKN	-	128
P_{total}	-	15
Loading (kg.d^{-1})		
BOD	**40**	**81**
COD	-	135
TKN	-	9.6
P_{total}	-	1.125

Using the McGarry Model (McGarry and Pescod 1970), an AFP area of 382.8 m^2 and a hydraulic retention time (HRT) of 9.2 d were obtained (Table 7), this is similar to the design specifications of the Belmont Valley IAPS suggesting that the AFP parameters are accordingly sufficient for a 500 population equivalent (PE) IAPS with an influent loading of 40 kg.d^{-1} BOD and indeed, the actual measured BOD of the outflow was between 300 and 444 mg.ℓ$^{-1}$ (cf. estimated BOD of outflow of 486 mg.ℓ$^{-1}$). Results however show that BOD loading of the I-PD is close to 0.4. Values above 0.4 are indicative of malodours, which would be a distinct disadvantage in deployment of the IAPS technology for treatment of municipal sewage. Secondly, closer scrutiny of the parameters for the AFP indicate that with an effluent BOD at 80 kg.d^{-1} and according to first order kinetics, pond surface area and HRT should be 1389 m^2 and 33.3 d respectively which contrasts with the design parameters used of 840 m^2 and 20 d (Rose *et al.* 2002).

Table 7 Design parameters for the Belmont Valley IAPS IPD and AFP based on the model developed to determine the kinetic parameters for IAPS implementation (Rose *et al.*2002a, Rose *et al.* 2007).

Component Parameter	Value	Units	Comments
In-Pond Digester			
Optimum retention time	3	d	
Volume needed	225	kℓ	
Depth	5	m	
Area	45	m^2	
Diameter	7.5	m	
BOD loading	<u>0.36</u>	kgBOD.kℓ.d^{-1}	Odour problem above 0.4
Estimated BOD reduction	55	%	
Estimated BOD of outflow	486	mg.ℓ^{-l}	
Advanced Facultative Pond (PFP)			
Depth	1.8	m	
Aspect	3		
Minimum temp	12	°C	For first order kinetics approach
Minimum temp	53.6	°F	For McGarry Model
Effluent BOD	80	mg.ℓ^{-l}	
<u>First Order Kinetics Approach</u>			
Area	<u>1389</u>	m^2	First order kinetics eq. 7.13
Diameter	42.04	m	
BOD surface loading	0.026	kgBOD.m^{-2}.d^{-1}	
HRT	<u>33.33</u>	d	Much longer than current 20 d
<u>McGarry Approach</u>			
Area	<u>382.87</u>	m^2	McGarry Model eq. 7.17 (valid 15-30°C)
Diameter	22.08	m	
BOD surface loading	0.0952	kgBOD.m^{-2}.d^{-1}	
HRT	<u>9.20</u>	d	Close to Belmont Valley IAPS

On the basis of measured values for nitrate-N and *ortho*-phosphate in the AFP effluent of 80 and 15 mg.ℓ^{-1} respectively, the specific loading of N and P to the HRAOPs was determined to be 600 000 and 112 500 kg.m^2. Using a raw sewage total Kjeldahl nitrogen value of 128 mg.ℓ^{-1} (Table 6), the N and P loads were determined to be 6.24 and 0.96 kg.d^{-1} with a required HRAOP surface area for abstraction of these nutrients in the range 850-1040 m^2, which compares favourably with the design specifications of the Belmont Valley IAPS HRAOPs. These have combined total surface area of 1000 m^2 (Rose *et al*. 2002b).

Actual strength of the raw sewage as measured over the course of 2 years revealed a mean organic loading (COD_{max}) of 1 800 mg.ℓ^{-1} and a maximum BOD of 1080 mg.ℓ^{-1}. These values are two-fold the initial design parameters and indicate that the strength of municipal sewage entering from the Belmont Valley WWTW, should be reduced by 50 % in order to achieve the correct BOD loading rate and derive the efficiency required. It also is worth noting that the Belmont Valley IAPS does not have sufficient HRAOP capacity to polish all the water from HRAOP A. This is as a consequence of insufficient land availability coupled with research needs. Thus, 50 % of the water is redirected to the municipal WWTW as shown in Figure 7. Ideally, and based on the HRT required to abstract nutrients, the total volume from HRAOP A (with HRT= 2 d) should gravitate to a second HRAOP of dimensions sufficient to achieve the required 4 d HRT for nutrient removal. Thus, for the Belmont Valley IAPS to mirror a commercial set-up an additional HRAOP C is required or alternatively, the size of HRAOP B should be increased two-fold.

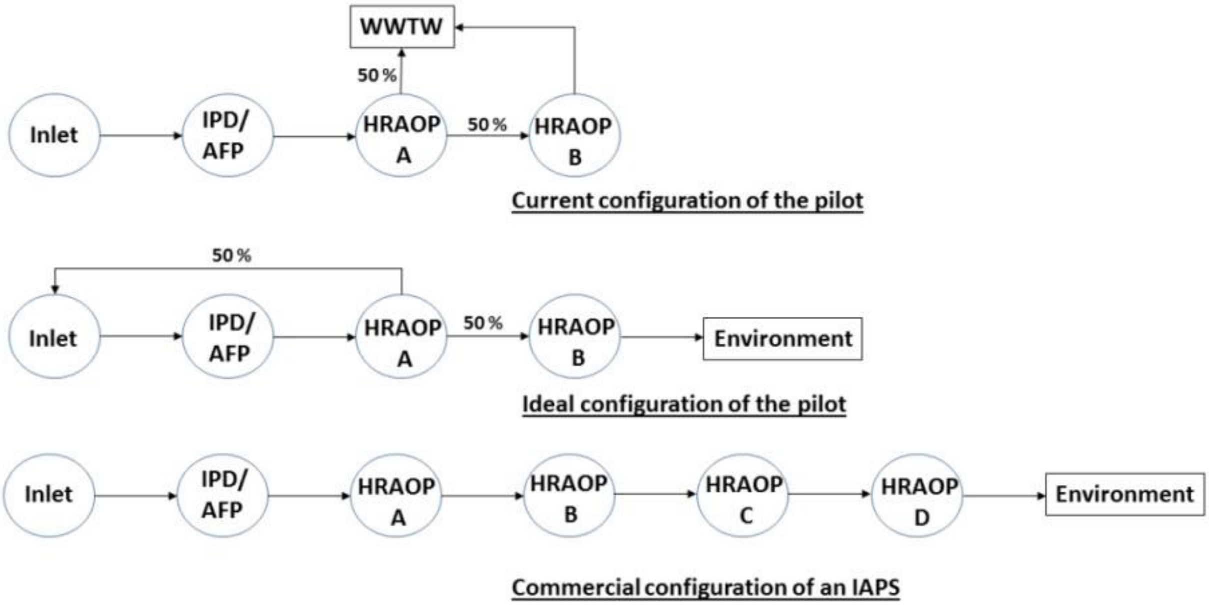


Figure 7 A basic schematic of the current configuration of the IAPS treating domestic in comparison the ideal operation of the Belmont Valley.

The AIWPS designed and implemented on a commercial scale in California, U.S.A. include an MP series for post treatment. Studies on a range of TTUs have revealed that using an MP series, SSF, or a CRF ensure the final treated water from the IAPS is of a quality suitable for either irrigation or discharge to the environment with CRF> SSF> MPS (Westensee 2015 MSc, Mambo *et al*. 2014a). Craggs *et al*. (2014) utilized a complete IAPS design, which included an MP and a CRF system for the treatment of the water from the ASPs. As a result the outflow was of a tertiary quality. Furthermore, Craggs *et al.* (2014) noted a 20 % improvement in the effluent quality when CO_2 was injected into the HRAOPs. Improved access to CO_2 increased algae photosynthetic activity. As algae assimilated CO_2 from the water, productivity increased and so did algae biomass yield. In addition the alkalinity increased causing ammonia volatilization and *ortho*-phosphate precipitation (Craggs *et al.*

2014). Elevation in alkalinity causes algae aggregation and the adsorption of inorganic phosphate indirectly improving the water quality (Tadesse *et al.* 2003, Oswald *et al.* 1953). The operating conditions for the IAPS pilot differ from the original Oswald designs; this may have a negative effect on the performance of the pilot. This system does not have a TTU, or an inlet works, further, it receives influent BOD loading of approximately 80 $kg.d^{-1}$, double the original design specifications and the total HRAOP capacity is insufficient. These findings render it imperative to measure the quality of the outflow over an extended period. For IAPS to serve as a viable WWTW treating domestic sewage in South Africa, the effluent is required to comply with general authorization limits for environmental discharge and irrigation.

3.6 IAPS and water quality: compliance and standard

In line with the Waste Act (Republic of South Africa *Waste Act* 2008), effluent from the IAPS should, in brief, not pose a threat to public or environmental health. Thus IAPS outflow is required to comply with DWS (2013) general authorization limits for discharge to the environment or irrigation. Final water quality from this pilot consequently holds significant sway over the overall perception of IAPS as a WWT technology in South Africa, ergo deployment readiness, applicability and reliability of the technology under South African conditions. Given the abovestated absence of an inlet works and lack of a post treatment system inconsistent with AIWPS convention, it appears to substantiate the poor final water quality measured by Rose *et al.* (2007). Also, as the facility is currently receiving an influent BOD load approximately double the original design specifications; it seemed pertinent to re-assess the water quality and to characterize the COD and TSS in the outflow.

To assess the water quality from the IAPS treating municipal sewage, spectrophotometric assays of the effluent decanting from ASP B to be returned to the Municipal WWTP, were carried out on a weekly basis over two periods specifically February 2013-February 2014 and September 2012-May 2013. The results for the February 2013-February 2014 period are presented in Figures 8, 9, 10, and 11 whereas those for September 2012-May 2013 are presented in Appendix 6.3 (Mambo *et al.* 2014).

Figure 8 shows changes in nutrient composition of the effluent for the period February 2013-February 2014. Figure 8B shows that nitrate-N concentration was consistently within the general authorization limits where the mean value detected was 2.3 ± 1.7 $mg.\ell^{-1}$. However *ortho*-phosphate was intermittently >10 $mg.\ell^{-1}$ (Figure 8A) while ammonium-N exceeded 6 mg.ℓ$^{-1}$ (Figure 8C). Though non-compliance was sporadic this outflow was unsuitable for either discharge to a water resource or irrigation. *Ortho*-phosphate peaked in week 44 at 13.2 ± 0.2 mg.ℓ$^{-1}$ while ammonium-N peaked in week 43 measuring 8.5 ± 0.1 mg.ℓ$^{-1}$. These findings indicate nutrient values slightly above the general authorization limits, however mean values for *ortho*-phosphate 4.3 ± 1.7 mg.ℓ$^{-1}$ and 2.6 ± 1.1 mg.ℓ$^{-1}$ ammonium-N imply an outflow capable of compliance. Therefore post treatment should be sufficient to render these deviant concentrations consistently compliant with DWS (2013) stipulations for discharge to a water resource or irrigation.

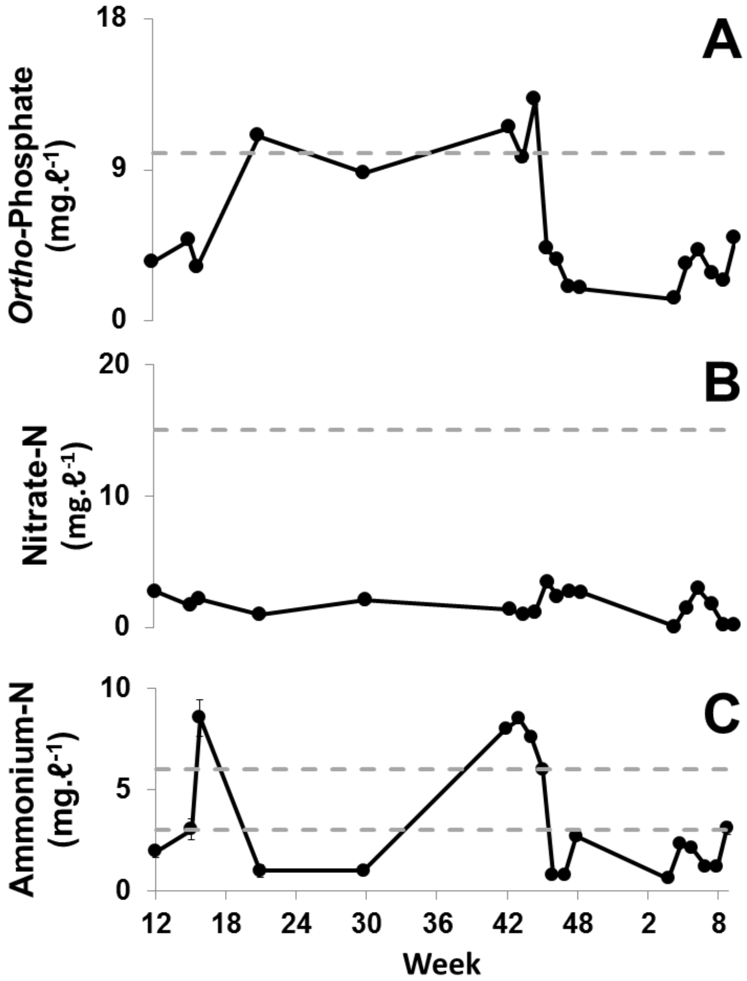


Figure 8 Changes in the nutrient composition in the effluent from the IAPS recorded from February 2013-February 2014. Data points are the average of triplicate measurements ± standard deviation while the gray-dotted line indicates the general limit values for environmental discharge for *ortho*-phosphate (≤10 mg.ℓ$^{-1}$), nitrate-N (≤15 mg.ℓ$^{-1}$) and ammonium-N (≤3 mg.ℓ$^{-1}$/≤6 mg.ℓ$^{-1}$ irrigation) respectively.

Figure 9 presents data collected from February 2013-February 2014 for changes in COD and TSS observed in the outflow from the IAPS. TSS concentrations were routinely above the general authorization limit 25 mg.ℓ$^{-1}$. Figure 9A shows elevated TSS in week 12 2013 and again in week 4 2014 at 61 ± 7.6 mg.ℓ$^{-1}$ and 60 ± 1 mg.ℓ$^{-1}$ respectively. Mean TSS for that period was 34.5 ± 13.2 mg.ℓ$^{-1}$ rendering the outflow consistently unsuitable for either environmental discharge or irrigation. Though mean COD was more favourably 66.6 ± 12.7 mg.ℓ$^{-1}$, Figure 9B shows several data points exceeding the 75 mg.ℓ$^{-1}$ general authorisation limit, specifically in week 42 where the COD concentration was 116 ± 2.5 mg.ℓ$^{-1}$. Yet following a significant rainfall event specifically 80 mm in week 42, an apparent dilution effect was subsequently observed in week 43 when COD declined significantly from 116 mg.ℓ$^{-1}$ to 89.3 mg.ℓ$^{-1}$ and TSS from 30 ± 2.5 mg.ℓ$^{-1}$ to 21.5 ± 4.3 mg.ℓ$^{-1}$.

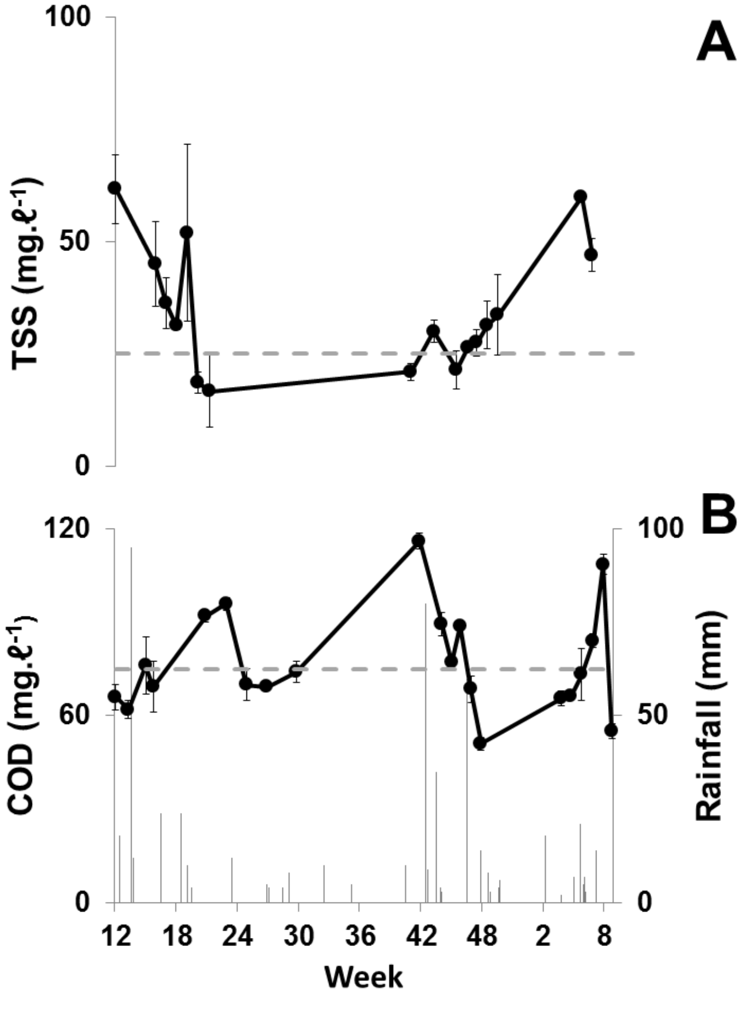


Figure 9 Changes observed in the TSS and COD of treated water from the IAPS at point of discharge. Gray-dotted line indicates the general limit values for TSS (≤25 mg.ℓ$^{-1}$) and COD$_{filtered}$ (≤75 mg.ℓ$^{-1}$) respectively. Data are presented as the mean of triplicate measurements ± standard deviation. Data was collected from February 2013-February 2014. The gray bar graph displays the rainfall events and the quantities measured.

Calculated mean values of 191 859 ± 40 169 cfu.100mℓ$^{-1}$ faecal coliform counts were higher than expected from February 2013-February 2014. Incorrect pipe placement feeding HRAOP

A from the AFP in weeks 12-18 resulted in short-circuiting which caused the elevated faecal coliform concentrations observed in the final effluent. Following correct placement of the inlet pipe, there was a significant decline in the occurrence of faecal coliforms as shown in Figure 10. However in the absence of a post treatment system mean faecal coliform counts were 3227 ± 2769 cfu.100mℓ^{-1}, remaining >1 000 cfu.100mℓ^{-1} and rendering the outflow unsuitable for release into the environment. However, this water quality is suitable for land and property irrigation as regulations require faecal coliform counts ≤ 100 000 cfu.100mℓ^{-1}.

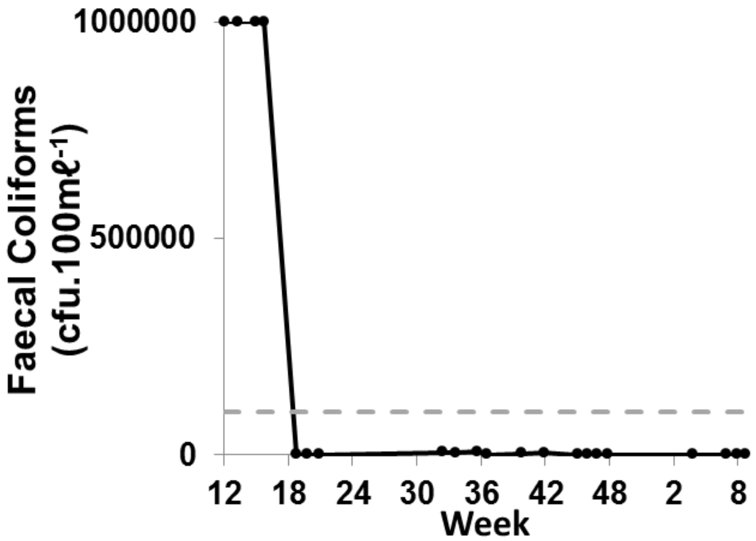


Figure 10 Changes in the faecal coliform counts observed in the the final effluent from the IAPS. Incorrect placement of a pipe feeding HRAOP A with water from the AFP resulted in the initial noise data recorded from week 12 to week 18 2013. Data points are the averages of triplicate measurements ± standard deviation. General limits require faecal coliform counts to be ≤1 000 cfu.100mℓ^{-1} for discharge to the environment, for which that gray dotted line is not apparent however for irrigation purposes faecal coliform counts should not exceed 100 000 cfu.100mℓ^{-1} as a result the outflow from the IAPS is of a quality suitable for irrigation.

Physical parameters measured in the outflow were routinely within the general authorization limits as shown in Figure 11 where DO should be ≥2 mg.ℓ^{-1} (Fig. 11B) and EC should not exceed 150 mS.m^{-1} of the influent EC (Fig. 11D). However where pH should range between pH 5.5-9.5, Figure 11C shows pH exceeded pH 9.5 from week 2-10 2014 averaging pH 10 ± 0.3.

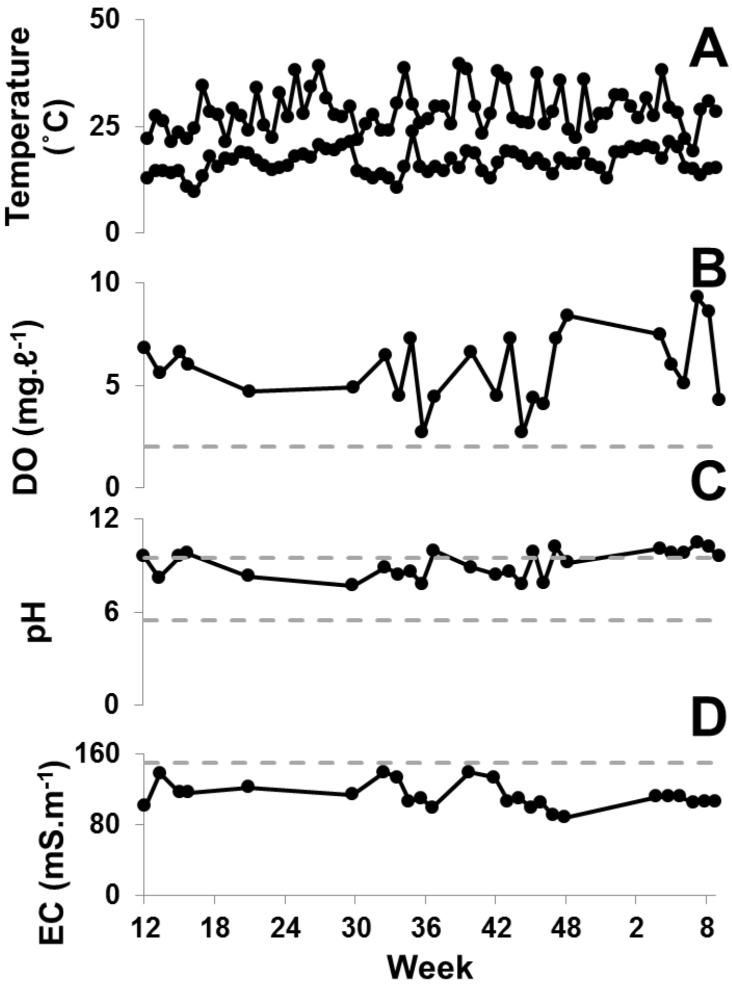


Figure 11 Changes in the physical parameters measured in the effluent from the IAPS. DO: Dissolved Oxygen and EC: Electrical Conductivity. Data was collected from February 2013-February 2014. Gray-dotted line indicates the general limit values for DO (≥2 mg.ℓ$^{-1}$), pH 5.5-9.5 and EC (≤70-≥150 mS.m^{-1}) respectively.

Table 8 provides a summary of the performance of the pilot IAPS in relation to the general authorization limits for discharge to the environment and irrigation. It should be noted that incorrect inlet pipe placement resulted in the initial elevated faecal coliform count, however once placement was corrected the counts were rendered suitable for irrigation. The elevated contaminating faecal coliform counts were sufficient to negatively impact the mean measurement. As expected the TSS content remained elevated due to the absence of a TTU, therefore the required treatment in an MP, CRF or SSF was not carried out.

Table 8 Summary data on water quality of the effluent from the IAPS for the sampling periods September 2012-May 2013 and February 2013-February 2014. Also shown are the General Authorization limits for discharge to a water course and irrigation (DWS, 2013). Data for effluent quality were determined on a per week interval during each period of monitoring. Findings presented in gray are mean values were significantly not compliant with the general limits.

Parameter	General Authorization Limit (Environmental Discharge)	General Authorization Limit (Irrigation)	IAPS Outflow Quality (September 2012- May 2013)	IAPS Outflow Quality (February 2013-February 2014)
Ortho-phosphate (mg.ℓ $^{-1}$)	10	10	5.3 ± 2	4.3 ± 1.7
Nitrate/nitrite-N (mg.ℓ $^{-1}$)	15	15	12.4 ± 4	2.3 ± 1.7
Ammonium-N (mg.ℓ $^{-1}$)	3	6	2.9 ± 1	2.6 ± 1.1
Chemical Oxygen Demand (mg.ℓ $^{-1}$)	75	75	72.2 ± 13	66.6 ± 12.7
Total Suspended Solids (mg.ℓ $^{-1}$)	25	25	34.5 ± 13.1	35 ± 14.2
Faecal Coliforms (cfu.100m ℓ $^{-1}$)	1000	100 000	>1 000	>1 000
pH	5.5-9.5	6-9.5	9.4 ± 1	9.1 ± 0.9
Dissolved Oxygen (mg.ℓ $^{-1}$)	≥2	≥2	5.5 ± 1	5.8 ± 1.7
Electrical Conductivity (mS.m^{-1})	70 mS.m^{-1} above intake-maximum 150 mS.m^{-1}	70 mS.m^{-1} above intake-maximum 150 mS.m^{-1}	107.8 ± 19	112.7 ± 14.3

Biochemical analyses carried out from March-November 2013 to characterize the COD in final effluent yielded the results presented in Table 9. These findings indicate that the bulk of COD is not retained by a 0.22 µm filter (specifically 71.3 mg.ℓ$^{-1}$ COD) strongly suggesting a COD composed mainly of dissolved organic rather than particulate organic C.

Table 9 Residual COD in the final effluent of the IAPS treating 75 kℓ.d^{-1} municipal sewage at the Belmont Valley WWTW, Grahamstown. Data followed by the same letter are not significantly different (one-way ANOVA, *p*<0.05). Data were from samples collected at regular intervals between March-November 2013 and are presented as the mean ± SD.

Filter pore size (µm)	Unfiltered	11	1.6	0.45	0.22
	COD (mg.ℓ$^{-1}$)				
Experiment 1					
Mean	126.8 ± 36.7^{a}	94.0 ± 15.6^{a}	74.4 ± 7.5^{a}	68.8 ± 10.6^{a}	71.3^{a}
% change in COD	0	-25.8^{b}	-41.3^{b}	-45.7^{b}	-43.8^{b}
% change in COD$_{filtered}$	-	0	-20.8^{c}	-26.8^{c}	-24.1^{c}
Experiment 2					
Mean	88.6 ± 25.5^{a}	76.8 ± 15.6^{a}	57.6 ± 11.1^{a}	58.8 ± 15.8^{a}	n.d.
% change in COD	0	-13.3^{b}	-34.9^{b}	-33.6^{b}	n.d.
% change in COD$_{filtered}$	-	0	-25.0^{c}	-23.4^{c}	n.d.

Results presented in Table 10 offer partial support for the COD being predominantly dissolved organic C. The amount of humic acid-like material in the water was measured both before and after sequential filtration. Though levels of humic acid-like substances were low, only after filtration through a 0.45 µm filter, was there any significant reduction in content of humic acid-like material suggesting that the bulk of this component comprised smaller humics (i.e. low molecular weight humics) and possibly fulvic acids.

Table 10 Concentration of humic acid-like substances measured in water from the Belmont Valley IAPS final effluent. Data are the mean ± S.D. of at least four determinations. Data were from samples collected at regular intervals between March-November 2013 and are presented as the mean ± SD.

Filter pore size (µm)	Humic-like substances ($mg.\ell^{-1}$)
Unfiltered	10.2 ± 0.10
11	9.7 ± 0.04
1.6	9.2 ± 0.10
0.45	6.4 ± 0.10

Biochemical analyses were also carried out to ascertain the contribution of other water-soluble compounds to the relatively high and persistent COD/TSS in the final effluent. Included in the analyses were measurement of carbohydrates, 49.7 ± 15.9 $mg.\ell^{-1}$; proteins, 15.3 ± 3.9 $mg.\ell^{-1}$; and reducing sugars, 19.3 ± 9.4 $mg.\ell^{-1}$; and these compounds made up the majority of the organics measured in the final effluent as shown in Table 11.

Table 11 Total soluble solids and metabolite content and composition of water samples from the IAPS final effluent. Data were from samples collected at regular intervals between March-November 2013 and are presented as the mean ± SD.

Total soluble solids ($mg.\ell^{-1}$)	31.3 ± 16.3
Metabolite	
Nitrate/nitrite-N ($mg.\ell^{-1}$)	14.9 ± 10.5
Ammonium-N ($mg.\ell^{-1}$)	3.3 ± 3.2
Ortho-Phosphate ($mg.\ell^{-1}$)	5.3 ± 4.0
Humic substances ($mg.\ell^{-1}$)	6.0 ± 1.7
Carbohydrates ($mg.\ell^{-1}$)	49.7 ± 15.9
Reducing sugars ($mg.\ell^{-1}$)	19.3 ± 9.4
Protein ($mg.\ell^{-1}$)	15.3 ± 3.9
Alpha amino nitrogen ($mg.\ell^{-1}$)	1.8 ± 1.1

Though water quality data from this study show this biological WWT technology can successfully remediate domestic wastewater, it does not however consistently produce water of a quality compliant with DWS (2013) general stipulations. A summary of data from the two periods of monitoring spanning September 2012-May 2013 and February 2013-February 2014 generated the results shown in Table 8. These results confirm that without post treatment the water quality from the IAPS is not suitable for discharge to the environment or for irrigation. Comprehensive assessment of the delinquent COD revealed that a significant portion of this was not retained by 0.22 µm filters, while the metabolite profile was predominantly composed of carbohydrates. It appears therefore that the operating configuration and conditions of the facility impact the overall performance of the plant. Undoubtedly, the incorporation of an appropriate TTU will further reduce COD and TSS levels of IAPS-treated water. Indeed, an evaluation of MP, SSF, and CRF revealed reductions in both COD and TSS to within the general limits either for discharge to a water resource or irrigation (Mambo *et al.* 2014b). Even so, WWT using IAPS without tertiary treatment still renders a significant improvement in overall water quality and presents a green technology that addresses current issues such as climate change (Rose *et al.* 2002) and sustainable development (Oswald *et al.* 1957) which, under South African conditions has the potential to render wastewater management both lucrative and efficient.

3.7 Suitability of IAPS as a technology for municipal sewage treatment

Poor effluent quality from an IAPS restricts the perceived efficacy and applicability of this technology. Rose *et al.* (2007) showed water from the Belmont Valley WWTW pilot IAPS contained COD_f >75 mg.ℓ^{-1}, TSS >25 mg.ℓ^{-1}, *ortho*-phosphate >10 mg.ℓ^{-1}, ammonia-N >3 mg.ℓ^{-1} while nitrate-N removal was inconsistent and sporadically, nitrate-N concentrations even increased. Present research has corroborated these findings and goes further to highlight the shortcomings of the original design and process flow. It is these operating conditions specifically the absence of TTUs that have undoubtedly contributed to poor water quality data and an apparent substandard performance by this pilot as reported earlier (Rose *et al.* 2007). In addition, the recent demonstration that programmed cell death (PCD) occurs in algae along with changes in dominance of the major algae biocatalyst indicates that algae physiology may play a vital role in contributing to nutrient addition rather than removal from both HRAOPs and ASPs.

First, this pilot IAPS was configured incorrectly; ideally and as presented in Figure 12, 50 % of the stream from HRAOP A should have been diverted from ASP A following algae harvest to the IAPS inlet to dilute influent sewage BOD from 80 kg.d^{-1} to approximately 40 kg.d^{-1}, the designed capacity. Reappriasal revealed however that this dilution stream was directed to the Belmont Valley municipal WWTW inlet works. Thus, the undiluted elevated organic load most probably contributed to poor system performance data and may also be responsible for intermittent sludge build up in the AFP, pipe blockages, and consequently poor outflow quality.

Second, the raceway system attached to the Belmont Valley WWTW pilot IAPS does not have the capacity to treat all the water from the HRAOP A. Thus HRAOP B should either be double its current size as shown in Figure 12 or a third HRAOP of equivalent capacity needs to be introduced. This will ensure the capacity of the raceways is sufficient to completely polish all the water from the HRAOP A. Currently only 50 % of the wastewater from HRAOP A and ASP A is completely treated from HRAOP B to ASP B.

Third, the pilot facility was designed and implemented without a dedicated inlet works and without an appropriate TTU. As a result, disinfection is erratic and the outflow can be described as being of a secondary quality.

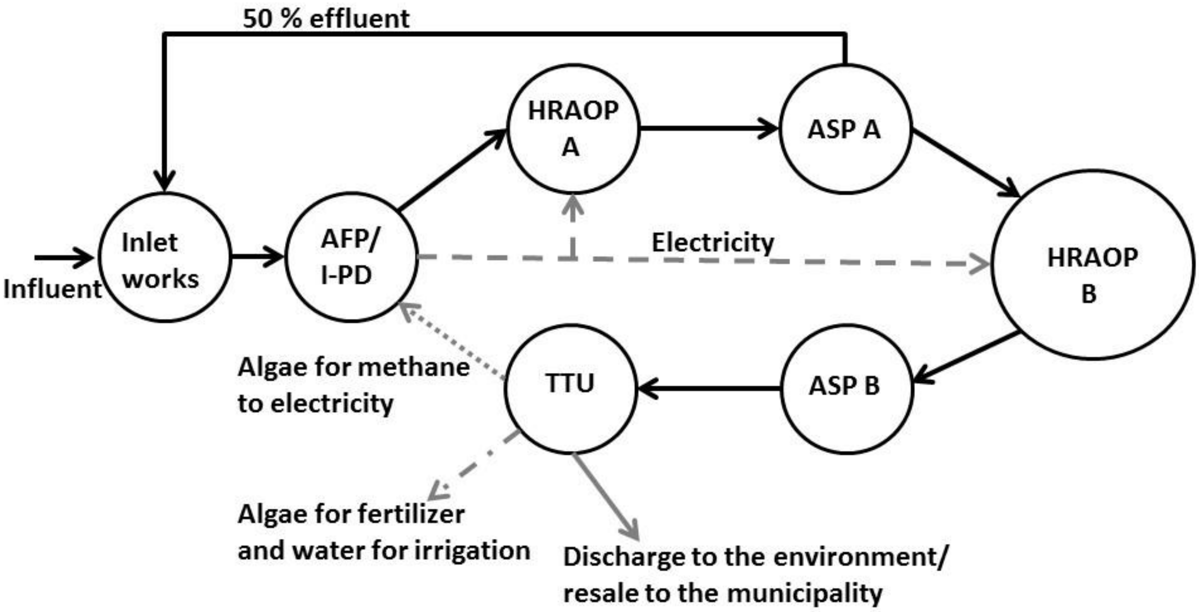


Figure 12 Process flow proposing how to potentially rectify shortcomings identified of the IAPS pilot. Ideally the influent will initially be diluted at the Inlet works then channeled to the AFP: Advanced Facultative Pond containing the I-PD: In-Pond Digester. This water will then be channeled to HRAOP A: High Rate Algae Oxidation Pond A, then sent to ASP A: Advanced Settling Pond A which will divert 50 % water to the Inlet works, while the rest is channeled to HRAOP B, double its current size. This water will then decant to ASP B following which the water will undergo tertiary treatment in the TTU: Tertiary Treatment Unit such as an MP: Maturation Pond, SSF: Slow Sand Filter, CRF: Controlled Rock filter or a rotating drum filter system. Potential benefits of the system include clean water for resale to the municipality or discharge to the environment, water for irrigation, algae for fertilizer and algae for methane generation for eventual electricity production.

Fourth, shifts in biocatalyst dominance were not only expected but observed. Biocatalysts shifts were observed when the dominant biocatalyst *Pediastrum* sp. in February 2012, was replaced by *Micractinium* sp. in October 2012, which was replaced by *Pediastrum* sp. in December 2012, which was then replaced by the unicellular diatom *Cyclotella* sp. in March

2013, which was replaced by *Pediastrum* sp. in December 2013, and then by *Dictyosphaerium* sp. in January 2014. The change in dominance from *Pediastrum* sp. to *Cyclotella* sp. in December 2013 is shown in Figure 13. Zhou *et al.* (2014) proposed that algae sensitivity to variations in nutrient profiles, at times deficiency of specific nutrients and/or trace elements, accompanied by the presence of inhibiting/toxic compounds and other competing algae in the system, may limit the number of strains able to adequately and consistently remediate municipal wastewater which may account for the intermittent shift in strain dominance presented in Figure 8. Strain dominance impacts settling efficiency in the ASPs and algae tend to be responsible for elevated dissolved organic C in secondary treated effluents from WWTW (Jeong *et al*. 2015, Green *et al.* 1995).

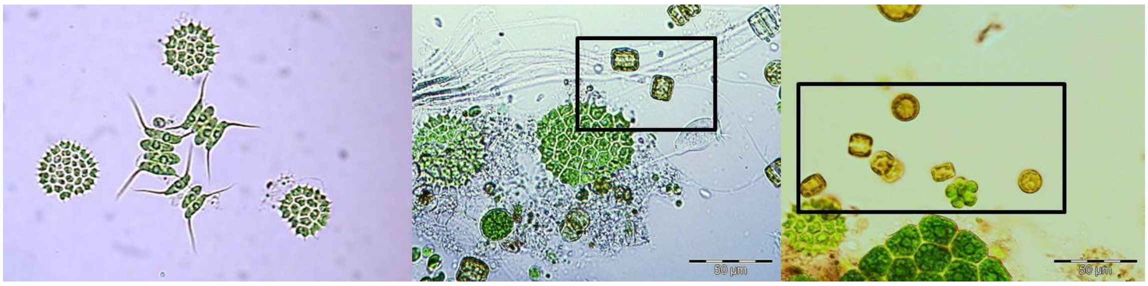


Figure 13 Changes in algae strain dominance observed from December 2012 to November 2013.

These concerns emphasize how contemporary practice can and does differ from prescribed procedure. In accordance with the Department of Public Works Design Guidelines for Small WWTW (2012), technological conformation is required where, in instances of failure to comply, accountability for failures must be carried out. These guidelines allow for troubleshooting to modify and adjust a technology to ensure compliance for future installations to render a technology suitably optimized for deployment in South Africa.

South African legislature mandates WWT technologies abide by prerequisites that ensure minimal current and future detrimental environmental impact. Predicated on its status as a scarce and national resource, both policy and management strategies for water tend to emphasize equal distribution and provision of basic services to all according to the Water Services Act (Republic of South Africa *Water Services Act* 1998). By default, government has to ensure the provision of sanitation to all. Thus, when sustainable strategies are implemented there should be a reduction in the amount of money required for the provision of clean water to an entire population, without sacrificing reliability and efficiency (Katukiza *et al.* 2012). Many policies mandate sustainable development and encourage water recycle and reuse where possible specifically the Integrated Pollution and Waste Management Act White Paper (Republic of South Africa *White Paper* 2000). Sustainable practices in developing countries should by right support peri-urban agriculture, reduce consumption of as yet unexploited water reserves and enhance livelihoods of affected communities (Wang *et al*. 2012). IAPS, if deployed correctly, offers an array of benefits to satisfy this niche. However the poor effluent quality, attributed to absence of an inlet works, lack of tertiary treatment and chlorination of the outflow, elevated influent BOD loading, and insufficient capacity of the HRAOPs to polish 100 % of the water from the ASP A, would seem to suggest that IAPS will continue to be viewed with skepticism.

Factors beyond control through pilot design and operation include variations in biocatalyst dominance, population dynamics and removal of the accumulated biomass after treatment. Dominance by one biocatalyst over another less desirable strain influences the performance of HRAOPs and ASPs leading to insufficient or poor settling of algae flocs. Unfortunately, over reliance on natural bioflocculation to remove algae biomass from the final effluent is insufficient to meet DWS (2013) regulations. Thus the status quo of the technology is that treated water contains elevated levels of TSS and COD. This can be improved by utilizing other more efficient technologies such as rotating drum filtration or MP series. The potential for downstream value adding is immense with minimal detrimental environmental impact. Table 12 presents a summary and quantitation of the major criteria for the ideal performance of IAPS as a WWT technology and a comparison between current and ideal practices. Department of Public Works Design Guidelines for Small WWTW (2012) requires a facility to accommodate the needs of the current population while estimating the needs of any future growth. To an extent design considerations need to include population shifts due to day labour, workers returning from employment in other areas, vacation migration to and from the area, stormwater flow and at times flooding. Construction of the facility needs to be solid, able to withstand environmental elements and require minimal maintenance over an extended period of time (González *et al.* 2012, Wallis *et al.* 2008). Generally a major disadvantage of pond technologies is the large areas required in comparison to electrochemical treatment systems and reverse osmosis plants (Craggs *et al.* 2012). However this is not yet applicable in South Africa as land is not limiting and a majority of the population are unable to settle the costs.

Table 12 A comprehensive summary of a gaps analysis of the IAPS.

Primary Factors	Secondary	Current Practices	Grade	The Gap	Best Practices in Future	Grade
Status quo of technology the	Water Quality	Not suitable for environmental discharge	5	Additional treatment needed	Suitable for discharge to a water course	7
	Sludge Handling	Faecal sludge -None	6		Faecal sludge - None	9
		Algae biomass discarded onto DBs	1	Algae biomass not valorised	Algae sludge digested to methane production and/or fertilizer production	8
	Operation	Spares not available on site	1	Insufficient equipment to support the system	Replacement paddles on site and spares	7
	Energy	Fossil fuel derived electricity used	1	Lack of knowledge for conversion of methane to electricity	Energy derived from the system	7
	Greenhouse gas emission	Vented to the atmosphere	2	Absence of gas capture equipment	Harvest biogas	8
	Skills	Minimal	8	No resident skill set	Minimal	8
	Knowledge	Advertising	7	Adoption	Universal acceptance and deployment of the technology	8
IAPS Process	Inlet works	Absent	1	Must be included in commercial design	Present legally	7
	Primary treatment	AFP/I-PD	5	Separate AFP and I-PD for inspection and maintenance	Separate AFP and I-PD	7
	Secondary treatment	HRAOPs	6	More HRAOPs/Increase size	Increase HRAOPs size/number	9
	Biomass removal	ASP	2	Adequate dewatering	Efficient ASPs/Drum filtration	8
Additional Treatment	Disinfection Final Polishing	None	1	Chlorination or UV light	MP, CRF system, drum filter, reverse osmosis, DAF	7

Biocatalyst	Varies	Determined by environmental influences *Pediastrum* sp., *Chlorella* sp., *Micractinium* sp. and *Scenedesmus* sp. result in optimal performance	5	Erratic species dynamics may negatively influence effluent quality *Cyclotella* sp. was observed to reduce the effluent quality	Recirculation of effluent from the raceways will ensure most ideal species dominate	7
Wastewater	Flow	Even low flow	7	Centralized wastewater treatment plants	Decentralization of facilities will ensure that an even low flow for maximum efficiency dominates	8
	Organic/hydraulic load	Organic load not within design specifications	1	Monitor organic loading	Organic load according to design	8
Features	Costs	Costs to deploy this full scale system unknown	1	Deployment of a full scale IAPS WWTW	Commercial scale fully costed IAPS example	9
Technical	Skills	Minimal	8	No resident skill set	Low	8

Figure 14 presents a summation of the criteria presented in Table 12. The comparative summative assessment between ideal and current practice clearly shows an underutilized technology with many benefits that have the potential to positively impact the South African water sector. Furthermore, these data highlight the need for a commercial scale IAPS to assess the full potential of this WWT technology.

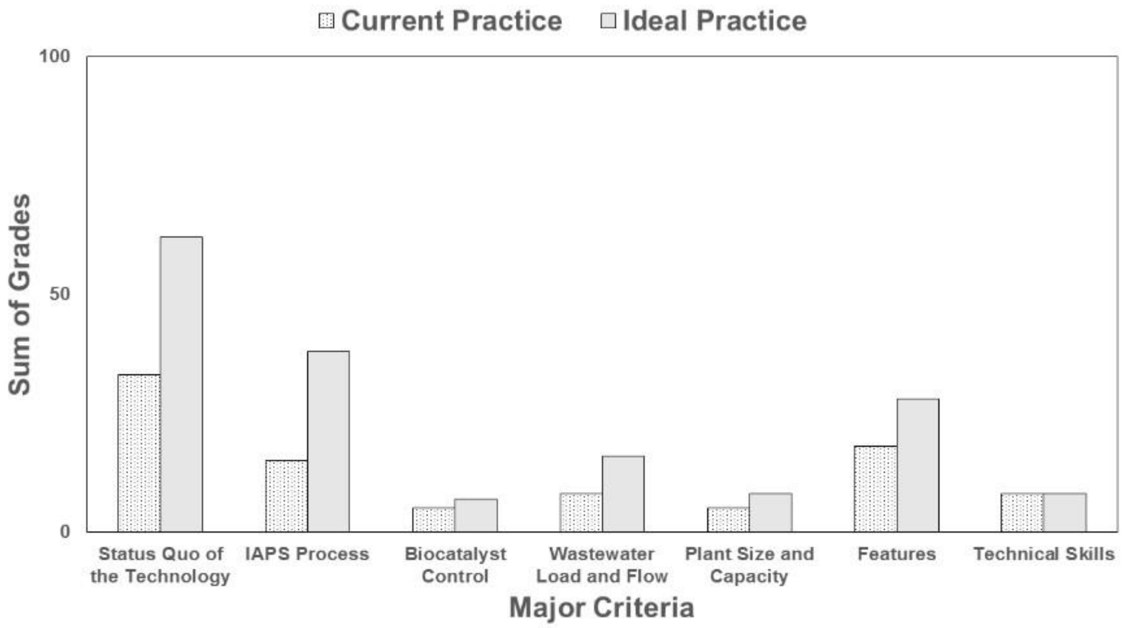


Figure 14 Grades in distinct major criteria. The grades were awarded according to current practices and most ideal practice.

Optimal environmental conditions for IAPS operation include light, space and warm temperatures, conditions indigenous to South Africa. Though potentially universally applicable, unlike AS and TF systems which can be deployed as package plants, IAPS must be adapted and optimized uniquely to suit the environmental conditions and the community needs where the system is to be deployed. Benefits of IAPS technology include: no faecal sludge handling; codigestion of the algae biomass with wastewater to produce methane (Tijjani 2011) and/or beneficiation of the accumulated biomass into saleable commodity products. Thus, not only is a desirable water quality possible but components of the system, namely the I-PD and the HRAOPs may in turn be utilized for energy and biomass production respectively.

In many respects, IAPS is a malleable and underutilized WWT technology. Underutilized because it can be a standalone WWTP or it can be fitted to pre-existing WWTW to improve the performance of a malfunctioning and/or overloaded facility. Armed with passive green technologies like IAPS, South Africa has the potential to lead the world in terms of sustainable development. Optimally unlike AS and TF, following community adoption the system can be operated and maintained off the municipal grid as a decentralized independent WWTW. The Belmont Valley IAPS pilot was originally designed for this purpose, specifically to be simple, robust, and sustainable and to remediate domestic wastewater while providing a platform for employment creation, skills development and income generation (Rose *et al.* 1996).

4 Conclusions

The pilot scale Belmont Valley IAPS, consistent with the WWT needs at the time of commission in 1996, in the absence of a TTU generates a treated water of secondary quality. Operating conditions for this pilot differ from the original design which has negatively affected overall performance and yielded less than satisfactory water quality data. This system does not have a TTU, or an inlet works and the total HRAOP capacity is insufficient to completely remediate all the water according to system design specifically the original 75 $k\ell.d^{-1}$ system design.

Nevertheless, two periods of assessment of this pilot indicated potential to produce treated water compliant with the general standard for both discharge to a water resource and irrigation. Indeed, studies on a range of TTUs have revealed that using an MP series, SSF, or a CRF ensure compliance on the final treated water from the Belmont Valley IAPS, rendering the outflow of a quality suitable for either irrigation or discharge to the environment with CRF> SSF> MP series (Westensee 2015 MSc, Mambo *et al*. 2014a).

The primary cause of poor water quality data appears to be persitent COD and TSS in the treated water due largely to an incorrect operating configuration. As IAPS technology is predominantly passive and biological, microbial debris may also contribute to elevated COD and TSS, specifically due to algae cell death and/or changes in biocatalyst population dynamics. Comprehensive follow up assessment of the delinquent COD revealed that it was soluble and comprised predominantly carbohydrates and humic acid-like substance strongly indicative of a biological origin.

Carbohydrates and humic acid-like substances are relatively stable C compounds and are not expected to have a detrimental impact on the environment. Furthermore, recirculating and selecting for desired algae strains that form readily settleable flocs should be practised routinely to optimize bioflocculation. It should also be emphasized that AIWPS deployments implemented on a commercial scale in California, U.S.A. include a tertiary treatment step, in the form of DAF, MP, UV light disinfection or multi media filtration (MMF). Thus the performance of the incomplete pilot IAPS is almost expected to be substandard, however this should not dissuade stakeholders from the technology as a whole.

Complete exploitation of the IAPS could have far reaching implications for South Africa, especially when biomethane is successfully harvested which converts to a potential 1.6×10^{11} $J.yr^{-1}$. Though current system functionality is dependent on electricity from the municipality, the IAPS is capable of generating enough electricity from methane conversion for all its current electricity needs. By operating off the grid this would ensure the system could be deployed in more remote locales. Improved access to electricity, clean water and sanitation will reduce current pressure on the health care sector, with fewer people falling ill. Cheap/free electricity could foster *in situ* development and *in situ* community accountability. Thus following the comprehensive assessment of IAPS as a WWT technology in the South African context, the study can conclude that this is an underutilized multifaceted technology with the potential to substantially benefit the South African water sector and boost the DWS Strategic Plan 2013-2017. Given the R 670 billion required over the next 10 years by the South African water sector for infrastructure refurbishment, maintenance and development (DWS 2012), findings thus far indicate IAPS as a viable sustainable WWT option capable for not only treating wastewater but also generating revenue.

4.1 Future Recommendations

IAPS as a WWT technology has been operated at pilot scale for nearly two decades in South Africa and to date, no demonstration/commercial scale systems have been implemented, presumably due to perceived performance inefficiencies. This thesis has highlighted factors that might have contributed to poor water quality data and it is unlikely that continued reference to the Belmont Valley pilot scale IAPS will change the status quo. With due consideration to influent loading, population size and projected growth, a commercial scale IAPS comprising an inlet works, sufficient HRAOP capacity, and an appropriate TTU is required with which to optimise this WWT technology for South African conditions. Thus implementation of a commercial scale plant is the next logical progression for the technology.

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6 APPENDIX

6.1 The Belmont Valley integrated algae pond system in retrospect

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The Belmont Valley integrated algae pond system in retrospect

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ABSTRACT

IAPSs (IAPS) are a derivation of the Oswald-designed Algal Integrated Wastewater Pond Systems (AIWPS) and combine the use of anaerobic and aerobic bioprocesses to effect sewage treatment. IAPS technology was introduced to South Africa in 1996 and a pilot plant designed and commissioned at the Belmont Valley WWTW in Grahamstown. The system has been in continual use since implementation, and affords secondarily treated water for reclamation according to its design specifications, which most closely resemble those of the AIWPS Advanced Secondary Process. In this paper IAPS as a municipal sewage treatment technology is re-examined in relation to design and operation, the underpinning biochemistry of nutrient removal by algae is described, and a retrospective is provided on the demonstration system at the Belmont Valley WWTW. In addition to presenting details of the process flow, several shortcomings and/or oversights are highlighted and, in particular, the need for an appropriate tertiary treatment component. However, despite the use of IAPS for sewage treatment in many countries, this technology is still viewed with some scepticism. Thus, a major purpose of this overview is to provide a synthesis of available information on IAPS and an appraisal of its use for municipal sewage treatment.

Keywords: advanced integrated wastewater pond system, IAPSs, wastewater, algae, nutrient removal, sewage

INTRODUCTION

Municipal sewage is an anthropogenically contaminated water body or stream which varies significantly depending on its origin and reaction to environmental influences, chiefly rainfall and evaporation (Adewumi et al., 2010). Rainfall dilutes the effluent and evaporation has a concentrating effect (Adewumi et al., 2010; Ahmad et al., 2011). Origins of municipal wastewater may be inclusive of, but not limited to, households, industry and agriculture (Bdour et al., 2009) and its source directly impacts its composition. However, factors such as social behaviour, economics, type and number of industries, area, climate, water consumption and the type and condition of the sewer system all contribute significantly to sewage composition (Sonune and Ghate, 2004; Su et al., 2012; Travis et al., 2012). Municipal sewage may contain contaminants such as plastics, rags, plant debris, pathogenic bacteria, fats, greases, nitrates, phosphates, heavy metals, and other potentially hazardous compounds (Sonune and Ghate., 2004; Ansa et al., 2012). Unless removed or rendered harmless in the WWT process these can adversely affect the environment. Thus, any remedial process must achieve an appropriate concentration of minerals and nutrients to avoid any acute or gradual influx into the environment of xenobiotics and toxic compounds (Lettinga, 1996; Debelius et al., 2009; Sekomo et al., 2012). The South African Government, through the Department of Water Affairs (DWA) has therefore mandated the remediation of all effluent prior to discharge to the environment to ensure that effluent streams released by municipalities (and industries) comply and will not be detrimental and/or damaging to the environment.

Innovation and advancement in the sector have proliferated wastewater treatment works (WWTW) and new process technologies are regularly made available as strategies to improve the management and remediation of wastewater (Bdour et al., 2009). Even so, management of WWT and control of final effluent quality/discharge is complex and some of the associated challenges include land, capacity, operations, maintenance and repair, technology developments, climate change, water course accessibility, and sustainability (Muga and Mihelcic, 2008; Gravelet-Blondin et al., 1997). These, coupled with available financial resources, directly impact wastewater infrastructure by influencing design, construction, operation, inspection, maintenance, and the overall efficiency of the WWTW (Korf et al., 1996). Since WWT is not a free-market enterprise in South Africa, acceptable process technologies are viewed by many as those that are either already optimised or can be immediately optimised, and without consideration of additional energy and monetary costs.

Wastewater treatment technologies currently deployed in South Africa for the treatment of municipal sewage include waste stabilisation ponds (WS) or oxidation ponds (OP), activated sludge plants (AS), bio-filtration (BF), biological nutrient removal (BNR), constructed wetlands (CW), and more (Adewumi et al., 2010; Oller et al., 2011; Tomar and Suthar, 2011). South Africa has approximately 970 municipal WWTWs which together treat an effluent stream of 7 589 000 $k\ell \cdot d^{-1}$ at an operational cost in excess of ZAR3.5 billion per year. Regional distribution of these WWTW according to size shows differences between the nine provinces:

- Gauteng Province has a relatively high number of medium (2–10 $M\ell \cdot d^{-1}$) and large (10–25 $M\ell \cdot d^{-1}$) WWTWs, with fewer micro (<0.5 $M\ell \cdot d^{-1}$) and small size (0.5–2 $M\ell \cdot d^{-1}$) plants.
- Eastern Cape, Northern Cape, Mpumalanga and Limpopo provinces mainly have micro-size and small size plants.
- North West, KwaZulu-Natal and Free State have a wider spread of WWTWs across all the plant size categories.
- Western Cape has a spread of WWTW sizes similar to the national situation.

Thus more than 80% of municipal WWTWs in South Africa treat less than 10 $M\ell \cdot d^{-1}$ and more than 50% of all WWTWs are micro-sized, while the preferred technologies are WS and AS at 41% and 35% respectively. Due to a paucity of information it is not possible to determine what proportion of the estimated total effluent stream (viz. 7 589 000 $k\ell \cdot d^{-1}$) is treated by each technology. Suffice it to say, together with bio-filtration at 16%, the range of WWT technologies commissioned by municipalities in South Africa is particularly narrow but distinctly biological.

Biological remediation of wastewater has for many years generally been favoured over conventional treatment techniques, even in light of the major limitation which is sensitivity to toxic components (Korf et al., 1996). Contemporary evaluation seems to share this opinion and is based largely on the costs involved in the construction and maintenance of biological treatment facilities (Cisneros et al., 2011). Toxicity, while a potential hazard to the microbial biocatalysts used in wastewater treatment, may be attributed to content and composition and factors such as shifts in pH and temperature (Muga and Mihelcic, 2008; Chan et al., 2009). Typically, it is maintenance of optimum activity of the biocatalysts that completely degrades organic pollutants (Gori et al., 2011; Mo and Zang, 2012; Daelman et al., 2012) and effects mineral and nutrient removal to yield a treated effluent that can be discharged regardless of shock loads (Gori et al., 2011; Rodriguez-Garcia et al., 2012).

The primary goal of the National Waste Management Strategy (NWMS) is the achievement of the objectives of the Waste Act [Act No 59 of 2008] (Republic of South Africa, 2008), which are, in summary: (i) minimising pollution, environmental degradation and the consumption of natural resources, (ii) implementing the waste hierarchy, (iii) balancing the need for ecologically sustainable development with economic and social development, and (iv) promoting universal and affordable waste services (NWMS). Framed within the context of the overall goals, approach and regulatory model of the NWMS, the introduction of a new WWT technology requires demonstration of proficiency, education, and increased awareness amongst all stakeholders including the public at large, the three spheres of government, and the private sector. The South African Government, parastatal and non-governmental organisations and citizens share a common concern regarding the national crisis relating to small and medium municipal WWTW, many of which are currently in a state of disrepair and are blamed for disease outbreak and infant mortality. A skills shortage, apparent lack of will to address these issues due mainly to the high costs of infrastructure repair and upgrade, and poor technology choices have not helped the situation. While any and all exposure and attention to this problem is real, there is the risk that efforts to mitigate the crisis will sow seeds for a new one through inappropriate or unsustainable technology choices. Population growth and migration patterns, financial constraints at local government level, water shortages in many areas, the shortage and cost of skilled personnel and the cost of electricity, among others, all challenge this choice. There are alternative technologies including algal ponding systems (Horjus et al., 2010). But are these being adapted for adoption in a changing South African scenario? (Laxton, 2010).

This paper presents an overview of Algal Integrated Wastewater Pond Systems (AIWPS)/IAPSs (IAPS) as a municipal sewage treatment technology, a retrospective on its introduction into the South African water sector, and a summary of current knowledge about this bioprocess technology.

IAPS as a bioprocess technology for wastewater treatment

Sewage treatment typically comprises 5 distinct phases. Primary treatment involves removal of suspended solids. Removal of dissolved biodegradable organic matter is a secondary treatment that reduces BOD to a level sufficient to prevent oxygen depletion of the water body into which the effluent flows. Nitrogen and phosphorus are removed by tertiary treatment to minimise growth of algae and other aquatic plants. Removal of refractory organic compounds is achieved by quaternary treatment while quinary treatment removes dissolved organics and salts including heavy metals. Successful waste treatment technologies should be sustainable, support peri-urban primary industry such as agriculture, prevent exploitation of water reserves and other resources, and enhance the quality of life of the community (Wang et al., 2012). Wastewater treatment must be biologically/mechanically rigorous, ecologically sound and environmentally friendly (Golueke and Oswald, 1963; Oswald, 1991, 1995) and the WWTW solid, able to withstand the elements and require minimal maintenance over an extended period of time (Wallis et al., 2008; González et al., 2012). Thus, for an implemented technology to be considered a sustainable process its use should, over the medium- to long-term, lower the overall cost without sacrificing reliability and efficiency (Katukiza et al., 2012).

Waste stabilisation (WS) ponds are a technology used prolifically by South African municipalities. As stated by Oswald (1995) 'The greatest advantages of ponds are their simplicity, economy, and reliability; their greatest drawbacks are their high land use, their potential for odour, and their tendency to eutrophy and fill in with sludge and to become less effective with age.' Research to maintain the advantages of WS ponds while mitigating the drawbacks resulted in the innovation known as the AIWPS (Oswald et al., 1957), which is still utilised globally for the remediation of domestic wastewater (Oswald, 1995; Green et al., 1995b; Craggs et al., 1996b; Craggs, 2005; Park et al., 2011a).

IAPS as a wastewater technology is a derivation of the AIWPS. Focus was initially on the symbiotic relationship between algae and bacteria in wastewater treatment (Oswald et al., 1955). Later, the term photosynthetic oxygenation was coined (Oswald et al., 1957) and used to describe the aeration effect caused by algal activity on treated wastewater (Ludwig et al., 1951, 1952; Oswald et al., 1953b, 1955). By 1957, Oswald had established the High Rate Algae Oxidation Pond (HRAOP). This algae-containing raceway amalgamated wastewater remediation via biological oxygenation and nutrient removal and led eventually to the fully developed bioprocess system (Oswald et al., 1957).

The system is designed to passively and biologically remediate domestic wastewater (Oswald, 1991). Based on studies at St Helena and Hollister in California, Oswald concluded that when properly designed systems were not only economical and effective but attractive and problem free. Moreover, AIWPS was deemed to provide for adequate, simple, and reliable wastewater treatment and afford communities opportunities for reclamation and environmental enhancement.

There are several versions of AIWPS and these have been categorised into first-, second- and third-generation processes depending on the quality of the final effluent required (Green, 1996). First-generation systems remediate domestic wastewater to a standard suitable for discharge to the environment whereas second- and third-generation AIWPS are self-sustaining and allow for the harvesting of methane, reclamation of water and in some cases harvesting of algae biomass (Green, 1996).

Like AIWPS, the IAPS relies on the combined activity of methane fermentation and photosynthetic oxygenation by algae coupled with biological oxidation in the high-rate ponds to remediate domestic wastewater. Furthermore, it uses gravity, solar energy and biological activity to treat wastewater (Downing et al., 2002) by processes that efficiently exploit the natural functionality of anaerobic, facultative and aerobic microorganisms within the system (Oswald, 1995; Craggs et al., 1996a), which comprises an in-pond digester (IPD), an advanced facultative pond (AFP), high-rate algae oxidation ponds (HRAOP), algae settling ponds (ASP) and maturation ponds (MP). A high-quality tertiary treated water can be reclaimed following filtration and UV sterilisation. As stated by Oswald (1991) 'when properly designed in appropriate locations, the systems virtually eliminate sludge disposal, minimise power use, require less land than conventional ponds, and are much more reliable and economical than mechanical systems of equal capacity'. Thus, IAPS is not just an adaptation of traditional pond systems but a design incorporating a series of low-cost reactors. No sludge management is required and the time in which sludge residues accumulate to require removal and disposal is of the order of decades. Carbon is transformed through two important mechanisms: methane fermentation and biological assimilation by microalgae. The conversion of waste organic solids to methane, nitrogen gas and carbon dioxide via methane fermentation and the assimilation of organic and inorganic carbon into algal biomass via photosynthesis provide the basis for primary, secondary, and tertiary treatment (Green et al., 1995a).

IAPS design and operation

Primary treatment takes place in an advanced facultative pond (AFP) which houses the in-pond anaerobic digester (IPD). The IPD is the point of entry of raw wastewater into the system and is responsible for the anaerobic decomposition of organic matter (Oswald, 1995). A coupled IPD/AFP promotes the deposition of organic material from suspension to facilitate decomposition at the base of the pond (Oswald, 1995). After 30 years of operation in the United States, sludge removal from the fermentation pits has yet to be conducted (Green et al., 1996; Daelman et al., 2012; Katukiza et al., 2012). The AFP is designed to reduce the BOD significantly and buffer the effluent prior to transfer to high-rate algae oxidation ponds (HRAOPs), while the aerobic surface layer of the pond neutralises odour-causing compounds, e.g., hydrogen sulphide (Lettinga, 1996; Oswald, 1995; Green et al., 1996; Muga and Mihelcic, 2008).

Secondary treatment is carried out in HRAOPs operated in series in which nutrients are extracted by a rapidly growing naturally occurring algae biomass. Algae photosynthesis directly supplies aerobic heterotrophic bacteria with oxygen, while the bacteria in turn oxidise recalcitrant material to increase the nutrient load in solution. It is the assimilation of waterborne nutrients such as nitrate, ammonium and phosphates, together with photosynthetic carbon reduction, that drives algae growth and development. Typically, HRAOPs are 0.1–0.5 m deep and the entire water column is oxygenated by both algae photosynthesis and paddlewheel mixing. The paddlewheel pumps water at a specific linear velocity and the action of the paddles on the water surface causes sufficient turbulence to allow for the introduction of oxygen, and CO_2, from the outside air. Total oxygenation capacity of the pond and the installed power of the paddlewheel give an oxygenation efficiency of 15 $kgO_2 \cdot kWh^{-1}$, which is a factor of 10 better than most mechanical aerators (Oswald, 1988; 1991). Thus, oxygenation capacity of HRAOPs can be contrasted with mechanical aerators which rarely transfer oxygen from air to water at more than 1 $kg \cdot kWh^{-1}$, indicating that photosynthetic oxygenation is 10–100 times more efficient as all energy is solar derived.

Although CO_2 availability within wastewater treatment HRAOPs depends primarily on the heterotrophic oxidation of organic compounds by bacteria (Weissman and Goebel, 1987; Oswald, 1988; Craggs, 2005), domestic sewage typically contains insufficient carbon to fully support optimal algal production (3–7 C:N ratio in sewage versus 6–15 C:N in algal biomass) (Benemann et al., 2003). Recently it was shown that addition of CO_2 to wastewater HRAOPs enhanced algal productivity by at least 30% (Park and Craggs, 2011) and reduced nitrogen loss by ammonia volatilisation providing more nitrogen for recovery by assimilation into biomass (Park and Craggs, 2011). Even so, unicellular green algae and cyanobacteria cultivated in ambient air levels of CO_2 develop a dissolved inorganic carbon (DIC) concentrating mechanism (also called a CO_2 concentrating mechanism or CCM) which is suppressed when cultured at elevated CO_2 and inhibited by O_2 (Ghoshal and Goyal, 2001). Similarly, denitrification and dissimilation, which converts nitrate to nitrogen gas, only occur in the absence of oxygen (Mitchell, 1974). In short, oxygen decreases the denitrification rate even if denitrifiers possess aerobic denitrification ability (Patureau et al., 1996). It might therefore be expected that the very high oxygenation capacity of HRAOPs would limit both growth of algae and denitrification. Although higher dissolved oxygen does favour nitrification, denitrification (and nitrification) rates increase with increasing temperature and the diel (i.e. during the adjoining dark period) loss of nitrogen via denitrification for algae ponds appears to be 15–25% of total influent nitrogen (Zimmo et al., 2004).

For growth, the mechanism by which inorganic carbon species are taken up by algae involves the light-induced drawdown of inorganic carbon by photosynthetic carbon reduction which maintains a concentration gradient between the external medium and the active site of the primary photosynthetic enzyme, ribulose bisphosphate carboxylase oxygenase (Raven and Hurd, 2012). CO_2 reacts in water and equilibrium is established between CO_2 and carbonic acid (H_2CO_3). The conversion of CO_2 to H_2CO_3 is kinetically slow and at equilibrium only a fraction of CO_2 exists as H_2CO_3 with most remaining as solvated molecular CO_2. Carbonic acid dissociates in water in two steps to produce carbonate anions as follows: $H_2CO_3 + H_2O \leftrightarrow H_3O^+ + HCO_3^-$ (pK_{a1} at 25 °C = 6.37) and; $HCO_3^- + H_2O \leftrightarrow H_3O^+ + CO_3^{2-}$ (pK_{a2} at 25 °C = 10.25). It is the formation of carbonate ions and their interaction with cations that leads to deposition of insoluble metal carbonates (e.g. $CaCO_3$; $MgCO_3$) and which provides an additional driving force (Lide, 2006). Consequently, net photosynthetic rate of an algae pond at optimal depth (0.3 m) and under optimal light and temperature is almost always constant at approximately 10 $t(C)\cdot ha^{-1}\cdot y^{-1}$ over the course of a day, because any increase in cell density, or decrease in photosynthetically active radiation, proportionately reduces the optimum pond depth and vice versa (Grobbelaar, 2007; Ritchie and Larkum, 2013). Even so, continual gravitation of effluent from the first HRAOP, via algae settling ponds (ASP), to the second HRAOP removes some of the accumulated algae biomass (and residual bacteria) to mitigate substantive changes in optimum pond depth thereby increasing nutrient abstraction efficiency (Oswald, 1995). In addition, sustained algae photosynthetic activity coupled with nitrification and nitrate consumption leads to an increase in medium pH. Most of the energy for nitrate assimilation arises from photosynthesis, photosynthesis is also reported to be responsible for light regulation of nitrate reductase gene expression and activity (Lillo et al., 1996; Oswald et al., 2001), and linear electron flow and generation of reducing equivalents are promoted by photosystem 1 (PS l) light absorption which is believed to facilitate reduction of assimilated nitrate alongside CO_2 (Sherameti et al., 2002). Thus, as long as nitrate is abstracted, reduced to ammonium and the ammonium assimilated into amino nitrogen, a 1:1 alkalinisation in relation to nitrate consumption is maintained (Ullrich and Novacky, 1990; Mistrik and Ullrich, 1996; Ullrich et al., 1998).

Alkalinisation in the HRAOPs has been suggested as a mechanism, separate from biological assimilation, to promote removal of phosphate in the form of an insoluble hydroxyapatite. Thus, elevated pH (>10) can stimulate not only ammonia-N removal from the HRAOP by ammonia volatilisation but phosphorus removal through phosphate precipitation with calcium, magnesium and non-chelated ferric iron (García et al., 2000; Craggs, 2005).

The above account on the biochemistry of nutrient abstraction and assimilation into biomass in HRAOPs has ignored, for the sake of brevity, some critical environmental (light and temperature), operational and other biological factors (zooplankton grazers and algal pathogens) that do impact wastewater treatment. However, this omission only serves to further strengthen the assertion by Oswald (1991) that correct design, locality and operation are paramount for successful implementation of this bioprocess technology. Secondary and tertiary treatment of wastewater can be fully accounted for by passage through a series of HRAOPs. Ideally suited to warm climates in which high BOD removal capacity is easily realised, these systems retain all of the advantages of WS ponds, and while land requirements are substantially more than needed for AS (not accounting for land used in sludge management), operational and capital costs have been estimated at half and one fifth, respectively, of those required for AS (Park et al., 2011b; Craggs et al., 2011).

The final reaction in the AIWPS/IAPS bioprocess is tertiary treatment, usually achieved in a series of maturation ponds (MP) or by filtration (e.g. slow sand filter). Maturation ponds hold secondary treated effluent and are typically positioned downstream from conventional treatment systems (Shillinglaw, 1977). The main function of maturation ponds is additional polishing of the water to clean and remove any residual pathogens carried forward from the secondary treatment process (Mara, 2005). Prevailing environmental conditions such as: high pH, low temperature, high dissolved oxygen (DO) and ultraviolet radiation are exploited (Von Sperling and De Lemos Chernicharo, 2005) and usually 2 or 3 ponds are constructed in series to provide the retention time (12 days) needed for adequate pathogen removal. Maturation ponds provide little or no biological stratification, have high algae diversity which increases further across a pond series, and tend to be fully oxygenated throughout the day providing ideal conditions for faecal coliform/pathogen removal (Mara, 2005). A high pH is found in maturation ponds which impacts faecal bacteria mortality (Von Sperling, 2007) and enhances nitrogen removal both by assimilation into biomass and loss via volatilisation (Kayombo et al., 2005). In fact, ammonia removal in maturation ponds exceeds that of other tertiary treatment processes (e.g. constructed wetland), and in a comparative assessment was second only to aerated rock filtration (Johnson et al., 2007).

Quaternary and/or quinary treatment are not typically components of AIWPS/IAPS wastewater treatment systems and will not be discussed here.

IAPS as a global wastewater treatment technology

During the past 6 decades IAPS have been successfully applied in America, Australia, Belgium, Brazil, Canada, China, Egypt, Ethiopia, France, Germany, India, Ireland, Israel, Italy, Kuwait, Mexico, Morocco, New Zealand, The Netherlands, Philippines, Portugal, Scotland, Singapore, South Africa, Spain, Switzerland, Thailand, Vietnam and Zimbabwe (Oswald, 1995). More recent deployments of this bioprocess technology are presented in Table 1. As a green technology, algae systems address imperative issues such as global warming and climate change (Wallis et al., 2008). Algae biomass generated by the remediation of wastewater serves as a carbon sink, thereby mitigating the negative effects of CO_2 as a greenhouse gas (Green et al., 1995a), which may be used to justify the use of algae ponds as a sustainable technology, economical and environmentally friendly, which alleviates pressure on environmental water reserves (Oswald, 1995). Furthermore, IAPS is a versatile and passive bioprocess that can be used to remediate, in addition to domestic sewage, brewery effluent, food processing waste, industrial effluent and abattoir waste (Rose et al., 1996; Boshoff et al., 2004; Van Hille et al., 1999).

Table 1

The Belmont Valley WWTW IAPS

Rhodes University commissioned and built an Oswald-designed version of the AIWPS, the IAPS, specifically for South African conditions. This pilot-scale demonstration is located at the Belmont Valley WWTW where it receives and treats a constant supply of raw domestic sewage extracted from a splitter box immediately after the inlet works. A process flow illustrating the configuration for operation of the Belmont Valley WWTW IAPS is presented in Fig. 1. The system has been in continuous operation since 1996 and receives 75 $m^3 \cdot d^{-1}$ of raw sewage. It is apparent from the schematic (Fig. 1) that any partially treated water and/or tertiary treated water (i.e. suitable for reclamation) is returned to the Belmont Valley WWTW. Thus, and due to research and development needs and various logistical issues, no effluent from this demonstration system is discharged to environment. While this IAPS is an operational, passive, sequential, sewage treatment facility that functions virtually in perpetuity and without any need for faecal sludge handling, the technology has yet to be adopted by the wastewater sector for implementation nationally. The reasons for the status quo are unclear, but in part may be due to ignorance about the technology, the perception that the final effluent generated does not comply with standards set by DWA, a perceived skills shortage, and an apparent lack of will to address sewage treatment management issues, due mainly to the high costs of infrastructure repair and upgrade. This is in direct contrast with global sentiment to IAPS/AIWPS technology which is currently in use in the USA, India, New Zealand and many other countries (Table 1)

Figure 1

Criteria for the Belmont Valley IAPS for the treatment of municipal sewage were as follows: capacity of 500 person equivalents (PE) – based on an average water consumption and disposal per capita of 150 $\ell \cdot d^{-1}$, the design flow was calculated at 75 $m^3 \cdot d^{-1}$. With an ultimate biochemical oxygen demand (BOD_{ult}) assumed to be 80 g BOD_{ult} per person per day, an organic loading to the system of 40 $kg \cdot d^{-1}$ was determined (Rose et al., 2002). It was postulated that the resultant treated effluent would comply with environmental discharge (Rose et al., 2007).

To eliminate the need for faecal sludge handling and disposal and to ensure complete breakdown of biodegradable solids, the volumetric capacity of the IPD was designed at 0.45 m^3 per capita, rather than the more conventional 0.3 m^3 per capita. Thus, raw sewage, after screening, enters at least 6 m below water level near the bottom of the IPD. A 'berm' wall (1.5 m below water level) extends the IPD 1.5 m above the floor of the AFP to direct gas flow and prevent any ingress of oxygen-rich water. An upflow velocity of 1.0–1.5 $m \cdot d^{-1}$ in the IPD was estimated as sufficient to allow solids to settle and parasites (e.g. helminth ova, worms, etc.) to remain in the sludge layer. With a volumetric capacity of 225 m^3 hydraulic retention time (HRT) in the IPD and at the designed flow is 3 d. By reducing influent flow rate or increasing IPD volume it is possible to manage digestion to near completion. The overlying water of the AFP contains an oxygen-rich layer near the surface which is populated by algae which sequester many gases produced as a consequence of anaerobic digestion. Even so, large emissions of CO_2 and CH_4 from the surface area can be expected and methane production rates of 0.17 kg $CH_4 \cdot kg^{-1}$ BOD_{waste} have been modelled for anaerobic ponds fed municipal waste (DeGarie et al., 2000; Van der Steen et al. 2003). Furthermore, about 3.3 $m^3 \cdot capita^{-1} \cdot y^{-1}$ of CH_4 is produced in real-scale ponds at a daily sewage production of 100 g $COD \cdot capita^{-1} \cdot d^{-1}$. While CO_2 is the best studied and most known, CH_4, which constitutes up to 75% of the total gas emitted during anaerobic wastewater treatment, is 25 times more potent as a greenhouse gas (Forster et al., 2007; Daelman et al., 2012). Similarly nitrous oxide, although emitted in relatively low amounts, is 300 times more damaging than CO_2 (Daelman et al., 2012; Strutt et al., 2008). While, harvesting and/or recycling of these gases can potentially avert any detrimental impact (Oswald, 1995; Green et al., 1995a), it is now understood that methane-oxidising bacteria (MOB), a relatively common group of bacteria capable of utilising CH_4 as their sole carbon and energy source if sufficient O_2 is present, utilise in-pond algae-derived O_2 to consume much of the emitted CH_4 before it reaches the atmosphere (Van der Ha et al., 2011). Even so, only part of the CH_4 produced by methanogens is consumed by MOB before it reaches the atmosphere and elementary extrapolation of measured pond emissions still show a total loss of about 3 000 $m^3 \cdot y^{-1}$ CH_4, equalling a yearly contribution of 55 ton CO_2-equivalents$\cdot y^{-1}$ or an emission of 0.98 $kg \cdot y^{-1}$ CH_4 per inhabitant.

For the AFP, HRT was determined using a temperature-dependent first-order decay rate for residual BOD (Rose et al., 2002) and for the Belmont Valley IAPS this is 20 days. Design of the outer reaches of the AFP was also important to limit, if not prevent, short-circuiting from the IPD overflow to the AFP outlet, which is typically 0.5–1.0 m below water surface to avoid skimming off of floating material and algae.

Effluent from the AFP is gravity fed to the first of 2 HRAOPs connected in series and with a combined HRT of 6 days, is subjected to photosynthetic oxygenation. High-rate algae oxidation ponds are shallow, paddlewheel-driven, continuously mixed raceways that promote algae growth. High photosynthetic rates elevate effluent pH (up to 11) and increase the dissolved oxygen concentration (DO; up to 3 times saturation). Excess oxygen is consumed by heterotrophic bacteria to degrade dissolved organic matter for assimilation by the algae biomass. At elevated pH other mechanisms also operate to reduce the nutrient load, e.g. phosphate precipitation and ammonia volatilisation.

Mixing, or turbulent flow, is essential to maintain optimum conditions for maximum production of microalgae. Apart from preventing thermal and oxygen stratification, paddlewheel mixing maintains the surface velocity required, keeping the algae and algal flocs suspended near the surface and within the sunlight penetration depth. A channel velocity of 50 $mm \cdot s^{-1}$ is sufficient to prevent algae settling and eliminate stratification but is very difficult to maintain due to frictional losses, especially in the bends. Thus, a linear velocity of 200–300 $mm \cdot s^{-1}$ is routinely used although this increases the energy demand. Power to drive the paddlewheels is a function of raceway length, wetted area, method of construction and channel velocity (note; friction increases as the square of the velocity increases). For raceway mixing a paddlewheel is, possibly by far, the most efficient means of consistently maintaining channel velocity. Paddlewheels are essentially pumps and as such provide the power to overcome the static head required to override frictional head loss in the raceway. Design and construction of the volute and the pump (paddlewheel) is thus very important. An 8-paddle configuration is sufficient to reduce shock on the drive and mounting assembly. Even flow velocity around the 180° bends is achieved using flow rectifiers. Various flow rectifiers have been tested, teardrops, reverse teardrops, etc., and the method determined as most successful is concentric semicircle walls spaced 1 m apart. Biomass produced in the HRAOPs must be removed prior to tertiary treatment or discharge of the effluent and this is achieved using algae settling ponds (ASPs). Algae in the effluent settle rapidly in a well-designed ASP and a HRT of 0.5 d in this pond is sufficient for adequate settling. Details of the design configuration and operation of this IAPS are described elsewhere (Rose et al., 2002, 2007).

The pilot-scale IAPS described above was commissioned at the Belmont Valley WWTW in Grahamstown in 1996, to demonstrate and evaluate the performance of the system for deployment as a green technology to address issues such as climate change (Rose et al., 2002) and sustainable development (Oswald et al., 1957). The costs associated with construction and operation (including maintenance) of the IAPS were, at the time, viewed as highly competitive (Rose et al., 2002). Job creation was evidently possible (Rose et al., 2002; Harun et al., 2010) while improved access to clean water was and still is understood to stimulate social and economic development (Oswald, 1995). An extended study to evaluate operation and performance of this IAPS as a full municipal sewage treatment system for South Africa was published in 2007 and revealed the following;

- The system did not achieve the 75 $mg \cdot \ell^{-1}$ discharge standard for CODt.
- Although a reduction in phosphate was observed, it was not within the 10 $mg \cdot \ell^{-1}$ required for discharge.
- Residual ammonia levels exceeded the 3 $mg \cdot \ell^{-1}$ discharge standard.
- Nitrate removal was at best erratic and at times nitrate concentration increased (Rose et al., 2007).

It is difficult to reference the data obtained from the IAPS at the Belmont Valley WWTW (Rose et al., 2007) against results for other systems due in part to incompleteness in mass balances and the apparent lack of empirical values to describe the nutrient load in both the raw sewage influent as well as the residual nutrient load in the final effluent (i.e. discharged from the final ASP) following treatment by operation of the full system. Also, these authors seemed more concerned with the performance of each of the component parts of the IAPS and little emphasis was placed on IAPS as a complete system for municipal sewage treatment. Consequently, much of the data is derived from operation of only a single HRAOP and attempts to develop this further as the ‗I-HRAP' for use as a tertiary treatment unit.

A further concern with the studies described by Rose et al. (2007) on the implementation and performance of the IAPS for sewage treatment in South Africa is the absence of a final polishing step. As discussed above, the original AIWPS was designed to always include a polishing step comprising of either a MP or similar which would allow the final effluent to meet the specifications for discharge as required by DWA, except for the presence of total coliforms which requires additional disinfection (e.g. chlorination, ozonation, UV-radiation, etc.). In fact, a recent report on the operation of hectare-scale HRAOPs for enhanced sewage treatment (Craggs et al., 2012) strongly advocates that additional treatment of the algae harvester effluent is required to meet specific discharge standards. These authors recommend the inclusion of one or a combination of MP and UV treatment by storage prior to discharge or rock filtration of the MP effluent or direct UV treatment if insufficient land is available, and, if funds are available, membrane filtration to achieve a high-quality final effluent for reuse. Without a final polishing step, and as demonstrated in other studies, the COD of the final effluent remains elevated resulting in the potential that, if discharged, water from an IAPS will be detrimental to any receiving water bodies (Park and Craggs, 2011). Thus, it is surprising that the model proposed by Rose et al. (2007) to link water treatment and job creation initiatives 'which is dependent on the system to produce a water quality that at least meets DWA irrigation water discharge standards' was based on a ‗secondary treated' water. Clearly, any considered implementation of IAPS technology for treatment of municipal sewage must include in the process design a final effluent polishing process.

Studies underway and funded by the Water Research Commission are currently addressing the above issues in detail. Furthermore, data on compliance of final water quality, following an 8-month extended study of the system for treatment of municipal sewage, has been completed. Results show that treated water from the IAPS is compliant with the discharge limits for phosphate, ammonium-N and nitrate/nitrite-N, and mean values were: 5.3 $mg \cdot \ell^{-1}$, 2.9 $mg \cdot \ell^{-1}$, and 12.4 $mg \cdot \ell^{-1}$, respectively. Chemical oxygen demand however fluctuated significantly and was dependent on full function of the IAPS. Mean COD of the final treated water was 72.2 $mg \cdot \ell^{-1}$. Although, these results suggest that the treated water discharged from this IAPS operating under South African conditions meets the standard for discharge, mean TSS was routinely above the limit at 34.5 ± 13 $mg \cdot \ell^{-1}$ and faecal coliforms were higher than expected. Tertiary treatment, either by a maturation pond series, slow sand filtration, or a controlled rock filter, ensured that the final treated water from IAPS met the standards with CRF>SSF>MP (Mambo et al., 2014).

Future of IAPS in South Africa

Despite the oversights and shortcomings alluded to already, IAPS is a technology that can be used to address some of the current challenges associated with sewage treatment and management in South Africa (Rose et al., 2002). Although there may be reservations around the quality of the final effluent from IAPS, current opinion is that these bioprocess systems are more cost-competitive to design, construct and operate than conventional sewage treatment technologies (Rose et al., 2002). Also, IAPS is a robust technology and is not energy demanding, thereby making sewage treatment sustainable and efficient (Oswald, 1995).

No chemical dosing is required for disinfection or sludge dewatering to improve the quality of the effluent generated by the IAPS (Oswald, 1988). Thus remediation of a secondary toxic sludge is not required (Lettinga, 1996) and the effluent generated can immediately be used for irrigation and aquaculture (Oswald, 1991). While IAPS can apparently be configured to generate a desired effluent quality depending on the degree of water treatment required (Rose et al., 2002), major disadvantages include the large land areas required in comparison to electrochemical treatment systems and reverse osmosis plants (Craggs et al., 2012), and the over-reliance on bioflocculation to remove algae from the final effluent. In addition, the higher than desirable COD can potentially be detrimental to receiving water bodies (Park and Craggs, 2011). These concerns aside, introduction of a more efficient system for separating out the algae (e.g. drum filtration, dissolved air flotation) and the implementation of a tertiary treatment step should allay any further scepticism and is easily achieved either by extending the facility at the Belmont Valley WWTW or by building a fully commercial system to allow for a thorough evaluation of IAPS as a sewage treatment technology under South African conditions.

CONCLUSION

The World Health Organisation estimated that in 2000 approximately 2.4 billion people did not have access to clean water and sanitation, which led to 1.7 million preventable deaths. Successful wastewater management reduces faecal-oral disease and environmental pollution caused by sewage (Cisneros, 2011). Furthermore, nearly 50% of the 7 billion people on Earth live in water-stressed countries and it is crucial therefore that sustainable water and wastewater management technologies are implemented and practised. In view of the ever-increasing cost of energy one candidate that deserves further scrutiny and consideration, at least for wastewater management, is IAPS.

Optimal conditions for an IAPS include light, sufficient land and warm temperatures. These conditions are in abundance in southern Africa. What remains is derivation of a comprehensive dataset which analyses and evaluates this bioprocess technology to provide authorities with the confidence needed when deciding on the implementation of an appropriate treatment technology. Satisfactory remediation of municipal sewage can reduce demand on potable water resources and reduce any detrimental anthropogenic imprint on the environment.

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Figure Headings

Figure 1

Schematic illustrating the process flow for the pilot IAPS designed, constructed and operational at the Belmont Valley WWTW, Grahamstown. The system receives 80–100 m^3 of raw sewage daily, after screening and passage through a grit/detritus channel (in duplicate – one operating, one cleaning). Effluent enters at the bottom of the AFP some 6 m below water level. AFP=Advanced Facultative Pond; IPD=In-Pond Digester; HRP=High Rate Pond; C/F=Coagulation/Flocculation; ASP=Algae Settling Pond; SB=Splitter Box; TTU=Tertiary Treatment Unit (e.g. maturation pond, slow sand filter)

Table 1
Examples of global deployment of IAPS since 2000

Country	Climate	Treatment	Performance	Deployed	Reference
China	Monsoon	HRAOP (×2)	Not Disclosed	2002	Chen et al., 2003
Morocco	Mediterranean	HRAOP (×2)	Not Disclosed	2003	El Hamouri et al., 2003
New Zealand	Temperate	HRAOP (×2)	100% faecal coliform disinfection	2001	Craggs et al., 2003
New Zealand	Temperate	HRAOP (×4)	82-91 % BOD, 64-67 % NH4,14-24 % P, >99 % coliform removal, pH 9.3	2010	Broekhuizen et al., 2012
Spain	Mediterranean	HRAOP (×2)	Not disclosed	2006	Garcia et al., 2006
Sweden	Cold	HRAOP (×2)	97 % BOD,64 % P, 90 % N removal	2003	Gröndlund et al., 2004

Figures

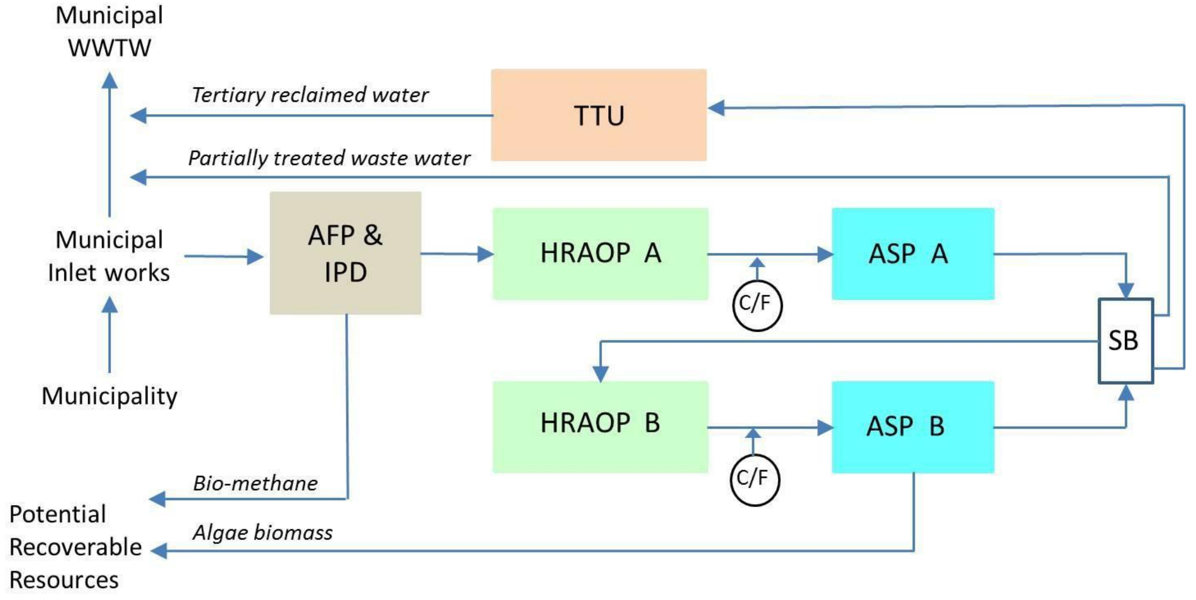

FIG 1

6.2 IAPS for Municipal Sewage Treatment: A Gap Analysis

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IAPSs for municipal sewage treatment: A gap analysis

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ABSTRACT
This study describes the results of a gap analysis used to evaluate the configuration and operation of IAPS for municipal sewage treatment. Both metadata and real time data derived from the pilot IAPS at the Belmont Valley WWTW were used in the analysis. Results show that the pilot plant receives an influent BOD of 80 $kg.d^{-1}$, which is two fold the original design specifications. Quality of water after treatment has elevated TSS and COD and as a consequence and does not comply with the limits for discharge to the environment. Recalculation of the kinetic parameters coupled with a gap analysis revealed that the system is incorrectly configured and that partially treated water from HRAOP A should be returned to the IAPS inlet to dilute influent BOD. Quantitiation of results from this gap analysis reveal a technology requiring full scale commercial plant to determine the benefit of this technology to the South African Water sector. Costing less than one third of conventional activated sludge systems and with potential for waste beneficiation and product generation, IAPS offers an avenue for community development, access to energy and a sustainable ‗green' technology.

KEY WORDS Gap analysis, IAPS

INTRODUCTION

Often described as a reliable and efficient WWT technology that outperforms traditional waste stabilization (WSP) and oxidation ponds (OP), municipal sewage treatment using IAPSs (IAPS) has remained relatively unexplored and full commercialization has not been realized (Meiring and Oellerman 1995; Craggs *et al.* 2004; Craggs *et al.* 2010). Although conceptualized and initially demonstrated on small scale approximately 60 years ago, IAPS have largely been deployed only at pilot scale. Nevertheless, several large scale (2-5 $ML.d^{-1}$) IAPS operate in the Central Valley of California (e.g. Dehli, Hilmar, and St Helena) and for each; the treated water is used solely for irrigation purposes. Meiring and Oellerman (1995) suggested that the elevated COD and TSS of the treated water from IAPS was a consequence of inadequate design and operation. This probably explains, in large part, why IAPS treated water is considered suitable for irrigation but not discharge to the environment. More recently, Craggs *et al*. (2003) confirmed the elevated TSS levels in effluent from an IAPS (64 $mg.L^{-1}$). In subsequent work, Craggs *et al*. (2004) showed TSS measurements remained elevated at 87 $mg.L^{-1}$. An extended study using a pilot scale IAPS operating under South African conditions confirmed that IAPS as a sewage treatment technology was incapable of producing an effluent suitable for discharge to the environment (Rose *et al.* 2007). Measurements showed COD_f >75 $mg.L^{-1}$, TSS levels >25 $mg.L^{-1}$, *ortho*phosphate >10 $mg.L^{-1}$ and residual ammonia-N levels were in excess of 3 $mg.L^{-1}$ while nitrate-N removal was at best erratic and at times, nitrate-N concentrations increased (Rose *et al.* 2007). To overcome these shortcomings Craggs *et al*. (2014) suggest that depending on the local environmental discharge requirements and standards, maturation ponds or other tertiary treatment processes should be used. Maturation ponds allow solar UV disinfection, polishing of the wastewater and effluent storage prior to discharge or irrigation, while the addition of rock filter systems reduce TSS levels. Where land is limiting, UV irradiation might be preferable for disinfection to ensure a good quality effluent stream. It is perhaps due to the abovementioned water quality issues, and in particular the unacceptable levels of TSS and COD, that IAPS as a municipal sewage treatment technology has not been widely accepted. Although there are no known commercially operated systems efforts by proponents of IAPS to have the technology constructed and implemented at full scale have not been dampened.

Several South African-based projects have been initiated including a UNEP WioLap sponsored IAPS (1 $ML.d^{-1}$, Bushman‘s River, Nlambe Municipality), two Partners-for-Water sponsored IAPS (2 $ML.d^{-1}$ Grahamstown, Makana Municipality; 1.5 $ML.d^{-1}$, Alice, Amathole Municipality), and the conversion of a WSP system to an IAPS (2-3 $ML.d^{-1}$, Bedford, Amathole Municipality). In each case the projects proceeded through the design stage but failed at implementation (A.K. Cowan, 2014, Personal Communication). Reasons for these failures though many and varied, were explored in a recent study by Nemadire (2011) who noted a failure of the IAPS to meet South African authorisation limits, different technology choices availing themselves i.e. activated sludge plants and delays due to conflicts with stakeholders.

Most recently, the Department of Science and Technology (through the Water Research Commission) awarded funds to Rhodes University to manage the design, construction and implementation of IAPS as a municipal sewage treatment technology. The project is a joint venture between the university and Chris Hani District Municipality. Construction of a >2 $ML.d^{-1}$ commercial scale demonstration system is planned for the town of Tarkastad. However, prior failure of four similar projects coupled with less than convincing water quality data suggests that due caution be exercised before proceeding with implementation of IAPS as a commercial scale WWT technology. In an effort to address concerns and technology weaknesses highlighted above a gap analysis of IAPS was undertaken. Gap analyses typically compare best practice with current processes to determine the –gaps‖ so that –best practice‖ is selected for implementation. In this study the configuration and operation of a generic IAPS was evaluated in terms of its component parts and water treatment efficacy. Both metadata and real time data derived from the pilot IAPS at the Belmont Valley WWTW were used to supplement the analysis.

METHODOLOGY

Metadata

Metadata of water quality analyses for the pilot IAPS at the Belmont Valley WWTW for the period 1999-2014 was obtained from the Institute for Environmental Biotechnology, Rhodes University (EBRU). All other data was obtained as previously described (Mambo *et al.* 2014a).

Gap analysis and criteria

For the purposes of the gap analysis, criteria were defined as either primary or secondary, where secondary criteria were grouped according to each primary criterion. Qualitative gap analysis was based on the set of primary criteria which included the status quo of IAPS technology, process flow or system configuration, plant size and capacity, biocatalyst, wastewater composition, technical skill, faecal sludge, effluent quality, technology features, environmental impact, operation, tertiary and quaternary treatment, and cost. Each primary factor provided the basis for comprehensively evaluating the technology through defining appropriate secondary criteria as specified in the Results. In addition, a quantitative assessment of the gap analysis was achieved by scoring each criterion on a scale from 1-10, where 10 represents the ideal.

Design and operation of IAPS

IAPS is an adaptation of traditional pond systems consisting of a series of low-cost ponds or earthwork reactors. A typical IAPS consists of a minimum of four ponds in series. These are; advanced facultative ponds, secondary facultative ponds or algal high rate ponds, algae settling ponds, and maturation ponds (Green *et al.* 1995). No sludge management is required and the time in which sludge residues accumulate to require removal and disposal is of the order of decades. Carbon is transformed in IAPS through two important mechanisms: methane fermentation and biological assimilation by microalgae. The conversion of waste organic solids to methane, nitrogen gas and carbon dioxide via methane fermentation and the assimilation of organic and inorganic carbon to algal biomass via photosynthesis provide the basis for primary, secondary, and tertiary treatment using IAPS technology (Oswald *et al.* 1994; Green *et al.* 1995) and schematics illustrating the process flow for each is shown in Figure 1.

Figure 1

The Water Research Commission (WRC) initiated a project in South Africa titled ‗Appropriate low-cost sewage treatment using the advanced algal high rate oxidation pond' - which commenced in 1994 with technology transfer, design and construction of an IAPS demonstration and research facility at the Environmental Biotechnology Experimental Field Station now, the Institute for Environmental Biotechnology, Rhodes University (EBRU). The first effluent entered the plant in February 1996. The rationale behind the project was to (re)-design the technology for South African operating conditions and at the same time demonstrate the technology and provide an engineering support base for the development of IAPS process applications in the treatment of different waste waters (Rose *et al.* 1996). This pilot scale IAPS continues to operate and treats 75 m^3 municipal sewage daily and the operating configuration is shown in Figure 2.

Following grit removal and screening by an inlet works the high BOD (COD)-containing wastewater is gravitated to an anaerobic digester (AD) where it is detained for at least 3 d. From the upflow sludge bed digester, water percolates into the advanced facultative pond (AFP) where it is detained for a minimum of 20 d before gravitating to the first of two high rate algae oxidation pond (HRAOP) operated in series. After 2 d retention in HRAOP A water is channelled into an advanced settling pond (ASP) with a HRT of 0.5 d from which settled biomass is harvested. Half of the water is then gravitated to the second raceway, HRAOP B where it is retained for 4 days before a second ASP (HRT=0.5 d) is used to settle the accumulated biomass. The remaining 50% of water from HRAOP A/ASP A is returned to the inlet works of the Belmont Valley WWTW. Currently biomass is harvested mannually and placed in drying beds (DB).

Figure 2

RESULTS

Organic Loading and Nutrient Removal

In order to test the kinetic parameters as reported for the Belmont Valley IAPS (Rose *et al.* 2002), the design criteria were evaluated using metadata obtained by analysing the raw sewage influent entering the system (Rose *et al.* 2007) and the results are shown in Table 1. Actual strength of the raw sewage as measured over the course of 2 years revealed a mean organic loading (COD_{max}) of 1800 $mg.L^{-1}$ and a maximum BOD of 1080 $mg.L^{-1}$. These values are two-fold the initial design parameters and indicate that the strength of municipal sewage entering from the Belmont Valley WWTW, should be reduced by 50% in order to achieve the correct BOD loading rate and derive the efficiency required.

Using the McGarry model (McGarry and Pescod 1970) however, an AFP area of 382.8 m^2 and a HRT of 9.2 d were obtained (Table 2) which is similar to the design specifications of the Belmont Valley IAPS suggesting that the AFP parameters are accordingly sufficient for a 500 PE IAPS with an influent loading of 40 $kg.d^{-1}$ BOD and indeed, the actual measured BOD of the outflow was between 300 and 444 $mg.L^{-1}$ (cf. estimated BOD of outflow of 486 $mg.L^{-1}$). Results also show that BOD loading of the IPD is close to 0.4. Values above 0.4 are indicative of malodours which would be a distinct disadvantage in deployment of the IAPS technology for treatment of municipal sewage. Second, closer scrutiny of the parameters for the AFP indicate that with an effluent BOD at 80 $mg.L^{-1}$ and according to first order kinetics, pond surface area and HRT should be 1389 m^2 and 33.3 d respectively which contrasts with the design parameters used of 840 m^2 and 20 d (Rose *et al.* 2002).

On the basis of measured values for nitrate-N, ammonium-N and *ortho*phosphate in the AFP effluent of 80 and 15 $mg.L^{-1}$ respectively, the specific loading of N and P to the HRAOPs was determined to be 60 and 11.25 $kg.ha^{-1}$. Using a raw sewage TKN value of 128 $mg.L^{-1}$ (Table 2), the N and P loads were determined to be 6.24 and 0.96 $kg.d^{-1}$ with a required HRAOP surface area for abstraction of these nutrients in the range 0.085-0.104 ha which compares favourably with the design specifications of the Belmont Valley IAPS HRAOPs. These have combined total surface area of 1000 m^2 (Rose *et al.* 2002).

Table 1

Table 2

Biocatalysts

Symbioses between bacteria and algae bring about an ecological pattern different from pure mass algae culture in which behaviour of these microorganisms depends on exploitation of mutualistic functionality leading to nutrient abstraction and remediation of wastewater (Kayombo *et al.* 2002; Craggs *et al.* 2004). Algae photosynthesis supplies aerobic bacteria with oxygen while bacterial oxidation produces CO_2, ammonium nitrogen and phosphate which is assimilated by algae into carbohydrates, lipids, proteins and other organic compounds (Oswald *et al.* 1957).

Based on geometrical equations to calculate the biovolume of algal species of different shapes from microscopically measured linear dimensions developed by Hillebrand *et al.* (1999) and Vadrucci *et al.* (2007), for algae species commonly found in wastewater treatment and the equations for five of the most dominant algae in wastewater HRAOPs (*Pediastrum* sp., *Desmodesmus* sp., *Micractinium* sp., *Dictyosphaerium* sp. and unicellular algae such as *Chlorella* sp., flagellates and *Thalassiosira* sp. (Park *et al.* 2011a). Belmont Valley IAPS HRAOPs show continual shifts in algae population dynamic with *Pediastrum* sp. replaced by *Micractinium* sp. in October 2012, which was replaced by *Pediastrum* sp. in December 2012, which was replaced by the unicellular diatom *Cyclotella* sp. in March 2013, which was replaced by *Pediastrum* sp. in December 2013, and then by *Dictyosphaerium* sp. in January 2014.

Similar changes in the dominant algae (including *Dictyosphaerium* sp., *Chlorella* sp., *Micractinium* sp., and *Desmodesmus* sp.) of a small-scale pilot wastewater treatment HRAOP in Spain have also been reported (Garciá *et al.* 2000). These shifts in population structure and algae dominance are possibly caused by changes in environmental conditions (in particular variations in solar radiation and pond water temperature) which are known to impact species selection, succession and co-existence. Additionally, HRAOP operational parameters and especially hydraulic retention time influences both algae population dynamic and species dominance (Benemann *et al.* 1977; Oswald 1988). Together these approaches for the determination of HRAOP biomass assume that microalgae (unicellular and colonial) are the major catalysts contributing to nutrient abstraction. Consequently, the contribution of bacteria to nutrient removal during water treatment is obscured and even overlooked.

Water quality

Cumulative data for water quality of the effluent from the IAPS at Belmont Valley WWTW from 1999-2014 is shown in Table 3. As can be seen COD and TSS of the final effluent exceeded the General Authorization for discharge (DWS 2010). However, percentage removal of COD was approximately 90 %. Average *ortho*phosphate and nitrate-N concentrations of the treated water were within the discharge limits stipulated by the Department of Water Affairs while the ammonium-N concentration was slightly above the discharge limit.

Table 3

Gap analysis of IAPS operation

The results of the gap analysis are illustrated in Table 4.The status quo of the technology is that treated water contains elevated levels of TSS and COD. Although there is no faecal sludge handling, biomass dewatering is required on a routine basis, which is clearly inefficient due to elevated residual TSS and COD. Originally designed to remediate an influent BOD of 40 $kg.d^{-1}$ recalculation of the kinetic parameters revealed that the strength of the raw sewage influent was not commensurate with design and that the actual BOD_{ult} load is close to 80 $kg.d^{-1}$ and not the specified 40 $kg.d^{-1}$. The current design and operational configuration of the IAPS at the Belmont Valley WWTW process lacks an inlet works, further the process is impeded via inadequate dewatering. The variability of the biocatalyst influences the performance of the settler causing poor dewatering. Insufficient dewatering presumably contributes to elevated TSS and COD of the treated water. This can be improved by utilizing other more efficient technologies such as drum filtration. Furthermore, faecal sludge handling is not required, technical skill and energy requirements are minimal, and IAPS technology has the potential to positively impact the environment. The potential for downstream value adding is immense with low environmental impact. Figure 4 presents a quantitation of the gap analysis for the major criteria used and presents a comparison between current and ideal practices.

Table 4

Figure 3

DISCUSSION

Using data generated by Rose *et al.* (2007) the results presented here indicate an IAPS receiving an 80 $kg.d^{-1}$ BOD load corresponding to 1000 PE, whereas the design specifications of the facility stipulate a 40 $kg.d^{-1}$ maximum BOD load concomitant with a 500 PE. Therefore, the influent BOD load needs to be reduced otherwise the water quality leaving the system will continue to contain elevated COD and TSS levels. Diverting the water from the raceways to the inlet of the IAPS would reduce this load through dilution. Also influencing the performance of the system is the dominant biocatalyst in the raceways, population dynamics shifted with *Pediastrum* sp. replaced by *Micractinium* sp. in October 2012, which was replaced by *Pediastrum* sp. in December 2012, which was replaced by the unicellular diatom *Cyclotella* sp. in March 2013, which was replaced by *Pediastrum* sp. in December 2013, and then by *Dictyosphaerium* sp. in January 2014. These shifts corresponded with changes in water quality, though able to treat wastewater to within DWS stipulations for *ortho*phosphate and nitrate-N, while removing 90 % COD; COD, ammonium-N and TSS levels did not comply with DWS discharge limits. Furthermore when a comprehensive gap analysis of current practices in comparison to best practices was conducted, results revealed, dilution of the influent would allow for the IAPS to generate a consistently compliant water quality, post-treatment would ensure more uses for the water emanating from the system while valorisation of by-products from the system such as methane and algae for fertilizer could be lucrative.

Raceways experience continual changes in population dynamics from *Pediastrum* sp. dominating to *Cyclotella* sp., however the action of paddlewheel mixing is postulated to encourage proliferation of more easily harvested filamentous algae strains encouraging natural settling (Mulbry *et al.* 2008). Sedimentation is a low cost dewatering option generating solids contents less than 1.5 %, however reliability of this technique is hindered by the fluctuating density of the algae biomass (Shen *et al.* 2008). Though ASPs of the IAPS reportedly remove 80 % algae biomass (Bailey-Green *et al.* 1996), this is not sufficient produce a water quality suitable for discharge into South African water courses. Thus Craggs *et al.* (2014) advocate the employment of tertiary treatment units inclusive of a maturation pond and a rock filter system for to treat water emanating from the ASPs. Oswald *et al.* (1994) used dissolved air flotation and sand filtration for post-treatment. Water quality improved to BOD <1 $mg.L^{-1}$, TSS 11 $mg.L^{-1}$, ammonium-N undetectable, nitrate-N 0.22 $mg.L^{-1}$ and *ortho*phosphate 0.12 $mg.L^{-1}$ (Oswald *et al.*, 1994). Total coliform count decreased to less than 200 $cfu.100mL^{-1}$. Thus post-treatment was able to generate a stream suitable for environmental discharge in South Africa. Craggs (2005) proposed discharging effluent from the raceways during the day time or in batches when algae are most active to improve the effluent quality from these biological systems. Carbon dioxide addition increase algae activity, increasing euphotic zones of raceways decreased settling of algae cells, optimal mixing speed improves cell exposure to sunlight elevating productivity while the retention time can be adjusted seasonally to retain ideal algae functionality. These abovementioned enhancements also increase the occurrence of desirable algae species, particularly the heavy and easily harvestable (Sutherland *et al.* 2014).

Gap analysis

Systems deployed globally in the absence of post-treatment do not generate a stream compliant with discharge standards. However, benefits of deploying the system surpass concerns. Faecal sludge handling is eliminated, the system can operate in a passive state when all the resources including technical skill for operation and maintenance are available on site. Further, by products in the form of methane harvested from the IPD can be used for heating and cooking, while algae biomass, can serve as either a substrate in methane generation or as a slow release fertilizer (Boshoff *et al.* 2004).

The IAPS can be configured to generate the desired effluent quality depending on the degree of water treatment required (Rose *et al.* 2002). A major disadvantage of ponding technologies is however the large areas required in comparison to electrochemical treatment systems and reverse osmosis plants (Craggs *et al.* 2012). Craggs *et al*. (2014) surmise that depending on the environmental discharge requirements, maturation ponds could serve as an adequate passive tertiary treatment step. These provide solar UV disinfection, polishing of waste water and effluent storage prior to discharge. While rock filter systems reduce effluent solid levels after treatment in a maturation pond. When land is limiting UV disinfection can replace maturation ponds generating a good quality effluent stream. For higher effluent quality this water can then be channelled through a membrane filtration system however, problems arise when the algae biomass clogs the pores. Algae sludge produced as a by-product of treating wastewater in by the IAPS directly and indirectly address issues associated with climate change (Oswald 1995), as algae serve as a carbon sink while aiding in the removal of excess atmospheric carbon dioxide (Benemann *et al.* 2003). Hindrances to the deployment of biological technologies for remediation of waste water also include environmental influences such as light, temperature, pH, predation by zooplankton, pathogens including bacteria, fungi and viruses as well as invading algae competition (Oswald *et al.* 1994).

IAPS is a simple technology requiring very low skill levels to operate. Other benefits of deploying an IAPS are that they require neither faecal biomass de-sludging nor sludge handling (Oswald *et al.* 1995; Downing *et al.* 2002). This is a vital attribute of the technology as it directly causes disease mitigation. In addition, construction of an IAPS is such that the land can be reclaimed in the future for a different purpose, ensuring construction of such a facility is not environmentally detrimental (Rose *et al.* 2002).

CONCLUSION

Technologies like the IAPS have been emphasized recently due to their ability to achieve the highest degree of purification at the lowest cost, with minimal maintenance (Adewumi *et al.* 2010; Park *et al.* 2011b). These passive systems are able to withstand varying shock loads while consistently generating the desired effluent quality when appropriately configured (Muga and Mihelcic 2008). As minimal skill levels are required for operation and maintenance of a technology like an IAPS, overall it will cost less over an extended period of time to run (Mwabi *et al.* 2011; Molinos-Senante *et al.* 2012). Therefore this gap analysis revealed a technology requiring a full scale commercial system to determine its full benefit to the South African water sector.

Figure 4

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Tables

Table 1 Effect of loading rate on influent flow as determined from the model developed to determine the kinetic parameters for IAPS implementation. Numbers in blue were derived experimentally and used to test the model using the documented design parameters reported by Rose *et al.* (2002)

Parameter	Design	Actual
Volume ($L.d^{-1}$)		
PE equiv	500	500
Vol per $PE.d^{-1}$	150	150
Flow	75 000	75 000
Strength ($mg.L^{-1}$)		
BOD	800	1080
COD	1000	1800
TKN	-	128
P_{total}	-	15
Loading ($kg.d^{-1}$)		
BOD	**40**	**81**
COD	-	135
TKN	-	9.6
P_{total}	-	1.125

Table 2 Design parameters for the Belmont Valley IAPS IPD and AFP based on the model developed to determine the kinetic parameters for IAPS implementation

Component Parameter	Value	Units	Comments
In-Pond Digester			
Optimum retention time	3	d	
Volume needed	225 000	L	
Depth	5	m	
Area	45	m^2	
Diameter	7.5	m	
BOD loading	360	$kgBOD.L.d^{-1}$	Odour problem above 0.4
Estimated BOD reduction	55	%	
Estimated BOD of outflow	486	$mg.L^{-l}$	
Advanced Facultative Pond (PFP)			
Depth	1.8	m	
Aspect	3		
Minimum temp	12	°C	For first order kinetics approach
Minimum temp	53.6	°F	For McGarry model
Effluent BOD	80	$mg.L^{-l}$	
First Order Kinetics Approach			
Area	1389	m^2	First order kinetics eq 7.13
Diameter	42.04	m	
BOD surface loading	0.026	$kgBOD.m^{-2}.d^{-1}$	
HRT	33.33	d	Much longer than current 20 d
McGarry Approach			
Area	382.87	m^2	McGarry eq 7.17 (valid 15-30°C)
Diameter	22.08	m	
BOD surface loading	0.0952	$kgBOD.m^{-2}.d^{-1}$	
HRT	9.20	d	Close to Belmont Valley IAPS

Table 3: Water quality from the Belmont Valley IAPS treating domestic wastewater

Parameter ($mg.L^{-1}$)	General Limit (Water Act, 1998)	Influent 1999-2014	Effluent 1999-2014
COD	75	780.5 ±539.5	83.8 ±27.8
TSS	25	n.d.	43.2 ±15.4
*Ortho*phosphate	10	12.9 ±5.2	5.0 ±1.3
Nitrate-N	15	6.9 ±1.8	5.3 ±3.7
Ammonium-N	3	11.9 ±10.3	3.2 ±1.9

Table 4: Results of a gaps analysis of the IAPS

Primary Factors	Secondary	Current Practices	Grade	The Gap	Best Practices in Future	Grade
Status quo of technology the	Water Quality	Not suitable for environmental discharge	5	Additional treatment needed	Suitable for discharge to a water course	7
	Sludge Handling	Faecal sludge -None	6		Faecal sludge - None	9
		Algae biomass discarded onto DBs	1	Algae biomass not valorised	Algae sludge digested to methane production and/or fertilizer production	8
	Operation	Spares not available on site	1	Insufficient equipment to support the system	Replacement paddles on site and spares	7
	Energy	Fossil fuel derived electricity used	1	Lack of knowledge for conversion of methane to electricity	Energy derived from the system	7
	Greenhouse gas emission	Vented to the atmosphere	2	Absence of gas capture equipment	Harvest biogas	8
	Skills	Minimal	8	No resident skill set	Minimal	8
	Knowledge	Advertising	7	Adoption	Universal acceptance and deployment of the technology	8
IAPS Process	Inlet works	Absent	1	Must be included in commercial design	Present legally	7
	Primary treatment	AFP/AD	5	Separate AFP and AD for inspection and maintenance	Separate AFP and AD	7
	Secondary treatment	HRAOPs	6	More HRAOPs/Increase size	Increase HRAOPs size/number	9
	Biomass removal	ASP	2	Adequate dewatering	Efficient ASPs/Drum filtration	8
Additional Treatment	Disinfection Final Polishing	None	1	Chlorination or UV	Maturation pond, rock filter system, drum filter, reverse osmosis, DAF	7

Biocatalyst	Varies	Determined by environmental influences *Pediastrum* sps, *Chlorella* sps, *Micractinium* sps and *Scenedesmus* sps result in optimal performance	5	Erratic species dynamics may negatively influence effluent quality *Cyclotella* sps was observed to reduce the effluent quality	Recirculation of effluent from the raceways will ensure most ideal species dominate	7
Wastewater	Flow	Even low flow	7	Centralized wastewater treatment plants	Decentralization of facilities will ensure that an even low flow for maximum efficiency dominates	8
	Organic/hydraulic load	Organic load not within design specifications	1	Monitor organic loading	Organic load according to design	8
Features	Costs	Costs to deploy this full scale system unknown	1	Deployment of a full scale IAPS WWTP	Commercial scale fully costed IAPS example	9
Technical	Skills	Minimal	8	No resident skill set	Low	8

Figure Headings

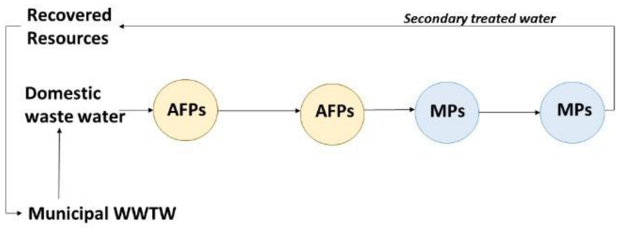
Figure 1 Schematic illustrating the process flow for various IAPS designs based on technology developed by Oswald to recover nutrients, energy and water from influent waste water. Nutrients - harvested algae from the algae settling ponds can be dried and sold as animal feed or fertilizer; Energy - near quality pipeline biogas can be captured, flared, or utilized to offset energy consumption from the WWTW. Water - the WWTW final effluent can be reclaimed for beneficial reuse. AFP=Advanced Facultative Pond; IPD=In-Pond Digester; HRP=High Rate Pond; C/F=Coagulation/Flocculation; ASP=Algae Settling Pond; MP=Maturation Pond; MMF=Multimedia Filtration; UV= Ultraviolet Light Disinfection. (http://www.go2watersolutions.com/process_schematic.html)

Figure 2: A schematic of the current process flow of the IAPS at the Belmont Valley WWTW. AFP: Advanced Facultative Pond; AD: Anaerobic Digester; HRAOP: High Rate Algae Oxidation Pond; ASP: Advanced Settling Pond; SB: Splitter Box; DB: Drying Bed.

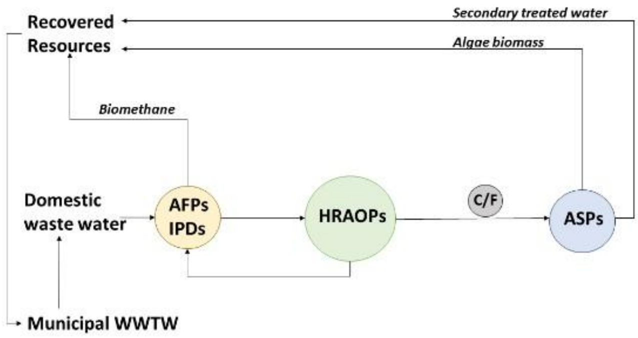
Figure 3: Grades in distinct major criteria. The grades were awarded according to current practices and most ideal practice.

Figure 4: Schematic of the current in comparison the ideal operation of the Belmont Valley IAPS relative to the commercial scale operating configuration.

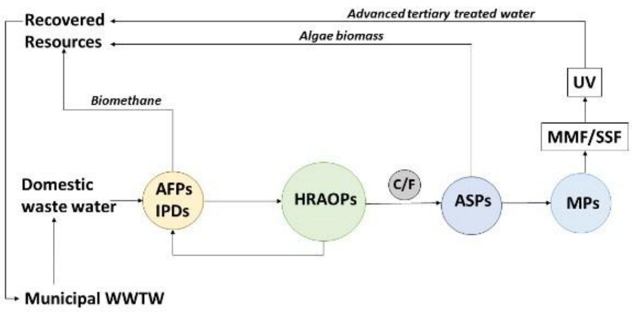
Figures



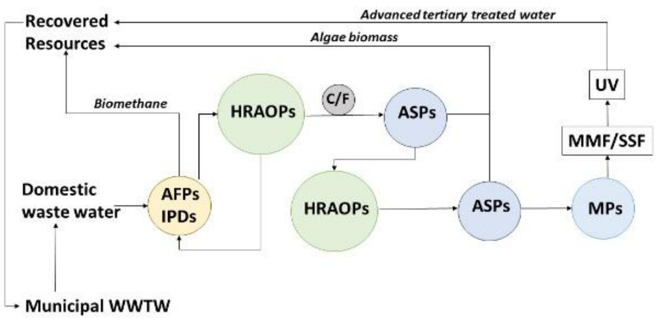
AIWPS Type II Process Schematic



AIWPS Advanced Secondary Process Schematic



AIWPS Tertiary Process Schematic



AIWPS Advanced Tertiary Process Schematic for Nutrient Removal

FIGURE 1

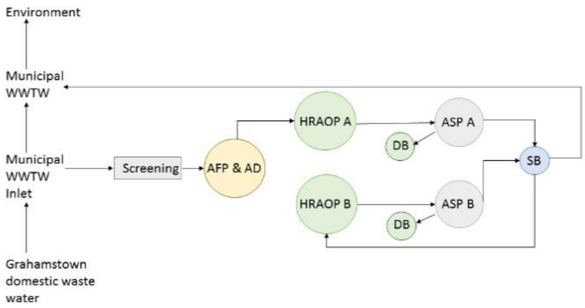


FIGURE 2

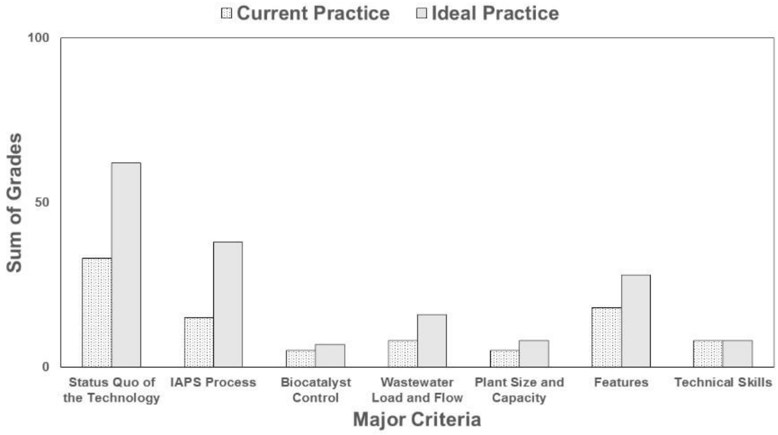


FIGURE 3

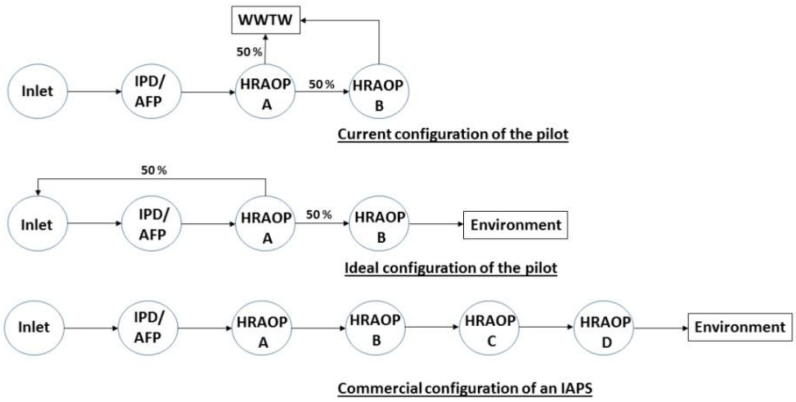


FIGURE 4

6.3 Operation of an IAPS for the treatment of municipal sewage: A South African case study

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ABSTRACT

IAPSs (IAPS) combine the use of anaerobic and aerobic bioprocesses to effect sewage treatment. In the present work the performance of IAPS was evaluated to determine the efficiency of this technology for treatment of municipal sewage under South African conditions. Composite samples were analysed over an eight month period before and after tertiary treatment. Spectrophotometric assays indicated that the treated water from this IAPS was compliant with the discharge limits for phosphate-P, ammonium-N, nitrate/nitrite-N and mean values were: 5.3 $mg.L^{-1}$, 2.9 $mg.L^{-1}$, and 12.4 $mg.L^{-1}$ respectively. Chemical oxygen demand however fluctuated significantly and was dependent on full function of the IAPS. Mean COD of the final treated water was 72.2 $mg.L^{-1}$. Although, these results suggest that the treated water discharged from this IAPS operating under South African conditions meets the standard for discharge, mean TSS was routinely above the limit at 34.5±13 $mg.L^{-1}$ and faecal coliforms were higher than expected. Tertiary treatment using a maturation pond series (MPS), slow sand filtration (SSF), or a controlled rock filter (CRF) ensured that the final treated water from the IAPS was of a quality suitable for discharge to the environment with CRF>SSF>MPS.

Key words | IAPS, IAPS, sewage, tertiary treatment

INTRODUCTION

Sewage and industrial pollution of water in South Africa pose risks to human and environmental health through the dispersal of waterborne pathogens and toxic organic and inorganic molecules. At present greater than 80% of South Africa‘s sewage treatment works are in disrepair, underperform or are overloaded. A rapid implementation of robust, easy to deploy and operate sewage treatment technologies is urgently required. Furthermore, climate change together with reduced water availability has major food security implications for South Africa, its neighbors and other arid, water-poor countries. These two factors alone have profound management implications for both government and business. Correct implementation and management of IAPSs (IAPS) developed for South African conditions can produce clean water for recycle and reuse (Rose, *et al*. 2002), provide energy, and generate a biomass suitable for valorization (Green *et al*. 1995a; Oswald 1995; Grönlund *et al*. 2004; Park *et al*. 2011; Craggs *et al*. 2012). Even so, and as with any treatment technology, there is an element of risk and/or failure to render a suitably treated final effluent.

The primary goal of the South African National Waste Management Strategy (NWMS) is the achievement of the objectives of the Waste Act, which are in summary: 1, minimizing pollution, environmental degradation and the consumption of natural resources, 2, implementing the waste hierarchy, 3, balancing the need for ecologically sustainable development with economic and social development, and 4, promoting universal and affordable waste services (Republic of South Africa, *Waste Act 2008*). Framed within the context of the overall goals, approach and regulatory model of the NWMS, implementation of a novel technology for the treatment of sewage demands that the final effluent meet General Authorisations in terms of Section 39 of the Water Act (Republic of South Africa, *Water Act 1998*) for discharge into a water resource that is not a listed water resource and comply with the General Limit values which are: faecal coliforms (per 100 mL) ≤ 1 000; pH 5.5-9.5; ammonia nitrogen ≤ 3 mg.L^{-1}; nitrate/nitrite nitrogen ≤ 15 mg.L^{-1}; *ortho*-phosphate ≤ 10 mg.L^{-1}; electrical conductivity 70-150 mS.m^{-1}; and COD ≤ 75 mg.L^{-1} (after removal of algae). Performance monitoring data for IAPS as a sewage treatment technology in South Africa is therefore needed not only to inform and educate through dissemination but to develop a roll-out strategy for implementation of full-scale commercial plants.

Prior research focussed on the four component ponds of the IAPS as standalone processes and the optimization of each but did not address performance of the system as a whole (Rose *et al*. 2002; Rose *et al.* 2007). As a consequence, there exists the perception that the treated effluent from IAPS does not meet the final COD and TSS concentrations due in part, to suspended algae moving over the weir of the algae settling ponds. In fact, a recent report on the operation of hectare-scale high rate algae oxidation ponds (HRAOP) for enhanced waste water treatment strongly advocated additional treatment of the outflow from algae settling ponds (ASP) by polishing to meet specific discharge standards (Craggs *et al*. 2012). These authors recommend the inclusion of one or a combination of maturation ponds (MP) and UV treatment by storage prior to discharge, or rock filtration of the MP effluent, or direct UV treatment if insufficient land is available, and if funds are available, membrane filtration to achieve a high quality final effluent for reuse. Clearly, there is therefore a need to establish an appropriate tertiary treatment unit (TTU) for implementation with IAPS and which compliments the low cost, environmental aspect of this sewage treatment technology. Despite concerns and in an effort to redress prevailing oversight, studies were initiated to examine the water quality of the final effluent from an IAPS treating municipal sewage. In this paper we report on operation of an experimental IAPS treating municipal sewage, the quality of the treated water, and the contribution of various tertiary treatment processes used to polish and enhance water quality prior to discharge.

MATERIALS AND METHODS

IAPS configuration and operation

The IAPS (IAPS) used in this study is located at the Institute for Environmental Biotechnology Rhodes University (EBRU), adjacent to the Belmont Valley Waste Water Treatment Works (33° 19‘ 07‖ South, 26° 33‘ 25‖ East) and operates continuously to treat 75 m^3.d^{-1} of municipal sewage and a schematic showing the operating configuration and process flow is presented in Figure 1. The complete system comprises an advanced facultative pond (AFP) with surface area of 840 m^2 containing a single in-pond digester (IPD) or fermentation pit (225 m^3), two 500 m^2 high rate algae oxidation ponds (HRAOP), and two algal settling ponds (ASP). Up-flow velocity in the fermentation pit is maintained at 1-1.5 $m.d^{-1}$ while hydraulic retention times (HRT) in the fermentation pit and AFP are 3 and 20 d, respectively. Screened raw sewage is sourced directly from an off take immediately after the inlet works and enters the system via the IPD, where suspended and dissolved solids are anaerobically degraded. Effluent then flows into the buffering AFP and is detained for 20 d before gravitating to the first HRAOP which has HRT of 2 d and then to an ASP for half a day. Mixing, or turbulent flow, is essential to maintain optimum conditions for maximum algae productivity in the HRAOPs and in the current system is 0.15 $m.s^{-1}$. Typically linear velocity is required to prevent stratification and is achieved using paddle wheels powered by a small electrical motor (0.25 kW). Due to configuration of the pilot demonstration and in accordance with original design parameters (Rose *et al*. 2002), partially treated water from the first ASP is pumped to the second HRAOP where it is detained for 4 d before release to the second ASP. The latter is where the bulk of suspended algae biomass is removed by sedimentation prior to tertiary treatment and eventual discharge of the treated water.

Tertiary treatment systems

For the purposes of the present study, three tertiary treatment processes were configured in parallel and investigated to determine water polishing efficacy. A maturation pond series (MPS), slow sand filtration (SSF) and controlled rock filtration (CRF) unit were plumbed to receive effluent from the IAPS immediately after separation of the algae biomass by passive settling in the algae settling pond. These tertiary treatment units were allowed to equilibrate for a period of four weeks prior to commencement of sampling of the respective final effluents and water analysis.

Three MP‘s were used and these were configured in series. MP 1 was constructed with a water depth of 1 m, from an inlet set at 1.2 m, to prevent water overflow and increase UV light penetration. As stated by Pearson *et al*. (1995), positioning and depth of inlet and outlet pipes tends to be more important for effluent quality and treatment competence than pond geometry itself. A retention time of between 10 and 20 d is sufficient for faecal coliform removal to levels less 1000 MPN per 100 mL (Craggs 2005). Thus, a total retention time across the MP series was, based on flow rate, configured to 12 days, i.e. 4 days in each pond. MP 1 was constructed using PVC lining (5×1.2 m) which was supported by steel fencing on the outside. The baffle, also of PVC lining was supported at the bottom by a weight to allow water flow under the baffle. MP 2 and 3 were 1 m^3 plastic containers equipped with an identical baffle system and the systems plumbed using 15 mm piping. Hydraulic retention time and flow through the MP series was constrained by the size of the receiving unit(s). Thus, the first MP with area 19.63 m^2 and depth 1.02 m, allowed for a holding volume of ~20 m^3. Using the expression;

$$A = Q\ \Theta m1/D$$

where, A=area (m^2); Q=influent flow (m^3/day); ϴm1=retention time (days) and D=depth, the flow rate to MP 1 was 4.9 $m^3.d^{-1}$, with both MP 2 and MP 3 receiving effluent at a flow rate of 0.2 $m^3.d^{-1}$ to give a HRT of 12 d.

Slow sand filtration (SSF) was achieved using a 1500 L JoJo® tank (1.5×1 m internal diameter) containing a 0.2 m layer of gravel covered by Geofabric (BIDIM®) followed by a

0.5 m layer of river sand covered in Geofabric and a head of water increasing to 0.8 m. A high water head pressure was required to overcome the effect of the Schmutzdecke (biofilm) which can cause clogging and decrease water flow into and through the system (Massmann *et al*. 2004). Thus, and according to recommendations (McNair *et al*. 1987) two SSFs were constructed in parallel – one in operation while the second was cleaned (scraping the biological layer from the surface of the sand). Hydraulic retention time and flow through the SSF were constrained by size and with area 0.785 m^2 and volume 1.18 m^3, a hydraulic loading rate (HLR) of ~1.3 $m \cdot d^{-1}$ was possible.

A controlled rock filter (CRF) was constructed using 3 plastic containers connected in series, each measuring 1.0×1.0×1.0 m and containing a 0.8 m layer of gravel (average particle size of 15-22 mm) as defined by Hussainuzzaman & Yokota (2005). The inlet pipe (15 mm i.d.) was positioned at the base of the CRF‘s to ensure water upflow into the filters and system performance (Middlebrooks *et al*. 2005). Flow rate into the CRF was 0.5 $m^3.d^{-1}$ with HLR of ~1.5 $m.d^{-1}$.

Water sampling and analysis

Composite sample collection was carried out weekly over an 8 month period in the summer from September 2012 to May 2013 during which the mean daily maximum and minimum temperatures were 24±6 and 11±4°C respectively. Secondary and tertiary treated water was collected at intervals spanning 24 h from points of discharge from both the IAPS and each of the three tertiary treatment units (i.e. MPS, SSF, and CRF), thoroughly mixed and a 500 mL subsample abstracted. Electrical conductivity and temperature were measured immediately using an EC Testr 11 Dual range 68X 546 501 meter (Eutech Instruments, Singapore), while dissolved oxygen content was determined using an 859 346 meter (Eutech Instruments, Singapore). The pH was measured using a Hanna HI8 424 microcomputer pH meter (Hanna Instruments, Woonsocket, RI). Total suspended solids (TSS) were measured according to APHA (1998) and nutrient analyses carried out according to the manufacturer‘s instructions using nitrate, phosphate, ammonium, sulphate and chemical oxygen demand (COD) test kits purchased from Merck Chem. Co., Darmstadt, Germany. A Thermo Spectronic Aquamate spectrophotometer (ThermoFisher Scientific, Waltham, MA) was used to interpolate values for the final concentrations from the respective limit curves. Whatman No. 2 filter paper with a pore size 8 µm was used to derive $COD_{filtered}$ values. Microbial analyses were carried out using MacConkey and m-Fc agar (BIOLAB CHEMICALS CC, South Africa) prepared according to the manufacturer‘s instructions. Petri dishes were spread-plate inoculated using 100 µL aliquots of treated water and incubated at 30 and 45 ºC respectively for 24 h prior to estimation of colony forming units (cfu).

RESULTS

Composite sampling was used to ensure that the values for the measured parameters were indeed thorough and comprehensively derived indicators of system performance. Sampling was at weekly intervals over an 8 month period and analysis of the physical, chemical and microbial characteristics of the treated water revealed the trends illustrated in Figures 2, 3, 4 and 5.

Figure 2 summarizes the physicochemical characteristics of the treated water after passage of municipal sewage through the IAPS process. Both DO and EC were in accordance with the General Authorization for discharge to a water course (Fig. 2A & B) whereas pH was routinely near or slightly above the upper limit (Fig 2A).

Figure 3 presents the time course of change in total suspended solids (TSS) and COD (after removal of algae i.e. $COD_{filtered}$) in the treated water from the IAPS. In addition, data for rainfall events were captured in an effort to account for any dilution effect on soluble COD and TSS concentration. Rainfall did not appear to have any significant impact on COD while values for TSS were reduced (Fig. 3B). Thus, there appears to be an interrelationship between precipitation events and TSS of treated water at point of discharge from IAPS presumably as a result of dilution. For discharge of treated water to a water course according to South African regulations, the TSS should not exceed 25 $mg.L^{-1}$. As shown in Figure 3B, extreme fluctuations in TSS were evident for treated water from the IAPS with minimum and maximum values of 5±1 $mg.L^{-1}$ and 90±9 $mg.L^{-1}$ respectively, and a mean for the 8 month period of 34.5±13 $mg.L^{-1}$.

Following removal of algae by filtration the $COD_{filtered}$ of the IAPS-treated water fluctuated from a minimum of 46.7±6 $mg.L^{-1}$to a maximum of 98±3 $mg.L^{-1}$ (Fig. 3C) and yielded a mean $COD_{filtered}$=72.2±13 over the 8 month period of analysis indicating that the effluent generated by this method of municipal sewage treatment does not consistently comply with the General Authorization and that a tertiary treatment process is required to further polish the treated water prior to discharge.

Nutrient removal efficiency of the IAPS was determined by analysing the ammonium-N, nitrate-N, and phosphate-P concentration in composite samples abstracted weekly from the final effluent stream and the results are shown in Figure 4. Aside from some initial noise in the data (scatter) water treated by the IAPS during the eight month period of monitoring appeared to comply with the General Authorization standard for environmental discharge for phosphate-P, nitrate-N, and ammonium-N (Fig. 4A, B & C). Indeed, calculation of the mean values (± standard deviation) for all determinations in the sampling window indicates that the final effluent from the IAPS routinely contained phosphate-P=5.3±2, nitrate/nitrite-N=12.4±4, and ammonium-N=2.9±1 $mg.L^{-1}$ (Table 1).

Figure 5 shows the total faecal coliform count in the final effluent from the IAPS. After week 3 (2013), there was a dramatic increase in colony forming units in the treated water caused by short circuiting due to incorrect positioning of the inlet pipe from ASP A. Consequently, partially treated water from HRAOP A was not detained in HRAOP B for sufficient time to allow for disinfection. However, following correct positioning of the inflow colony forming units of coliforms returned to levels acceptable for discharge. Thus, disinfection is dependent upon full function and correct operation of IAPS.

Table 1 presents the mean values for all data collected to determine water quality of the final effluent from the IAPS in relation to the General Authorization standards for discharge to a water course and compares these data to those obtained after passage of the effluent through three different tertiary treatment units. Clearly, water quality of the final IAPS effluent does not fully comply with the required limits for discharge. Tertiary treatment was carried out in parallel using a MPS, SSF and CRF and, although each unit received a fraction of the total IAPS final effluent, indications are that CRF and SSF were the more effective water polishing methodologies. This, coupled with an apparent lower land requirement suggests that these two tertiary treatment processes might be appropriate for use in combination with IAPS although other tertiary treatment processes such as a constructed wetland or reed beds still need investigation.

DISCUSSION

IAPS as a sewage treatment technology is a derivation of the Algae Integrated Wastewater Pond System (AIWPS) developed by Oswald who is credited as the pioneer of algae pond technology (Ludwig *et al*. 1952). Oswald initially focussed on the relationship between algae and bacteria in sewage treatment (Oswald *et al*. 1955). Later, he coined the term photosynthetic oxygenation (Oswald *et al*. 1957) and used this concept to describe the aeration effect caused by algae on treated water (Ludwig *et al*. 1951; 1952; Oswald *et al*. 1953; 1955). By 1957, Oswald had established the HRAOP which amalgamated sewage remediation via biological oxygenation and nutrient removal (Oswald *et al*. 1955; 1957) and eventually led to the fully developed AIWPS (Oswald *et al*. 1957).

The IAPS used in this study is similar in design to Oswald‘s –AIWPS Secondary Process‖, which typically produces a final effluent that does not comply with environmental discharge standards (Green *et al*. 1995b) and was designed and constructed without final polishing e.g. a maturation pond. Thus, and at best, the final effluent generated by this IAPS can only be described as a ‗secondary treated‘ water. It may seem surprising that a demonstration IAPS would be commissioned that did not produce an effluent compliant with standards. However, the IAPS pilot in question was configured as an aside to the Belmont Valley WWTW and commissioned for research and demonstration purposes only, with raw sewage sourced directly from an off take immediately after the inlet works and the final treated water discharged via the municipal WWTW to maturation ponds and the detail of its implementation and operation are recounted elsewhere (Mambo *et al*. 2014).

The present study was carried out to evaluate IAPS as a technology for municipal sewage treatment and to determine whether quality of the treated water was of a standard suitable for discharge to the environment. Results confirm as might have been expected that water quality of the IAPS final effluent operated under the conditions used, which are according to original design specifications (Rose *et al*. 2002) does not meet the standard for discharge in terms of both TSS and COD. Even so, levels of nitrate/nitrite-N, ammonium-N and phosphate-P, and DO and EC are all within the General Authorization limits (Republic of South Africa, *Water Act 1998*).

Apoptosis, the commonest phenotype of programmed cell death (PCD), has recently been elegantly demonstrated in microalgae and shown to cause the release of organic nutrients which are used by others in the population as well as co-occurring bacteria for growth (Orellana *et al*. 2013). As determined by these authors, a significant proportion of the algae population (55%±15) can undergo PCD at night after daytime growth. It is distinctly possible therefore that the elevated TSS and COD of water from an IAPS is due to algal PCD in the HRAOP and ASP suggesting that settled algae be removed as quickly as possible. Furthermore, the incorporation of an appropriate tertiary treatment unit (i.e. MPS, SSF or CRF) to consistently reduce levels of both TSS and COD will allow for discharge to a water course. From the present study CRF was more effective than either a MPS or SSF. Land availability notwithstanding, it is recommended that either CRF or SSF be further explored for possible incorporation as part of the design and process flow of IAPS destined for the commercial scale treatment of municipal sewage.

Disinfection remains a major concern and faecal coliform count increased during the course of sampling which negatively impacted the quality of the treated water produced by the IAPS. This is perhaps not unexpected considering system design and strongly supports the conclusion by Craggs *et al*. (2012) that additional treatment of the algae harvester effluent (i.e. outflow from the ASP) requires polishing to meet specific discharge standards. Furthermore, and during the course of the present investigation, short circuiting caused by incorrect influent pipe positioning into HRAOP B aggravated the coliform count. Once pipe placement was rectified there was a dramatic decline in coliforms present in the final effluent from week 13 onwards emphasising the disinfection potential of HRAOP B. Thus correct configuration and operation of the IAPS at all times is vital for effective waste water treatment

In conclusion the work described here set out; 1) to determine the water quality of the IAPS effluent over an extended operating period, 2) demonstrate compliance with the South African General Limit Values for discharge of up to 2 000 cubic metres of waste water on any given day into a water resource that is not a listed water resource, and 3) evaluate the contribution of tertiary treatment on levels of $COD_{filtered}$ and TSS in the treated water. Notwithstanding several design and operational constraints, it is concluded based on the presented data that:

- physicochemical characteristics including pH, DO, and EC of the IAPS effluent comply with the General Limit Values for discharge and that introduction of tertiary treatment to further reduce both TSS and $COD_{filtered}$ is essential;
- nutrient characteristics of the IAPS effluent comply with the General Limit Values for discharge and that addition of tertiary treatment will ensure that phosphate-P, nitrate/nitrite-N and ammonium-N concentrations are routinely below the limit values of 10, 15 and 3 $mg.L^{-1}$ respectively;
- a reduction in faecal coliforms in the treated water to comply with the General Limit Values for discharge is guaranteed only when the system is correctly operated (i.e. correct retention of the effluent in HRAOP B) and with an appropriate tertiary treatment unit.

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Table 1 │ Summary data on water quality of the effluent from the IAPS before and after tertiary treatment either by a maturation pond series (MPS), slow sand filtration (SSF), or controlled rock filtration (CRF). Also shown are the General Authorization limits for discharge to a water course (DWA, 2010). Data for effluent quality were determined on a per week interval over a period of 8 months.

Parameter	**General Authorization limit**[A]	**Water quality of final effluent**			
		IAPS	**IAPS + MPS**	**IAPS + SSF**	**IAPS + CRF**
pH	5.5-9.5	9.4 ± 1	9.9 ± 0.5	8.3 ± 0.7	8.0 ± 0.4
Dissolved oxygen ($mg.L^{-1}$)	>2	5.5 ± 1	13.5 ± 3.6	6.0 ± 2.5	12.6 ± 1.4
Electrical conductivity ($mS.m^{-1}$)	70 $mS.m^{-1}$ above intake to a maximum of 150 $mS.m^{-1}$	107.8 ± 19	94.5 ± 47.1	95.4 ± 48.3	100.1 ± 12.6
Chemical oxygen demand ($mg.L^{-1}$)[B]	75	72.2 ± 13	72.0 ± 10.1	59.3 ± 12.2	62.1± 4.4
Nitrate/nitrite-N ($mg.L^{-1}$)	15	12.4 ± 4	4.0 ± 1.9	6.7 ± 2.5	5.1 ± 1.7
Ammonium-N (mg.L-1)	3	2.9 ± 1	0.5 ± 0.5	2.3 ± 0.9	0.3 ± 0.1
Phosphate ($mg.L^{-1}$)	10	5.3 ± 2	4.3 ± 1.7	1.4 ± 0.6	0.5 ± 0.2
Total suspended solids ($mg.L^{-1}$)	25	34.5 ± 13	22.1 ± 12.3	19.3 ± 8.5	19.5 ± 9.8
Total coliforms ($cfu.100mL^{-1}$)	1 000	>1 000	<1 000	<1 000	<1 000

[A] General Authorisations in terms of section 39 of the national water act (Republic of South Africa, *Water Act 1998*)
[B] After removal of algae

FIGURE HEADINGS

Figure 1 │ Process flow and configuration of the experimental IAPS located at the Belmont Valley WWTW. The system receives raw sewage, screened for the removal of plastics via a grit or detritus channel. Pond and reactor surface area, volume and flow rates are shown in parentheses. Effluent enters at the bottom of the IPD some 6 m below water level. SB=Splitter Box; TTU=tertiary treatment unit.

Figure 2 │ Physicochemical characteristics of water following sewage treatment using an IAPS. A) pH B) electrical conductivity (EC) and C) dissolved oxygen (DO) were determined for composite samples collected weekly for 24 h over a period of 8 months. Data presented are the average of duplicate measurements.

Figure 3│ Relationship between rainfall events and total suspended solids and chemical oxygen demand of treated water from the IAPS at point of discharge. A) Precipitation was recorded weekly and, B) total suspended solids (TSS) and C) chemical oxygen demand ($COD_{filtered}$) were determined for composite samples harvested weekly over a 24 h period for 8 months. Stippled line indicates the General Limit values for TSS and $COD_{filtered}$. Data presented are the average of duplicate measurements.

Figure 4 │ Nutrient content and composition of treated water from the IAPS at point of discharge. A) Phosphate, B) nitrate/nitrite-N and C) ammonium-N concentrations in composite samples harvested weekly over a 24 h period for 8 months were determined using test kits as described in the Materials and Methods. Stippled line indicates the General Limit values for each parameter. Data presented are the average of duplicate measurements.

Figure 5 │ Estimation of faecal coliforms in treated water from the IAPS at point of discharge. Composite samples were harvested weekly over a 24 h period for 8 months. Data are the mean of 3 independent determinations.

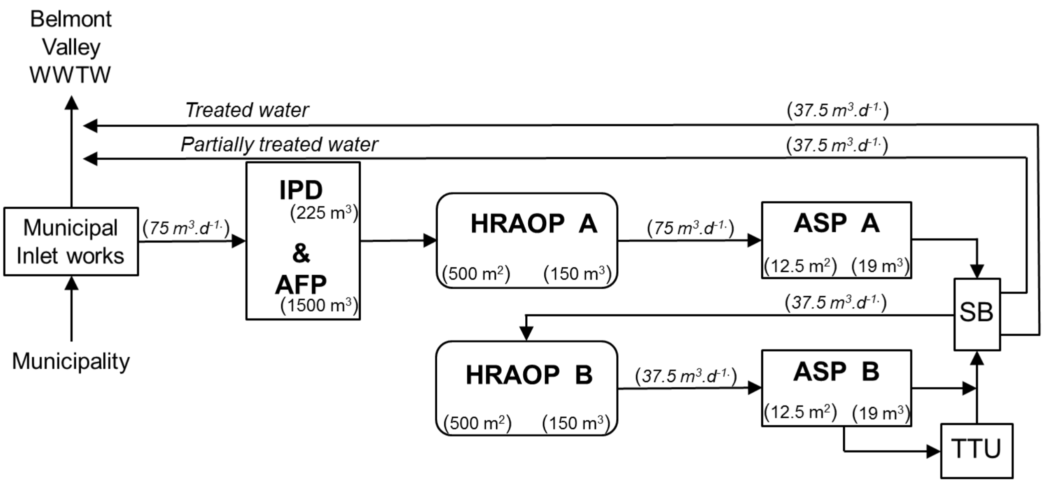

FIG 1

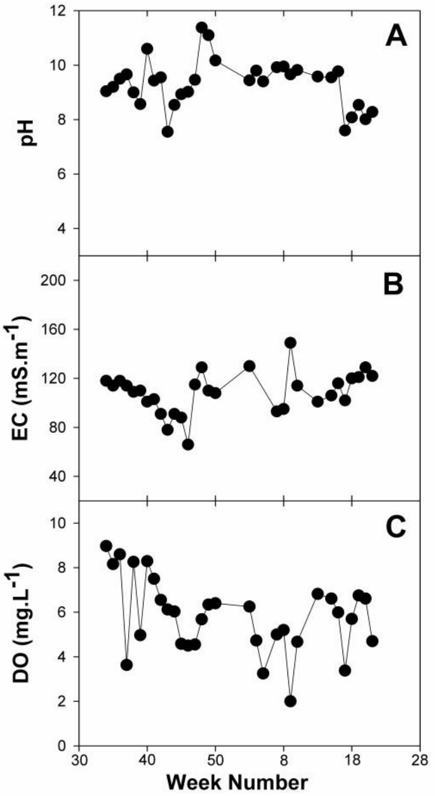
A
B
C
pH
EC (mS.m-1)
DO (mg.L-1)
Week Number

FIG 2

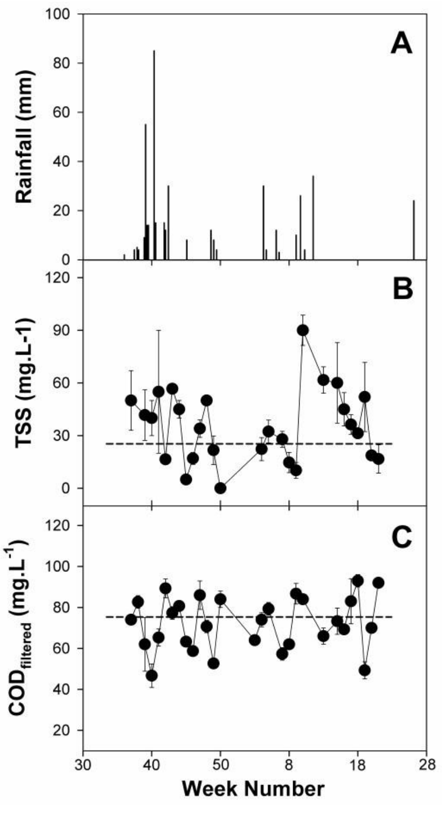
A
B
C
Rainfall (mm)
TSS (mg.L-1)
COD filtered (mg.L-1)
Week Number

FIG 3

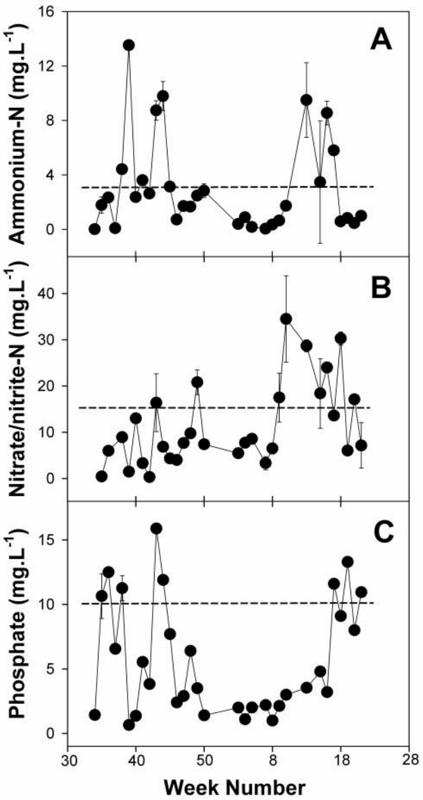
A
B
C
Ammonium-N (mg.L-1)
Nitrate/nitrite-N (mg.L-1)
Phosphate (mg.L-1)
Week Number

FIG 4

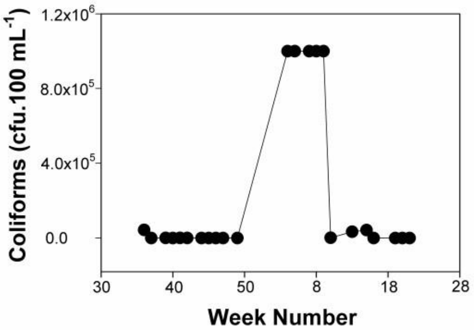
Coliforms (cfu.100 mL-1)
Week Number

FIG 5

6.4 Chemical oxygen demand and total suspended solids as limiting factors in IAPS as a wastewater treatment technology

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Chemical oxygen demand and total suspended solids as limiting factors in IAPS as a waste water treatment technology

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ABSTRACT

This study attempted to characterise the COD and TSS components in the final effluent from the Belmont Valley WWTW IAPS and to consider options to ensure compliance of this bioprocess. Further the study aimed to demonstrate that the elevated COD and TSS of water from an IAPS do not pose any threat to the environment. Shifts in dominance of the algae biocatalyst were also observed. *Pediastrum* sp. was replaced by *Micractinium* sp. in October 2012, which was replaced by *Pediastrum* sp. in December 2012, which was replaced by the unicellular diatom *Cyclotella* sp. in March 2013, which was replaced by *Pediastrum* sp. in December 2013, and then by *Dictyosphaerium* sp. in January 2014. Changes in species composition of the high rate algae oxidation ponds, inefficient removal of algae from the water column in the algae settling ponds, coupled with an organic load to the system at twice design capacity, appeared to be major factors contributing to elevated mean chemical oxygen demand 97.7 ± 15.7 mg.L^{-1} and total suspended solid concentration 35.0 ± 12.3 mg.L^{-1} measured between March 2013-November 2013, coincided with the dominance of *Cyclotella* sp. in the raceways. Sequential filtration and coupled chemical and biochemical analyses on water samples abstracted from the outflow demonstrated that residual chemical oxygen demand comprises mostly soluble organic carbon in the form of carbohydrates (49.7 ± 15.9) mg.L^{-1}, proteins (15.3 ± 3.9) mg.L^{-1} and lipids (6.0 ± 0.1) mg.L^{-1} derived presumably from algae biomass.

INTRODUCTION

The South African Department of Water and Sanitation (DWS) regulations for discharge of a stream from a WWTW into a natural water resource stipulate COD ≤75 mg.L^{-1} and TSS ≤25 mg.L^{-1} (DWS, 2013). Comprehensive evaluations of domestic waste water treated by the Belmont Valley Integrated Algae Pond System (IAPS) reveal a water quality unsuitable for release into the environment (Rose *et al.* 2007; Mambo *et al.* 2014). Oswald *et al* (1994) also observed an outflow containing TSS 163 mg.L^{-1} at the Richmond Advanced Integrated Waste Water Pond System (AIWPS) in California, U.S.A. However following dissolved air flotation and sand filtration, water quality improved as TSS measurements decreased to 11 mg.L^{-1}. These results are emulated by Puskas *et al*. (1991) in Kuwait, COD 130 mg.L^{-1} and Craggs *et al* (2003) in New Zealand TSS levels, 64 mg.L^{-1}, then later TSS 87 mg.L^{-1} (Craggs *et al.* 2004). Elevated COD is problematic as the receiving water body experiences anaerobicity which reduces biodiversity, while turbidity due to high TSS concentrations inhibits light penetration and concomitantly photosynthetic activity in the affected area. Meiring and Oellerman (1995) suggested that elevated COD and TSS in treated water from IAPS was a consequence of inadequate design and operation. Mambo *et al.* (2014a) suggested that the dewatering technique, specifically algae settling ponds (ASPs), used at the pilot plant is inadequate, given the biological nature of the system. Thus given the observations above, IAPS and other high rate algae pond WWT plants seldom meet the COD and TSS criteria set by DWS which has prompted the need for additional treatment of the outflow from ASPs by polishing to meet specific discharge standards (Craggs *et al*. 2012). It should be stated that due to the biological nature of the facility, elevated COD and TSS may not necessarily result in the detriment of the environment. Therefore the present study attempts to characterise the COD and TSS content in final effluent from the Belmont Valley WWTW IAPS and to consider other unit operations which might ensure compliance for this bioprocess technology. The aim of the present study is also to demonstrate that the elevated COD and TSS content of water emerging from an IAPS do not pose any threat to the environment.

MATERIALS AND METHODS

The IAPS

The IAPS pilot located at EBRU adjacent to the Belmont Valley WWTW (33° 19′ 07″ S, 26° 33′ 25″ E) was evaluated. The facility is used to treat 75 kL.d^{-1} municipal sewage and comprises an AFP containing an I-PD (840 m^2) and a primary facultative pond with a surface area of 225 m^2, two HRAOPs (500 m^2) and two ASPs. This IAPS does not have a post treatment unit to polish effluent from the ASPs.

Figure 1

Sample collection and analysis

Composite sample collection, microbial and physicochemical of the influent and effluent was carried out as described by Mambo *et al*. (2014b), while TSS, TVS and TDS analyses were conducted in accordance with APHA (1998).

Biochemical analyses

Carbohydrate content using the phenol sulphuric acid assay according to Dubois *et al*. (1956) was used to determine the polysaccharide content of the final effluent. Reducing sugar levels were determined in accordance with the dinitrosalicylic acid assay described by Miller (1959). Bradford (1976) was used to determine the protein concentration in the water samples, while the ninhydrin assay described by Hwang and Ederer (1975) was used to detect α amino nitrogen in the water. Humic acid-like substance concentrations were determined utilizing humic acid obtained from Fluka. This humic acid-like substance was dissolved in 0.1 M sodium hydroxide to a 200 mg.L^{-1}, following which, this standard was serially diluted. In line with protocols described by Wang and Hsieh (2000), the wavelength at or near the absorbance maxima was determined to be 300 nm. A standard curve of absorbance versus concentration at 300 nm was consequently constructed and used to quantify the level of humic acid-like substances in solution in the final effluent from the IAPS.

RESULTS

A summary of the performance of the Belmont Valley IAPS for the period March to November 2013 (i.e. through the autumn and winter) is shown in Table 1. Measurements of COD and TSS at 97.7 ±15.7 mg.L^{-1} and 35.0 ±12.3 mg.L^{-1} respectively do not comply with DWS (2010) environmental discharge limits. Elevated NH_4-N in the treated water may have arisen due to the low ambient temperatures experienced during the sampling period. Also, during the sampling period the microbial dynamic in the HRAOPs shifted from dominance by green alga *Pediastrum* in favour of the diatom *Cyclotella* (Fig. 2).

Table 1

Belmont Valley IAPS HRAOPs show continual shifts in algae population dynamics with *Pediastrum* sp. and *Cyclotella* sp. examples observed during March to November 2013 presented in Figure 2.

Figure 2

Analysis of the COD in final effluent after sequential filtration of the water through 11, 1.6, 0.45 and 0.22 μm pore size filters revealed there to be no significant change (Table 2). Initially, a preliminary investigation revealed that percentage change relative to COD and $COD_{filtered}$ remained unchanged despite the decreasing filter pore size (Experiment 1, Table 2). This prompted a more rigorous analysis spanning several months which again revealed no significant difference between COD and $COD_{filtered}$ in water from the IAPS final effluent (Experiment 2, Table 2). Taken together, these findings strongly suggest that the recalcitrance of COD in IAPS final effluent is largely due to dissolved organic carbon rather than particulate organic carbon.

Table 2

Partial support for the COD being of predominantly dissolved organic carbon in origin was obtained by measuring the amount of humic acid-like material in the water both before and after sequential filtration and the results are shown in Table 3. Levels of humic acid-like substances were low and only after filtration through a 0.45 μm filter, was there any real reduction in content of humic acid-like material suggesting that the bulk of this component comprised smaller humics (i.e. low molecular weight humics) and possibly fulvic acids.

Table 3

In addition to the above analyses, final effluent from the IAPS was analysed for C, N, H, and S using an elemental analyser and the C:N ratio determined (Table 4). Results show that the relative concentrations of these elements was unaffected by filtration and that the C:N ratio closely approximated the Redfield ratio (i.e. 6.6) indicating a nutrient composition typical of that found in both fresh water and marine environments (Redfield 1958).

Table 4

An in depth screening of the IAPS final effluent for compounds that might have originated due to programmed cell death of algae in either the HRAOPs or the ASPs revealed elevated TSS 31.3 ±16.3 $mg.L^{-1}$, NO_3-N 14.9 ±10.5 $mg.L^{-1}$ and NH_4-N 3.3 ±3.2 $mg.L^{-1}$ as shown in Table 5.

Table 5

DISCUSSION

Results of the present investigation reveal an IAPS producing a water of a quality that is not suitable for discharge into the environment. Measurements of COD 97.7 ±15.7 $mg.L^{-1}$ and

TSS 35.0 ±12.3 $mg.L^{-1}$ may have been impacted by changes in the population dynamics in the HRAOPs. *Cyclotella* sp. dominated the raceways during the course of the study, having replaced a predominantly *Pediastrum* sp. population. It was further found that a substantial portion of the COD was <0.22 µm in size, strongly suggesting a COD composed of dissolved organic carbon rather than particulate organic carbon. Further, programmed cell death in the HRAOPs and ASPs may be contributing to the elevated TSS and COD resulting in the non-compliance of the final effluent.

Perhaps the best studied example of an IAPS and its performance for the treatment of municipal sewage is the pilot plant designed for use in arid regions shown in Fig. 3 (Esen *et al*. 1987; 1991; Puskas and Esen 1989; Banat *et al*. 1990; Puskas *et al*. 1991; Al-Shayji *et al*. 1994). Similar to the Belmont Valley IAPS, the system described by Banat *et al*. (1990) comprised a deep facultative pond (AFP) followed by shallow high-rate oxidation ponds (HRAOPs) and separation or sedimentation ponds (i.e. inclined ASPs) and was based on the typical Oswald design. In this system the facultative ponds were designed to receive wastewater and partially reduce organic loading through fermentation (presumably an IPD) while the high-rate ponds were for both wastewater treatment and the production of algae biomass. This system therefore resembles very closely the demonstration IAPS under study in the present project. According to the authors the algal-bacterial ponding system performed satisfactorily providing treated effluent with less than 20 $mg.L^{-1}$ BOD, 130 $mg.L^{-1}$ COD, 40 $mg.L^{-1}$ total nitrogen and 25 $mg.L^{-1}$ NH_4-N. The average production of algal biomass was 0.025 $kg.m^2.d^{-1}$. Proper disinfection was achieved, indicated by average bacterial counts of 5 N/ml total coliforms and 1000 N/ml total bacteria (Puskas *et al*. 1991). Furthermore, seasonal weather variations, dense wastewater and fluctuating organic and hydraulic load, did not appear to adversely affect performance of the system. Apparently, this pilot system routinely yielded efficient algae sedimentation resulting in an appropriate effluent with total suspended solids less than 20-30 $mg.L^{-1}$ but only when operated at optimum. More typically, and as stated by Esen *et al*. (1991), the effluent was rich in algae, and did not meet stringent water-quality criteria on suspended solids. In mitigation of these negatives, slow sand filtration was introduced to remove algae and algae residues from high-rate oxidation pond effluents. Thus, when agricultural sandy soil with an effective grain size of 0.08 mm was used as the filter medium, an average filtration rate of about 1.3 $m^3/m^2.d^{-1}$ was obtained and the final effluent had a BOD value less than 20 $mg.L^{-1}$ and undetectable faecal coliforms (Esen *et al*. 1991).

Figure 3

It has been hypothesized that nitrogen-replete diatoms release NO_2^-, NH_4^+ or dissolved organic nitrogen (DON) following rapid increases in irradiance and consequently an increase in cellular electron energy (Lomas and Gilbert 1999). Similarly, a decrease in temperature, due to the temperature dependency of biosynthetic enzymes, increases cellular energy. Indeed, release rates of NH_4^+ under increased irradiance were shown to be nearly fivefold greater than release rates at the growth irradiance, and to account for 84 % of the NO_3^- uptake rate (Lomas *et al*. 2000). Thus, it is perhaps not surprising that in HRAOPs populated with a dominant diatom species such as *Cyclotella* levels of ammonium-N were in amounts 2.5

times above the General Limit for discharge. Metabolic processes are distinctly coupled in microbial communities such as those present in HRAOPs: photosynthetic primary producers (bacteria and/or unicellular algae) release C- and N-based DOM comprising various organic compounds and amino acids that are readily assimilated and re-mineralized by heterotrophic bacteria/archaea and protozoa (Pomeroy 1974; Azam 1983). Phytoplankton and bacteria are the main sources of DON (Jioa *et al*. 2010) and the biopolymers produced arise by mechanisms including direct release, mortality by viral lysis, regulated exocytosis of metabolites and polymergels, grazing, and apoptosis. Apoptosis, the commonest phenotype of programmed cell death (PCD), is well documented in chlorophytes allowing cellular materials to become dissolved in the environment and it has recently been elegantly demonstrated that PCD in microalgae causes the release of organic nutrients which are used by others in the population as well as a co-occurring bacteria to re-mineralize the dissolved material promoting algal growth. PCD in algae is therefore a mechanism for the flow of dissolved photosynthate between unrelated organisms. Ironically, programmed death also played a central role in the organism's own population growth and in the exchange of nutrients in the microbial loop (Orellana *et al*. 2013). Algal species typically found in HRAOPs associated with WWTWs include: *Desmodesmus* sp., *Micractinium* sp., *Pediastrum* sp., *Dictyosphaerium* sp. and *Coelastrum* sp. As indicated by others, these algae often form large settleable colonies (diameter: 50-200 μm), which enable cost-effective and simple algae biomass removal by gravity sedimentation (Lavoie and de la Noue 1986; Garcıá *et al*. 2000; Craggs *et al*. 2011; Park *et al*. 2011a) and, as might be expected, the HRAOPs of the Belmont Valley IAPS are also dominated by these colonial microalgae species.

Gravity sedimentation is the most common and cost effective method of algal biomass removal in wastewater treatment because of the large volumes of wastewater treated and the relatively low value of algal biomass generated (Nurdogan and Oswald 1996). However, algae settling ponds, and as indicated above, have relatively long retention times (1-2 d) and only remove 50-80% of the biomass (Nurdogan and Oswald 1996; Brennan and Owende 2010; Park and Craggs 2010; Park *et al*. 2011a). Chemical and mechanical approaches to algae removal have the disadvantage that they increase the overall operating costs making the bioprocess far less attractive. Thus, it is clear that an alternative simple and cost effective method is required. One such method and one which has been used at EBRU, involves the recycling of algal biomass already harvested by gravity settling to increase the dominance of readily settleable algae. Success in this regard was demonstrated earlier in small-scale laboratory cultures (Benemann *et al*. 1977; Weissman and Benemann 1979). More recently, the influence of recycling selectively harvested algae on species dominance and harvest efficiency in a pilot-scale wastewater treatment HRAOP was investigated over one year (Park *et al*. 2011b). These researchers demonstrated that recycling of harvested algae biomass back to the HRAOP maintained the dominance of a single readily settleable species (*Pediastrum* sp.) at >90% over one year (compared to HRAOPs with no recycling in which only a 53% dominance was recorded) and increased the average size of *Pediastrum* sp. colonies by 13-30%. It follows that algae of larger biovolume will more likely form larger flocs which readily settle in the ASPs. A final consideration involves the incorporation of drum filtration into the process flow and design criteria for IAPS at the expense of ASPs.

CONCLUSION

This study has shown that species composition of the HRAOPs and gravity settling and ASP operation contribute to both the COD and TSS of water from IAPS treatment of municipal sewage and in terms of these parameters, the final effluent does not meet the General standard for discharge to a water course. Although insufficient data has been collected thus far it is tempting to suggest that dark-induced programmed cell death occurs in algae both in the HRAOPs and the ASPs which causes release of ‗nutrient' into the water to fulfill the microbial loop as recently demonstrated by Orellana *et al*. (2013). Support for this is indicated by the C:N ration of sequentially filtered water samples which were closely allied to the Redfield ratio. As pointed out by Al-Shayji *et al*. (1994) algae floc formation and settling are sensitive to climate and operational parameters (e.g., sunshine, hydraulic conditions in settling tank, variations of the flow rate, and pond depth) and if conditions fluctuate, 20-30% of the biomass can be lost – presumably by onset of cell death. It is this aspect of IAPS technology that clearly requires further attention. In the interim, possible considerations to address the above shortcoming include careful and stringent operation and implementation of an appropriate tertiary treatment (e.g. slow sand filtration). In addition, recycling of algae to select specifically for readily settleable flocs should be further investigated.

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Tables

Table 1 Water quality parameters of the final effluent from an IAPS treating municipal sewage at the Belmont Valley WWTW, Grahamstown. Data were from samples collected at regular intervals between 1 March and 20 November 2013 and are presented as the mean ± SD.

Water quality parameter (units)	
pH	8.8±0.4
Dissolved oxygen (mg.L^{-1})	5.5±1.3
Electrical conductivity (mS.m^{-1})	116.4±18.6
Nitrate/nitrite-N (mg.L^{-1})	7.0±4.4
Ammonium-N (mg.L^{-1})	7.4±1.1
Phosphate (mg.L^{-1})	10.2±3.6
Chemical oxygen demand (mg.L^{-1})[A]	97.7±15.7
Total suspended solids (mg.L^{-1})	35.0±12.3
Total coliforms (cfu.100mL^{-1})	>1000

[A]Following removal of algae

Table 2 Recalcitrance of COD in the final effluent of the IAPS treating 75 $m^3.d^{-1}$ municipal sewage at the Belmont Valley WWTW, Grahamstown. Data followed by the same letter are not significantly different (one-way ANOVA, *p*<0.05).

Filter pore size (µm)	Unfiltered	11	1.6	0.45	0.22
	COD (mg/L)				
Experiment 1					
Mean	126.8±36.7[a]	94.0±15.6[a]	74.4±7.5[a]	68.8±10.6[a]	71.3[a]
% change in COD	0	-25.8[b]	-41.3[b]	-45.7[b]	-43.8[b]
% change in $COD_{filtered}$	-	0	-20.8[c]	-26.8[c]	-24.1[c]
Experiment 2					
Mean	88.6±25.5[a]	76.8±15.6[a]	57.6±11.1[a]	58.8±15.8[a]	n.d.
% change in COD	0	-13.3[b]	-34.9[b]	-33.6[b]	n.d.
% change in $COD_{filtered}$	-	0	-25.0[c]	-23.4[c]	n.d.

Table 3 Concentration of humic acid-like substances in water from the Belmont Valley IAPS final effluent. Humics were analysed as described in Materials and Methods. Data are the mean ± S.D. of at least four determinations.

Filter pore size (µm)	Humic-like substances (mg.L^{-1})
Unfiltered	10.2 ± 0.10
11	9.7 ± 0.04
1.6	9.2 ± 0.10
0.45	6.4 ± 0.10

Table 4 Carbon, nitrogen, and sulphur content of the IAPS final effluent determined before and after sequential filtration of water.

Filter pore size (µm)	C	N	H	S	C:N
	(%)				
Unfiltered	6.8	1.0	0.9	2.7	6.8
11	6.8	1.2	0.7	2.4	5.7
1.6	6.6	1.3	0.8	2.5	5.1
0.45	5.9	0.8	0.7	2.3	7.4

Table 5 Total soluble solids (TSS) and metabolite content and composition of water samples from the IAPS final effluent. Data were from samples collected at regular intervals between 1 March and 20 November 2013 and are presented as the mean ± SD.

TSS	31.3±16.3
Metabolite	
Nitrate/nitrite-N ($mg.L^{-1}$)	14.9±10.5
Ammonium-N ($mg.L^{-1}$)	3.3±3.2
Phosphate ($mg.L^{-1}$)	5.3±4.0
Humic substances ($mg.L^{-1}$)	6.0±1.7
Reducing sugars ($mg.L^{-1}$)	19.3±9.4
Carbohydrates ($mg.L^{-1}$)	49.7±15.9
Protein ($mg.L^{-1}$)	15.3±3.9
Alpha amino nitrogen ($mg.L^{-1}$)	1.8±1.1

Figure headings

Figure 1 Schematic of the operating process flow of the Belmont Valley municipal WWTW IAPS (IAPS). The system receives 75 $m^3.d^{-1}$ of raw sewage, screened for the removal of plastics via a grit or detritus channel. Effluent enters at the bottom of the IPD some 6 m below water level. SB=Splitter Box.

Figure 2 Transition of species dominance in the HRAOPs from *Pediastrum/Scenedesmus* to *Cyclotella* (boxes) between March and April/May 2013.

Figure 3 Process flow of a concept IAPS with in-line drum filtration.

Figures

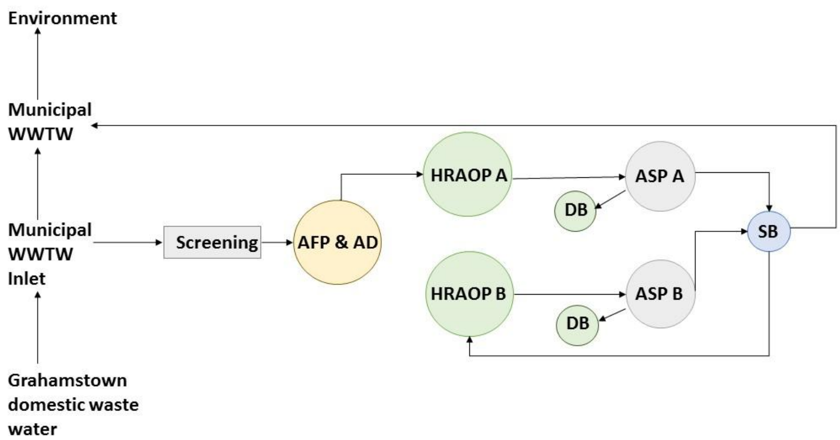

FIGURE 1

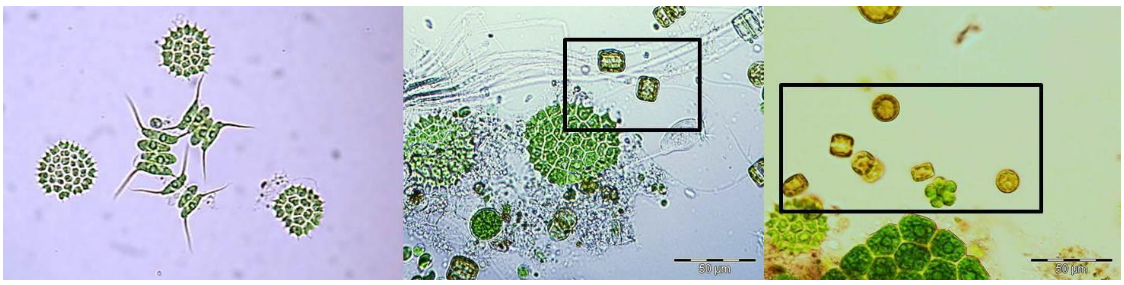

FIGURE 2

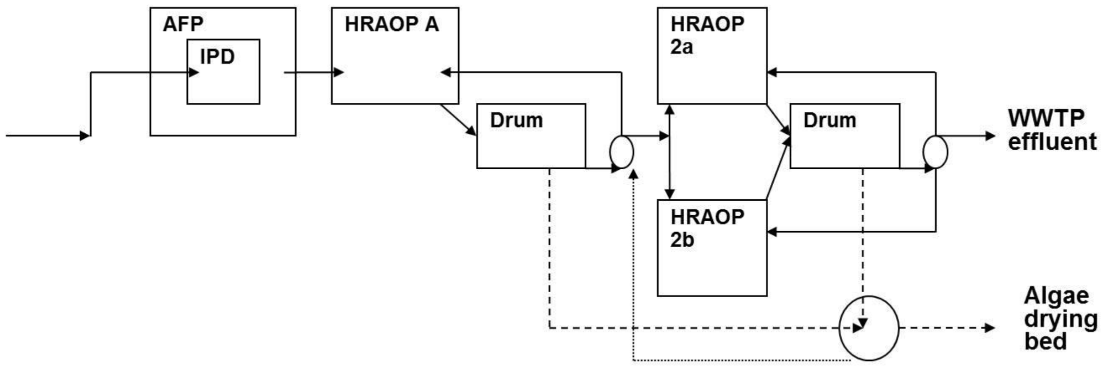
AFP
IPD
HRAOP A
Drum
HRAOP 2a
Drum
WWTP effluent
HRAOP 2b
Algae drying bed

FIGURE 3

7 Supplementary Information

Mambo, P. M., Westensee, D. K., Zuma, B. M. & Cowan, A. K. 2013 Golden Pond? – Yes! ‗Water Sewage & Effluent **33(**2**),** 42–44.

‗Golden Pond?' by Laxton (WS&E, January 2010) summarized the configuration, operation and capacity of an IAPS (IAPS) located at the Belmont Valley WWTW in Grahamstown. The system is modelled on one of a variety of advanced integrated waste water pond system (AIWPS) designs created by the late Professor Bill Oswald and his co-workers and was in fact designed by Oswald himself. This IAPS comprises a primary facultative pond with submerged anaerobic digester or fermentation pit, which together feed two high rate algae oxidation ponds linked in series for nutrient removal and two algae settling ponds that separate the resultant biomass from the treated water. Configuration and operation of each component of the system determines quality of the final effluent.

The Belmont Valley IAPS is very similar in design to Oswald's ―AIWPS Secondary Process‖, which typically produces a final effluent that does not comply with environmental discharge standards. It may seem surprising that a demonstration IAPS would be commissioned that did not produce an effluent compliant with standards. However, the IAPS pilot in question was configured as an aside to the Belmont Valley WWTW with raw sewage sourced directly from an off take immediately after the inlet works and the final IAPS treated water discharged to the maturation ponds, both integral parts of this municipal waste water treatment works. Thus, the Belmont Valley IAPS treats a typical South African waste water stream and produces a final effluent which can be considered a disinfected water suitable for reclamation and presumably discharge.

The reality in South Africa is, unfortunately, a perception that IAPS does not meet the final COD and TSS concentrations due in part, to suspended algae moving over the weir of the algae settling ponds. Despite concerns by the Department of Water Affairs, the Water Research Commission (WRC) has lauded the technology which it believes can benefit South Africa in many ways. In an effort to redress prevailing oversight and misconception the WRC is funding a technology evaluation study at the Institute for Environmental Biotechnology, Rhodes University (EBRU) to produce information on: 1) how IAPS meets final effluent, sludge and sustainability standards; 2) operation and maintenance guidelines; 3) design analysis; and 4) sustainability benefits. The outcome will be an uptake document which answers any and all questions posed by authorities when deciding whether or not to implement a technology such as IAPS.

Compliance and standards

According to the General Authorisations in terms of Section 39 of the Water Act (Republic of South Africa, *Water Act 1998*) the discharge of domestic and industrial waste water into a water resource that is not a listed water resource must comply with the General Limit values of amongst others: faecal coliforms (per 100 mL) $\leq$ 1 000; pH 5.5-9.5; ammonia nitrogen $\leq$ 3 mg/L; nitrate/nitrite nitrogen $\leq$ 15 mg/L; *ortho*-phosphate $\leq$ 10 mg/L; electrical conductivity 70-150 mS/m; and COD $\leq$ 75 mg/L (after removal of algae). Apart from short periods of downtime for routine maintenance and repair, the Belmont Valley WWTW demonstration IAPS has been operating continuously since 1996. There has been no need to desludge the system and no sludge handling/management has been required. Data accumulated during 2012-2013 confirm that both the secondary and tertiary treated water emerging from the IAPS comply with the General Limit values. Thus, the IAPS final effluent (post maturation pond) with $COD_{filtered}$ and TSS values of 63±10 mg/L and <10 mg/L respectively compares favourably with data over the same period for the Belmont Valley WWTW final effluent (cf. $COD_{filtered}$=56±23; TSS=29±6) which discharges from maturation ponds into a water resource. Thus, IAPS which is a passive process that uses gravity, solar energy and biological activity to treat waste water is a technology that can be adapted to prevailing conditions and used to bolster conventional waste water treatment processes such as oxidation ponds, biofilters and activated sludge that are either underperforming, overloaded or in a state of disrepair.

Rural is a state of mind

IAPS technology is advanced in concept although basic in design and requires minimal skill to operate and maintain. Laxton's appraisal of IAPS as ‗rural' however, suggests a technology with limited potential. On the contrary, IAPS can be implemented virtually anywhere depending on public needs, awareness and resources. Unlike in developed countries, land is currently not a limiting factor in South Africa, thus ‗rural' is simply a state of mind. But with a growing population, South Africa might be forced to decentralize and possibly privatize much of its waste water treatment. Although decentralization carries with it an element of risk, IAPS is ideally suited and is easily configured to cater to specific needs of industries and institutions, suburbs, cluster houses, holiday resorts, golfing estates and remote villages. Furthermore, although Laxton positioned IAPS as a coalesce solution to local domestic waste water problems by depicting the technology as low cost, sustainable, low skill and robust he missed one real advantage which is, that once constructed the technology operates in relative perpetuity.

Water concerns in context

South Africa is a water stressed and semi-arid country due to low average annual rainfall, high evaporation rates, regular droughts and limited natural freshwater reserves. Management of the water resource and waste water must therefore be in concert with population growth, industrial development and urban migration. ‗Golden Pond?' brought to light an archetype where electricity and water were now more costly, the skilled work-force was emigrating and waste water treatment facilities were either dysfunctional or in disrepair. Although, this may be true, it is not entirely due to governmental failure as alluded to by Laxton. It is the result of the glaring need to provide equal access to basic services. Waste water will always be generated and in ever increasing quantities, and will continue to be detrimental to the environment if managed incorrectly. Rather than deploy energy expensive, unsustainable technologies to remediate waste water, widely considered a non-income generating blight on environmental and public health, sustainable strategies that valorise key by-products of waste water remediation should be implemented. For IAPS, these by-products include but are not limited to treated water, methane, hydrogen, and biomass.

Mambo, P. M. 2013 Simplicity – the key to sanitation sustainability. *The Water Wheel* **12**(6), 36-37.

There is universal acknowledgement of inadequate access to sanitation in the developing world. According to the World Health Organization (WHO) 115 people every hour succumb to preventable illnesses aggravated by poor sanitation, hygiene and water contamination. Independence in 1994 revealed that 20.5 million South Africans were without basic sanitation. To eliminate this exigency the Water Supply and Sanitation White Paper was introduced by the government. However, hindrances attributed primarily to poverty, underemployment and high operational costs incurred by bulk water supply and sanitation schemes ensured the previously marginalized poor were unable to pay service delivery charges. Governmental efforts, primarily the commissioning of 2410 water and sanitation projects by the Department of Water Affairs and Forestry (DWAF) culminating in an audit by the Council for Scientific and Industrial Research (CSIR) in 2007 revealed approximately 60 % were incomplete or not operational. The aforementioned concerns thus served as primary drivers for the National Waste Management Strategy (NWMS) of the *Waste Act* (2008) of South Africa, which in summary mandates minimizing pollution, environmental degradation and the consumption of natural resources in an effort to conserve access for future generations. It further mandates implementing waste hierarchy, balancing the need for ecologically sustainable development with economic and social development, while promoting universal and affordable waste services (Republic of South Africa, *Waste Act* 2008). Waste water is environmentally detrimental due to its nutrient rich, copious nature and frequency of generation. This causes the proliferation of detrimental conditions such as the explosion of microorganism and plant populations, to the detriment of endemic flora and fauna. Microorganisms in domestic waste water are enteric and generally disease causing thus it is in the interest of governments to implement sustainable solutions for the remediation of waste water.

At present greater than 80% of South Africa‘s waste water treatment works are in disrepair, underperform or are overloaded. Further, South Africa has experienced the deployment of ill-advised, energy/cost/expertise expensive, ‗advanced‘ technology choices that have proven unsustainable in the long term. The reality is that technologies like reverse osmosis and desalination schemes are better suited to the developed world where the capital resources, infrastructure and expertise are pre-established. Thus, employment of foreign technologies ensures employment of foreign expertise for construction, operation, maintenance and eventual training, while the implementation of simple indigenous technologies ensures the employment creation and up-skilling of the immediate community where the system is deployed. Integration of pre-existing knowledge and technology awareness will ensure system adoption and that the communities take ownership of their sanitation management resulting in improved, cost effective, relatable and consistent service delivery as the communities become self-accountable.

Rapid implementation of robust, easy to deploy and operate waste water treatment technologies is urgently required. Furthermore, climate change together with reduced water availability has major food security implications for South Africa, its neighbours and other arid, water-poor countries. These factors pose profound management implications for both government and business. Correct implementation and management of IAPSs optimized for South African conditions can produce clean water for recycle and reuse, provide energy, and generate a biomass suitable for valorization. Even so, and as with any near market-ready technology, there is an element of risk and/or failure to comply. Conceptualized by the late Prof Bill Oswald of the University of California in Berkeley, a staunch advocate of sustainable development and access to sanitation for all, the IAPS streamlines conventional waste water treatment technologies to essentially remediate waste water biologically. He combined the primary facultative pond with an in-pond anaerobic digester and enhanced the efficiency of waste stabilization, facultative and maturation ponds by introducing a raceway with a paddle wheel to evenly mix and polish waste water while reducing the usual retention time within ponds. This resulted in reduced retention times throughout the system in comparison to other conventional treatment technologies, while the construction of the an in-pond digester underground ensured the temperature required by anaerobic microorganisms remained constant regardless of the influent temperature. This ensured the global applicability of the technology.

So what sets the IAPS apart from other waste water remediation technologies?

This technology can be implemented where a waste water treatment system is most needed. It can be operated and maintained by the community in which it is deployed as minimal skill is required for system operation and maintenance. The system can further and most importantly be retrofitted to support pre-existing overburdened and under-performing technologies improving their efficiency and reliability. Land is not limiting in South Africa and in most parts of Africa thus this technology can easily be implemented and sustainably deployed. The technology can further be retrofitted to suit the type and quantity of waste water requiring remediation. With simple amendments to the components of the system, it can easily generate methane gas for heating and cooking and when passed through a generator generate electricity. Thus this system can operate off the electrical grid, while generating an effluent stream suitable for irrigation in agriculture and cattle rearing. The algae biomass, generated in copious amounts can serve as a bio-fertilizer, a substrate for the anaerobic digester and further is responsible for carbon sequestration which mitigates carbon dioxide emissions into the environment. The IAPS also has the potential to decrease the costs involved in capturing more water and alleviating the demand on already strained and untapped environmental water reserves. It further reduces the costs involved in treating more water, most notably in terms of the energy required to harvest the water and to divert it to where it is most required.

Shortcomings of the IAPS as deployed at the Belmont Valley waste water treatment plant

The IAPS at the Belmont Valley waste water treatment plant was constructed to supplement the remediation capability of the of the trickling filter system deployed by the municipality. Thus, there is a pump that diverts 10 % of the incumbent waste water from the plant into the IAPS. Therefore when the pump is not operational, there is no influent into the IAPS until the pump is running. The IAPS was designed to remediate waste water to a secondary level, which has in the past not been emphasized. Thus the IAPS at the Belmont valley waste water treatment plant does not disinfect to a level compliant with the standards set forth by the Department of Water Affairs (DWA) for discharge into the environment primarily due to the lack of a tertiary treatment component, which should not imply that that the IAPS cannot be designed to generate a specific effluent quality. Rather, with the addition of a tertiary treatment unit and supplementary UV disinfection/chlorination/flocculation, the effluent from the system can be utilized for domestic purposes like flushing toilets, lawn irrigation, washing cars, showers and in the case of agriculture, irrigation. Another shortcoming of the present configuration is that the anaerobic digester was not designed for methane harvesting. Had the system been configured to harvest methane, the waste water treatment plant would be operating off the grid and would concomitantly be better equipped to place itself as a sustainable self-sufficient technology. However future designs could easily amend this shortcoming ensuring that even urban and rural communities would have constant, renewable, reliable and carbon neutral electricity generation as sources of energy like wood and fossil fuels are environmentally and health detrimental.

The current paddlewheel design deployed at the Belmont Valley waste water treatment plant often disconnects from the shaft resulting in impaired waste water remediation. This has implications for the quality of water effluent discharged from the IAPS. In actuality at times no effluent is discharged from the system due to a lack of sufficient propulsion from a functional paddlewheel as a result the water in the raceway may stagnate. Another concern is that the paddlewheel utilizes electricity to turn, it would be preferable if it either utilized solar energy or methane gas captured from the in-pond anaerobic digester as once there is a power cut both paddlewheels are not operational resulting in an effluent stream that is partially remediated and not suitable for environmental discharge. Of greatest concern is that the current raceway design is flawed in that only 50 % of the final effluent is remediated. This should have been amended by either doubling the size of the second raceway or adding a third raceway to increase the retention time in the system, which currently results in the final effluent not being suitable for environmental discharge.

The world is edging closer to a scenario where water will have to be recycled and more efficiently utilized in order for sustainable and adequate access for all, most especially as water is a right and not a privilege. Education is vital to ensure that water utilization and remediation costs are reduced to ensure an adequate supply for all. Sustainable remediation of waste water is imperative in a drought prone, water stressed country like South Africa. Technologies like the IAPS carefully considered these ever pertinent needs and comprehensively addressed them by generating a treated water, energy and a biomass suitable for agricultural applications, resulting in a system which can be tailored to suit the regions and the needs of the community where it is deployed, ensuring access to sanitation for all and resulting in reduced preventable sanitation and water contamination related disease incidences. The system has been shown to be cost efficient, versatile, deployment ready, sustainable, robust, environmentally safe, sustainable, green and capable of facilitating up-skilling in relatively remote communities. Land is available in South Africa and the climate of conducive for efficient year round remediation. So, why not, this system can serve as the stepping stone for improving the quality of life for all.

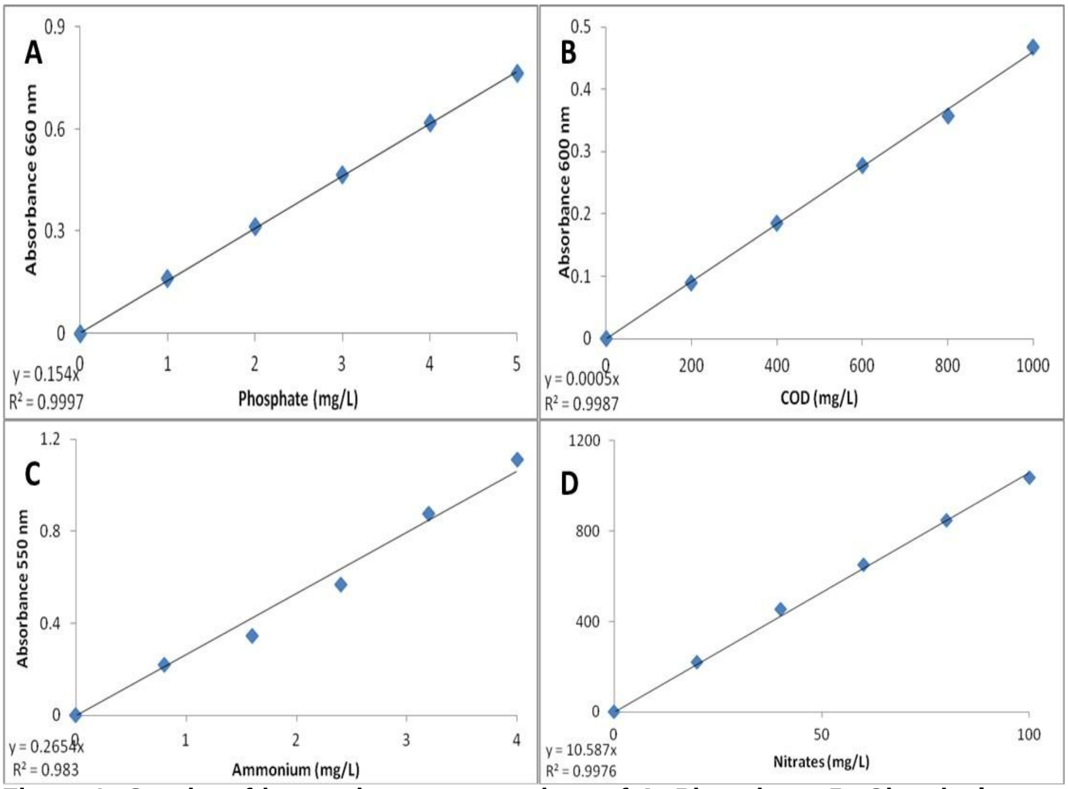

Figure 1: Graphs of increasing concentrations of A: Phosphate; B: Chemical oxygen demand (Potassium hydrogen phthalate); C: Ammonium; D: Nitrate which were used to interpolate the respective concentrations in the final water stream discharged from the IAPS.

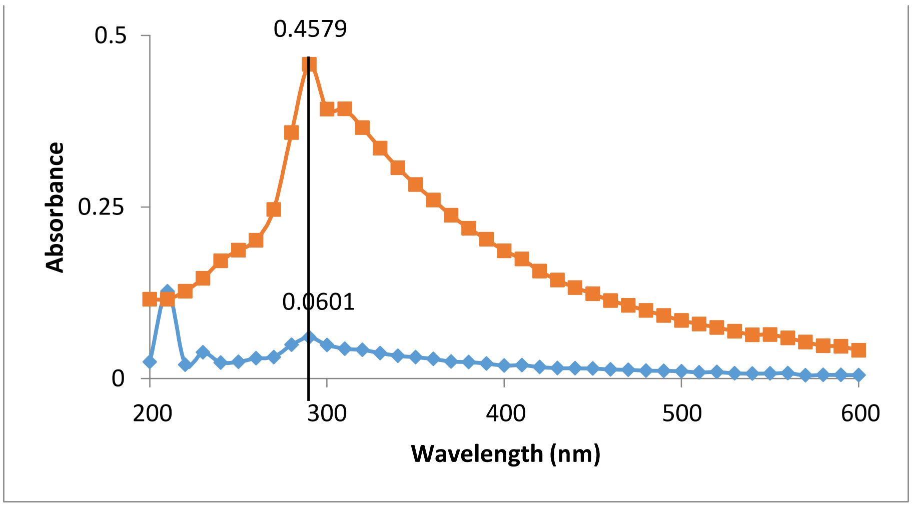

Figure 2: A graph generated using a Shimadzu UV-Vis spectrophotometer (Kyoto, Japan) for the comparison of the absorbance readings detected utilizing 2 and 20 mg.L^{-1}of humic like substance standards over a wavelength range between 200-600 nm.

Overseeing the general operation of the IAPS technology

Majority of the operational problems concerning the IAPS can be mitigated through observation and diligence. At the, start and end, of the day, the pump that channels water into the system needs to be checked for debris and other obstructions to ensure that it is working properly. Following a visual assessment the pump may need to be cleaned. If this is the case, the pump needs to be switched off, cleaned and then replaced. However if there are no obstructions wastewater should evenly flow into and through the IAPS. Wastewater should flow easily through the system, however once a pipe becomes blocked, to prevent bursting, the pump should be switched off, the blockage timeously removed and the pump restarted. Ideally within a commercial facility, following a blockage, water should be diverted into a storage pond until the system is fully operational and to prevent further damage. Presently, the pipes within the pilot IAPS are generally easily removed and replaced. Splitter boxes which channel wastewater from component to component need to remain covered to prevent entry of plant debris which may cause obstructions in the system. Components of the system, namely the algae settling ponds and the drying beds require emptying intermittently to prevent biomass build up. The system does not require sludge handling. Daily maintenance can be easily conducted by following the protocol listed below:

1) Check the pump at the inlet works is operational ensuring even flow into the IAPS, rainfall may cause pump malfunctions
2) Remove the AD splitter box cover (all splitter boxes throughout the system should be cleaned once a week)
3) Check for and remove debris with a stick
4) Ensure even flow into the splitter box
5) Replace the cover
6) Check the pipe leading into the AFP and IPD
7) If below the surface of the AFP
    a. Stop the pump at the inlet works
    b. Remove the pipe from its shaft
    c. Clean the pipe using a stick, to dislodge any blockages
    d. Replace the pipe and restart the pump
    e. Check the flow
    f. The pipe should remain slightly above the surface of the AFP
8) Remove the HRAOP splitter box cover
9) Check for and remove debris with a stick
10) Ensure even flow into the HRAOP splitter box
11) Replace the cover
12) Check that the paddlewheel is turning, if not
    a. Switch off power to the gear box
    b. Check that the paddlewheel is secured on the shaft, if not move it back into place
    c. Check for any cracks that may require welding
    d. Remove any rocks and pebbles that may have displaced the paddlewheel
    e. Turn on the gear box to check if it is operational otherwise it may require repairs
13) Conduct same check for subsequent raceways
14) Depending on rate of algae biomass build up, ASPs may need to be emptied weekly or once every two weeks

a. If algae sludge build up is rapid then it is imperative to clean the ASP more often than once a week otherwise the water quality leaving the IAPS will decline
b. ASPs are emptied easily using a pump, cleaned then refilled using water from the raceways

15) ADBs are where the algae sludge from the raceways is discarded, these can be emptied twice a year depending on quantities of sludge generated by the IAPS
a. This biomass can be diverted into the AD for methane generation b. This biomass can act as a soil amendment or slow release fertilizer

Composite Sampling

Composite sampling is regarded as the most suitable method to give an appropriate value for a performance criterion in the assessment of a final effluent from a WWTW. The IAPS final effluent is continuously pumped using an electric pump into a collection vessel which is allowed to fill for 24 h following which the composite sample is stirred with a rod and a 500 mL sample abstracted using a sterile acid washed Schott bottle.

Figure 3 The setup of the collection of the composite sample abstracted over a 24 h period. An aliquot of the sample following mixing is then analysed in the laboratory for its physical, chemical and biological characteristics.

Subsampling - A 500 mL Schott bottle is rinsed using the final effluent in the collection vessel. After discarding this was a Schott bottle is filled to the brim and then taken to the laboratory for analysis.

Water Sample Analysis

Physicochemical analyses - Electrical conductivity and temperature measurements may be taken using an EC Testr 11 Dual range 68X 546 501 detector (Eutech Instruments, Singapore), while the dissolved oxygen content is determined using an 859 346 detector (Eutech Instrument, Singapore). The pH can be measured using a Hanna HI8 424 microcomputer pH meter (Hanna Instrument, Romania). Merck test kits analogous to the methods described in Standard Methods (1998) are generaly used to determine the nitrate-N (1.14773.0001; 20- 300 mg. L^{-1}), *ortho*phosphate (1.14848.0001; 0.01- 5 mg. L^{-1}), ammonium-N (1.14752.0001; 0.013- 3.86 mg. L^{-1} and chemical oxygen demand A and B (1.14538.0065 and 1.14539.0495; 100- 1 500mg.L^{-1}) concentrations of the final effluent generated by the IAPS, according to the manufacturer‘s instructions. An Aquamate Thermospec AQA 102 708 (Helios Aquamate, England) was used to interpolate the concentrations of the samples from the respective standard curves. It should be noted that

Whatman Grade 1 filter paper (1 002 125) with a pore size 11 µm was used to remove algae from the filtered COD, as the algae in the final IAPS effluent range in size from >10 µm and the DWA standard stipulates the complete removal of algae.

Microbial analyses - MacConkey (HG0 00C 92 500) and m-Fc (HG0 0C 120 500) agar (Biolab, South Africa) are made up according to the manufacturer‘s instructions. A hundred µL aliquots were used to spread plate the sample onto the agar. The Petri dishes are then stored at 30 ºC and 45 ºC respectively for 24 h. And appropriate indicator colonies counted.

Total suspended solids - A pre-weighed Whatman Glass Fibre A (1 820 110) filter paper with a pore size 1.6 µm is placed in an oven at 105 ^{0}C for 1 h. Following which 20 mL of sample is filtered through the Glass Fibre filter paper, 20 mL milli-Q water served as the blank. The filter paper is then oven dried overnight at 105 ^{0}C according to Standard Methods (1998) and TSS determined as follows;

$$\text{mg Suspended solids/L} = ((A-B) \times 1000) / \text{mL sample}$$

where A = weight of filter (or filter and crucible) + residue in mg, B = weight of filter (or filter and crucible) in mg, and C = mL of sample filtered.