PERFORMANCE EVALUATION AND COST ANALYSIS OF SUBSURFACE FLOW CONSTRUCTED WETLANDS DESIGNED FOR AMMONIUM-NITROGEN REMOVAL

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Abstract

Subsurface flow constructed wetlands (SSF CWs) is a low-cost, environmentally friendly sanitation technology for on-site treatment of domestic/municipal sewage. However, these systems are apparently unable to produce treated water of a quality suitable for discharge particularly in terms of nitrogen concentration, which has been attributed to design and operation based on biological oxygen demand as the parameter of choice. The aim of this study was to evaluate the performance, support medium, and techno-economics of a vertical-horizontal (V-H) SSF hybrid CW designed and operated using ammonium-nitrogen (NH_4^+-N) as the major parameter. Two pilot scale V-H SSF hybrid CWs were designed, constructed, and the performance of each monitored over two seasons and under two phases i.e. an initiation phase, and an optimization phase. Laboratory-scale horizontal SSF CWs were used to evaluate the support medium while the techno-economic study was framed to determine the cost effectiveness of V-H SSF hybrid CWs relative to high rate algal oxidation pond (HRAOP) systems to increase capacity of overloaded and/or under-performing waste stabilization pond (WSP) sewage treatment plants. Results revealed that under optimal operating conditions of hydraulic loading rate, hydraulic retention, and influent NH_4^+-N loading rate, treated water from the V-H SSF hybrid CWs achieved a quality commensurate with current South African standards for discharge into a surface water resource for all parameters except chemical oxygen demand and faecal coliforms. This suggests that NH_4^+-N is an important design and operational parameter for SSF CWs treating municipal sewage that is characterised as weak in terms of NH_4^+-N with a requirement of only simple disinfection such as chlorination to eliminate faecal coliforms. Use of discard coal to replace gravel as support medium in horizontal SSF CWs revealed an overall reduction in elemental composition of the discard coal support medium but without compromising water quality. This result strongly supports use of discard coal as an appropriate substrate for SSF CWs to achieve acceptable water quality. Furthermore, simultaneous degradation of discard coal during wastewater treatment demonstrates the versatility of SSF CWs for use in bio-remediation and pollution control. Finally, a techno-economic assessment of V-H SSF hybrid CWs and a HRAOP series was carried out to determine the suitability of each process to increase capacity by mitigating dysfunctional and/or overloaded WSP sewage treatment plants. Analysis revealed that the quality of treated water from both systems was within the South African General Authorization standards for discharge to a surface water resource. Even so, each technology system presented its own set of limitations including; the inability to satisfactorily remove NH_4^+-N and chemical oxygen demand (i.e. for V-H SSF hybrid CWs) and total suspended solids and faecal coliforms (i.e. for

HRAOPs), and a requirement for substantial land footprint while, HRAOPs required significantly less capital than V-H SSF hybrid CWs for implementation. The latter suggests that HRAOPs could be preferred over V-H SSF hybrid CWs as a technology of choice to increase the capacity of overloaded WSP sewage treatment plants especially where financial resources are limited. Overall, the results of this thesis indicate the potential to use NH_4^+-N as a design parameter in constructing SSF CWs treating weak strength municipal sewage (i.e. in terms of NH_4^+-N concentration) and to supplant gravel as the treatment media with industrial waste material like discard coal to achieve wastewater treatment, bio-remediation, and pollution control. The results of this work are discussed in terms of using SSF CWs as a passive and resilient technology for the treatment of domestic sewage in sub-Saharan Africa.

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Abbreviations

AFP	Advanced facultative pond
ANAMMOX	Anaerobic ammonia oxidation
APHA	American public health association
ASP	Algal settling pond
ASS	Activated sludge system
BNR	Biological nitrogen removal
BOD_5	5 day-biological oxygen demand
CFU	Colony forming units
COD	Chemical oxygen demand
CRF	Controlled rock filters
CW	Constructed wetland
DO	Dissolved oxygen
EC	Electrical conductivity
EU	European Union
FC	Faecal coliforms
GWP	Global warming potential
HLR	Hydraulic loading rate
HRAOP	High rate algal oxidation pond
HRT	Hydraulic retention time
IPCC	Intergovernmental panel on climate change
ISO	International Organization for Standardization
IWMI	International water management institute
MP	Maturation pond
MPN	Most probable number
NEMA	National Environmental Management Authority
NH_4^+-N	Ammonium nitrogen
NO_2^--N	Nitrite nitrogen
NO_3^--N	Nitrate nitrogen
O & M	Operation and maintenance
PE	Population equivalent
PO_4^{3-}-P	Phosphate phosphorus
SSF	Slow sand filter
SWM	Sustainable water management

TN	Total Nitrogen
TP	Total phosphorus
TSS	Total suspended solids
UNCEF	The United Nations Children's Education Fund
US. EPA	United States Environmental protection Agency
WSP	Waste stabilization pond
WWT	Wastewater treatment
WWTP	Wastewater treatment plant

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Chapter 1 Introduction and literature review

1.1 Background

Surface water pollution due to discharge of untreated or partially treated municipal sewage is one of the major environmental problems facing the world today (Abdel-Raouf et al., 2012). Municipal/domestic sewage contains a myriad of macro- and micro-pollutants comprising of, but not limited to, nutrients (nitrogen & phosphorus), organic matter (biological oxygen demand (BOD) and chemical oxygen demand (COD)), total suspended solids (TSS), pathogens, metal ions, pesticides, pharmaceuticals, personal care products and endocrine disruptors (Tchobanoglous et al., 2003; Vidal-Dorsch et al., 2012; Petrie et al., 2017). Presently, among the pollutants contained in wastewater, nutrients particularly nitrogen is of great concern due to its adverse effects both on the environment and public health (Akpor & Muchie, 2011; Petrie et al., 2015).

Among the various nitrogenous compounds that exist, the most important inorganic nitrogen species are nitrates (NO_3^-), nitrites (NO_2^-) and ammonium (NH_4^+) (Vymazal, 2007). The use of nitrate-contaminated drinking water to prepare infants' food is for instance a known cause for infant methemoglobinemia (blue baby syndrome) (Knobeloch et al., 2000). If not properly treated, discharge of municipal wastewater effluent with high ammonium concentration also affects aquatic life. Un-ionised ammonia even in small concentrations is reported to be toxic to many aquatic life forms including fish. Additionally, wastewater rich in nitrogen results in excessive growth of aquatic plants a phenomenon referred to as eutrophication (Wetzel, 2001; Akpor & Muchie, 2011; Petrie et al., 2015). The effects of eutrophication on the surface water resources include inhibition of light penetration into the water body and hence, a reduction in algal productivity which results in depletion of oxygen. Lack of oxygen leads to massive fish kills and the demise of other aquatic organisms e.g. bacteria, zooplankton, protozoa, insects etc. that require oxygen for survival (Wetzel, 2001). Loss of aesthetic value of some water bodies and increased costs of treating drinking water due to algal blooms has also been reported (Smith et al., 1999; Ansari & Gill, 2014). To overcome the effects of eutrophication on the receiving water bodies, it is required that municipal wastewater must be treated to certain standards before disposal into the environment. Hence, environmental regulating bodies around the world set-up effluent discharge standards for wastewater treatment plants (WWTPs) for disposal of treated effluent into surface water resources. For example, some nitrogen effluent discharge standards for municipal WWTPs from selected countries in sub-Saharan Africa are shown in Table 1-1.

Table 1-1. Discharge standards nitrogen species for municipal WWTP from selected countries in sub-Saharan Africa.

Parameter	**General standards**				
	South Africa[1]	**Uganda[2]**	**Zimbabwe[3]**	**Tanzania[4]**	**Kenya[5]**
NH_4^+-N (mg/L)	6	10	0.5	10	1
NO_3^--N+NO_2^--N (mg/L)	15	22	3	20	1

Source: [1]= DWS (2013), [2]= NEMA-Uganda (1999), [3]= Nhapi (2004), [4]= NEMA-Tanzania (2007), [5]= NEMA-Kenya (2003).

To meet effluent discharge standards for nitrogen, various wastewater treatment technologies are adopted depending on the economic region. In the developed world for instance, advanced wastewater treatment systems particularly activate sludge systems (ASS) with biological nitrogen removal (BNR) are the most adopted wastewater treatment system due to the high standards and strict regulations for nitrogen removal (Tchobanoglous et al., 2003). Although very effective in nitrogen removal (Yang & Zhang, 1995), AS systems require chemicals and are associated with sophisticated mechanical equipment which requires high fossil fuel-derived energy and highly skilled personnel. The need for chemicals, spare parts, constant energy supply and skilled labour results in high operation and maintenance costs, which, limits the use of AS WWT technology in most developing countries (Mahmood et al., 2013). Therefore, countries in tropical and sub-tropical regions choose natural systems, which are typically powered by sunlight (Zhang et al., 2015). These include conventional waste stabilization ponds (WSPs), constructed wetlands (CWs), bio-filtration (BF) and integrated algae pond systems (IAPS) to mention a few (Kayombo et al., 1998; Mahmood et al., 2013; Mambo, 2016). According to Kayombo et al. (1998) and Mahmood et al. (2013), wastewater treatment by these systems relies mainly on the interaction of bacteria and algae and or macrophytes utilizing sunlight as a major source of energy to remove pollutants from wastewater. The resultant biochemical and physical processes which occur in a more natural environment rather than tank reactors, result in a system that is passive, consumes less energy and requires less operation and maintenance and hence, the overall costs are lower (Smith, 1989).

Among the natural treatment systems, WSPs are the most commonly adopted wastewater treatment technology for treatment of both domestic and municipal sewage in tropical and subtropical regions. Unlike AS, WSPs are not associated with any sophisticated mechanical equipment, which reduces the requirement for skilled personnel and there is little or no fossil fuel-derived energy or chemical requirement (Kayombo et al., 1998). However, the major limitation associated with WSPs include:

- A requirement of relatively large land footprint (Alexiou & Mara, 2003; Mara, 2003).
- Long hydraulic retention time ranging from days to months (Mara, 2003; Shilton, 2006).
- Susceptibility to mosquitoes and breeding of other vectors and production of unpleasant odours (Mara, 2003).
- Sludge accumulation, which is higher in colder climates (USEPA, 2002b) and a need to desludge at least once every 2 to 3 years (Alexiou & Mara, 2003).
- Very narrow zone for nitrification, since the aerobic zone is restricted to the upper 50 cm (Baskaran et al., 1992) and the lower nitrifier biomass in the water column (McLean et al., 2000) attributed to a small surface area provided by algae and duckweeds for attachment of nitrifying bacteria (Zimmo et al., 2000; Babu, 2011). Thus, it is difficult to control or predict nitrogen effluent quality from WSP effluent (USEPA, 2002b). In fact, for these reasons, several studies have reported that the treated water from WSPs does not meet nitrogen effluent discharge standards (Babu, 2011, Mburu et al., 2013).

To protect surface watercourses from eutrophication, more appropriate and sustainable wastewater treatment technologies are required especially in regions where stringent effluent discharge limits for nitrogen exist. Thus, CWs have been proposed and approved as a wastewater treatment system for adoption especially in developing countries (Kivaisi, 2001; Mthembu et al., 2013). Indeed, CWs are considered a less costly wastewater treatment system and environmentally friendly sanitation alternative to conventional WSPs for on-site wastewater treatment by small communities (Rousseau et al., 2004; Massoud et al., 2009).

Among the different types of CWs available (Vymazal, 2005b, Vymazal, 2007, Sayadi et al., 2012), subsurface flow (SSF) CWs are widely accepted for treatment of wastewater due to their health and environmental benefits (Kayombo et al., 1998, Vymazal, 2010). The SSF conditions prevent odours and breeding of mosquitoes and other vectors especially in tropical and subtropical regions (Kayombo et al. 1998). Furthermore, SSF CWs have been reported to be very efficient where compliance with chemical oxygen demand (COD), biological oxygen demand (BOD), total suspended solid (TSS) and total phosphorus (TP) effluent standards is required (Vymazal, 2005b; Mburu et al., 2013). However, efficient nitrogen removal continues to be challenging where either horizontal subsurface flow (HSSF) or vertical subsurface

(VSSF) CWs is used independently. The anaerobic conditions that prevail in HSSF limit nitrification (Vymazal, 2005b) while this process is accomplished in the VSSF producing nitrates, which are detrimental to the environment (Brix & Arias 2005). Therefore, to achieve maximum nitrogen removal from CW, vertical-horizontal (V-H) SSF hybrid CWs have been proposed as a promising wastewater treatment technology with the major objective to; maximize nitrogen removal through nitrification (in the VSSF) and denitrification (in the HSSF) processes (Vymazal, 2013). Nevertheless, several studies on the performance of V-H hybrid CWs reveal that NH_4^+-N effluent quality does not comply with the effluent discharge standards of most environmental regulating bodies (Keffala & Ghrabi, 2005; Abidi et al., 2009; Herrera Melián et al., 2010; Foladori et al., 2012), which could be attributed to design limitations.

Literature shows that optimum performance of a CW depends on using an appropriate area for a given organic load (Ulsido, 2014). Because of this requirement, different methods have been proposed for sizing the effective area of SSF CWs including the population equivalent (PE) method, pollutant loading method, and non-mechanistic models (Tousignant et al., 1999; UN-Habitat, 2008). Nevertheless, non-mechanistic models including the Kickuth (1977) and Reeds et al. (1995) models are widely used for estimating the area required for effective performance of a SSF CW. The drawback of using the Reeds et al. (1995) model for estimating the area of CWs is that over the years, wetland designers have employed BOD as a critical parameter for remediation. Moreover, using BOD under-estimates the area required to treat wastewater for a given PE; hence, this method is appropriate for organic matter and TSS removal but not appropriate for nutrient removal (Vymazal, 2005). Thus, and as pointed out by Huang et al. (2000), nitrogen removal by a subsurface flow CW is an important design criterion even though it has not been fully explored. Until now, there is no published information on the performance of SSF CWs designed using nitrogen as a critical parameter, and it is not clear whether CW designed for nitrogen removal will meet the BOD effluent discharge limits.

While the reason for using BOD as a parameter for CWs treating municipal wastewater is not fully addressed by previous studies, it can be argued that in Europe where CW systems originated, effluent discharge standards from wastewater treatment plants (WWTP) are categorized based on the size of the population served. Thus, the standard effluent discharge limit for urban WWTPs is given in Table 1-2.

Table 1-2. Standard provisions for effluent discharge from urban WWTPs into surface watercourses. Data obtained from Blöch (2005).

Parameter	Concentration (mg/L)	Minimum % reduction*
BOD_5	25	70-90
COD	125	75
TSS	35	90

However, to protect sensitive water bodies from eutrophication, it is a requirement that WWTPs receiving wastewater for "above 10,000 population equivalent (PE)" must additionally meet nutrient removal standards (Table 1-3).

Table 1-3. Additional provisions effluent discharge from urban WWTPs into surface watercourses. Data obtained from Blöch (2005).

Parameter	Concentration (mg/L)	Minimum % reduction*
TN		
Plants of 10,000-100,000 PE	15	
Plants of > 100,000 PE	10	70-80
TP		
Plants of 10,000-100,000 PE	2	
Plants of > 100,000 PE	1	80

Therefore, since most CWs in Europe are designed for rural setting serving populations normally below 10,000 (Vymazal, 2010), removal of nutrients is not a requirement which is the major reason as to why these CWs are designed based on BOD. However, in most sub-Saharan African countries, the effluent discharge standards are general for all pollutants regardless of the population served by the WWTP (Appendix 1). In this case, it could be necessary to prioritise the parameter to be used while designing a CW depending on the treatment objective. Thus, if the effluent from the CW is to be discharged into a fragile water body that is susceptible to eutrophication, in my opinion, nitrogen could be the preferred nutrient to be employed in the design. Thus, by doing so, Reeds et al. (1995) argument that the design of CWs should be based on the treatment objective will be fulfilled. The reason for using nitrogen in preference to other parameters is that according to Petrie et al. (2015), the effects of nitrogen pollution on water resources and public health seem to greatly outweigh those resulting from BOD pollution and considering that phosphorous concentration is usually very low in domestic wastewater (typical value for untreated domestic wastewater is 6 mg/L) (Spellman, 2013).

The use of nitrogen as a design parameter for CWs could potentially be ideal especially where the wastewater being generated contains higher concentrations of nitrogenous compounds than organic matter. Besides domestic/municipal sewage, CWs are applied to treat various wastewaters including industrial wastewaters (Vymazal, 2014). Some industries for instance, produce wastes rich in nitrogenous compounds e.g. fertilizer plant wastes (Table 1-4) and probably amino acid producing industries whose wastewater characteristics is not known. Therefore, in instances where wastewater contains nitrogenous compounds as major water quality parameters, it is suggested that nitrogen should be the design parameter for consideration where a CW is chosen as a wastewater treatment system.

Table 1-4. Average characteristics of wastewater from an Indian fertilizer plant producing both nitrogenous and phosphatic fertilizers.

Parameter (unit)	Value
pH	7.5-9.5
Total solids (mg/L)	5400
NH_4^+-N (mg/L)	700
Urea-N (mg/L)	600
PO_4^{3-} (mg/L)	75
Arsenic (mg/L)	1.5
Flouride (mg/L)	15

Source: http://www.gitam.edu/eresource/environmental/iwm_tsrinivas/fP_waste.htm#4.

While the use of nitrogen as a design parameter for CWs could be more important than BOD, the use of BOD by most CW designers could be that, while some engineers may be knowledgeable about CWs, some engineers, policy and decision makers not only in the developing world but also in the developed nations may lack basic information about CWs. The following section therefore presents an overview of literature pertaining to CWs.

1.2 Constructed wetlands for domestic/municipal wastewater sewage treatment

Constructed wetlands for wastewater treatment are engineered systems that are planned, designed and constructed to imitate natural wetland systems utilizing natural wetland processes including wetland plants, soil, and associated microorganisms to remove contaminants from wastewater effluent in a controlled environment (Vymazal & Kröpfelová, 2008). This implies that like WSPs, CWs don't require chemical and energy inputs resulting in lower O & M costs than AS. However, although CWs require a large footprint like WSPs (Zapater-Pereyra et al., 2014), they are believed to serve as a better alternative than WSPs for municipal/domestic wastewater treatment due to the following reasons:

Firstly, CWs produce minimal sludge compared to WSPs. Unlike in the WSP system where sludge is accumulated from the primary-secondary-tertiary treatment units, as secondary treatment systems, CWs do not produce sludge. This is important in two ways: (i) No extra land requirement for sludge disposal and (ii) minimizes the maintenance costs and operations since desludging is only required for the primary treatment unit; thus, a more reliable effluent quality is expected. This point will be elaborated later in this chapter.

Secondly, CWs can be established at the source of waste generation hence can be designed and adopted to suit small communities such as individual households, farms, small industries, schools or even hospitals. This would save costs for sewage collection and disposal currently experienced with the WSPs while minimizing disposal of sewage into the environment.

Besides wastewater treatment, CWs can be designed for benefits such as wildlife habitat and recreational purpose (US.EPA, 2000). However, animal waste may contribute additional pollutants particularly nutrients and faecal coliforms thus compromising the performance of the CW. Therefore, if the aim of the CW is to treat domestic wastewater with an objective of meeting the effluent discharge standards of a given country, it would be appropriate not to mix the wastewater treatment objective with the extra benefits functions of the CW. Furthermore, CWs may provide educational opportunities when artistically designed and constructed at homesteads, schools and hospitals or even in municipal parks using ornamental plants (IWMI, 2006). This makes the wastewater treatment more appealing to the community especially when ornamental plants such as *Cyperus sp.* are used. An illustration of such an application was demonstrated at Koh Phi Phi Island in Thailand; where the flower and butterfly park is a major tourist attraction illustrating a new and/ or different method to wastewater treatment (Figure 1-1). In this case, the income from tourists may help to cover the costs for maintaining the site.

Constructed wetlands can also be used as sources of income for rural and urban communities particularly when a macrophyte that benefits a community e.g. *Phragmites sp* and *Cyperus sp* are used while at the same time making wastewater treatment cheaper in terms of O & M. For example, in a conversation with one of the residents of Grahamstown, Mr Baba noted that stems of *Phragmites australis* are used for making fences and sealing houses (Olwethu Baba, personal communication, 20 July 2014) while *Cyperus textilis* (locally known as imisi in isiXhosa) is used for making mats, baskets in many rural communities in South Africa.

Figure 1-1. Flower and butterfly constructed wetland system at Kho Phi Phi in Thailand. Image obtained from Brix et al. (2011).

So, under minimal supervision, the individuals willing to harvest the vegetation for making such profitable goods may be assigned duties of operating and maintaining the wastewater treatment works such as cleaning the inlet and outlet works, checking flow and cutting grass around the treatment facility among others.

Studies report that besides municipal/domestic wastewater treatment, CWs can be applied to treat various wastewater types. These include acid mine drainage (Sheridan et al., 2013), agricultural runoff (Tyler et al., 2012), landfill leachate (Bialowiec et al., 2012), abattoir wastewater (Odong et al., 2013) and pulp and paper mill wastewater (Abira, 1998). Therefore, with the application of CW technology, there is increased chance to reclaim water from various wastewater sources for use for different purposes including irrigation of farmlands and golf courses, flushing toilets, washing vehicles and pavements especially in water stressed countries like South Africa. Thus, high quality potable water could be reserved exclusively for drinking.

Most importantly, the CW technology has positive implications for climate change. Recently, global warming - the gradual rise in earths' temperature is one of the major environmental issues. It is caused by accumulation of ozone depleting pollutants including greenhouse gasses (GHG) mainly carbondioxide (CO_2) and has been exacerbated by combustion of fossil fuels, e.g. coal, petroleum and natural gas for commercial energy production (IPCC, 2000). Besides wastewater treatment, natural wetlands are known for their CO_2 sequestration function (Mistch & Gosselink, 2000). Unfortunately, natural wetlands, a natural process that can remove CO_2 from the environment are being destroyed globally due to population pressure and urbanization. For instance, many natural wetlands are used as dumping grounds for solid waste and raw sewage in addition to being reclaimed for agricultural production and industrial development e.g. Nakivubo wetland in Uganda (Turpie et al., 2016). Due to destruction of these natural systems, their CO_2 sequestration function is consequently compromised probably leading to conclusion that the rate of CO_2 released to the atmosphere is far larger than can be balanced by the biological and geological processes that naturally remove it from the atmosphere and store it in terrestrial and marine environments (King et al., 2007). Constructed wetlands are therefore believed to serve as a better alternative than WSPs to the degraded wetlands for wastewater treatment while enhancing atmospheric CO_2 capture to prevent the cumulative effect of global warming since they have a smaller carbon footprint due to their potential to capture and store carbon for hundreds of years (Jacquot, 2008). In contrast, although microalgae are able to fix CO_2 from the atmosphere, grow faster than plants, these are easily decomposed, and do not allow long-term CO_2 storage. Hence, commercial culture of microalgae for mitigation of CO_2 has been centered on production of microalgae for bio-fossil fuels as an alternative to fossil fuels (Fernández et al., 2012). The major drawback to the commercial applications of this technology however, is the high production cost of microalgae realized when compared to petroleum (Hughes & Benemann, 1997) hence, this could be a major reason as to why even countries in the developed world have not gained interest in microalgae culture for wastewater treatment and resultant CO_2 capture. In the developed world, due to the high cost of microalgae production, microalgae production is carried out on small scale focusing on the production of high-value compounds such as pigments, food supplements for both humans and animals and fertilizers (Cuellar-Bermudez et al., 2015) whose contribution to carbon capture is low. Furthermore, climate is a major factor limiting microalgae production in developed countries. Almost the entire developed world is located outside the tropics and, in regions where both irradiance and day light length lead to long periods of near darkness with extremely low day and light temperatures. Indeed, this is a very reason that underpins modern day techno-mechanical equipment such as ASS and others for wastewater treatment.

Nevertheless, there is debate over implementation of CWs for commercial wastewater treatment for fear of production of methane due to anaerobic mineralization, which is estimated to have a global warming potential (GWP) of 28-36 over 100 years that is about 20 times more than that of CO_2 (US.EPA, 2017). On a promising note, however, advances in research provide hope for mitigation of CH_4 production through other biotechnologies. Efforts such as coupling the CW with microbial fuel cells (MFC), commonly referred to as the CW-MFCs could be one option (Doherty et al., 2015). However, while this technology has existed for about a decade, research into the CW-MFC biotechnology in the previous years has focused on improvement in wastewater treatment and bioenergy production (Villasenor et al., 2013; Zhao et al., 2013; Xu et al., 2015). Yet, studies addressing the application of CW-MFC technology in mitigating GHG production; could form a strong base for decision and policy makers for implementing a technology that is environmentally and economically sustainable.

Thus, it is probably due to the various advantages of CWs discussed above, that various campaigns aimed at addressing sustainable water management have approved CWs as one of several natural wastewater treatment technologies that can be adopted for sustainable management of water resources especially in water stressed countries. Following the drought that was experienced by South Africa in 2015, a meeting aimed at addressing challenges and solutions to the management of the water sector of South Africa was held on 27th September 2016, Emperors Palace, Kempton Park, South Africa. During this meeting, it was stated that most African cities will become more urbanized by 2030 thus increasing the demand for sustainable water and wastewater infrastructure. Thus, to address this concern, CWs were chosen for wastewater treatment in the ongoing project in Western Cape, South Africa entitled "The Water Hub" aimed at testing green infrastructure for developing confidence in water sensitive urban design (Winter Kevin, lecturer and researcher in Environmental and Geographical Science, University of Cape Town, personal communication, 27th September 2016, Emperors Palace, Kempton Park, South Africa). The reason is probably that since wetlands are known for their various functions including wastewater treatment as mentioned earlier, in cases where wetlands don't exist naturally, there is a need for these to be created. Additionally, at the International Conference on Sustainable Water Management (Perth, Australia 2015), in his key note speech, Dr Florent, an expert in CWs for wastewater treatment emphasized that, CWs have been successfully applied to treat domestic wastewater in small rural communities in Western Europe including France, Czech Republic and Austria to mention a few and have achieved acceptable performance (Florent Chazarenc, Mines Nantes, Personal communication, 2nd December 2015). Hence, CW systems can be replicated for application for domestic wastewater treatment in low-income areas and water stressed regions to protect surface and ground water resources.

Moreover, several CWs for domestic wastewater treatment are applied in various parts of the developed world including the USA (Kadlec & Knight, 1996), The Netherlands (de Jong, 1976), The United Kingdom (Cooper & Green, 1998) and in China (Liu et al., 2009) to mention a few. Much as CWs application is also reported in a few developing countries including Colombia, Egypt, India, Mexico, South Africa and Tanzania to mention a few (Wallace & Knight, 2006), the available data (Table 1-5), indicates that there is sluggishness in implementation of the CW technology especially in sub-Saharan Africa.

Table 1-5. Number of constructed wetlands in the different regions.

Continent/region	No. of CWs	Reference
North America	6000	Shi et al., 2004
Europe	1000	Shi et al., 2004
Africa (South Africa)	70	Woods, 1999

According to Denny (1997), the slowness in implementation of the CW technology in developing countries is attributed to aid programs from developed countries, which tend to favour more overt technologies that have commercial spin-off to donors. However, Verburg et al. (2006) attributed it to engineers and decision makers in the developing world and a tendency to prefer tried and tested technologies to avoid the risk that may be associated with newer technologies. Furthermore, lack of readily available information about CWs could be the other contributing factor. The following sections therefore discuss the different types of CWs, their performance, factors influencing their performance and mechanisms responsible for pollutant removal from CWs with special attention to nitrogen.

1.2.1 Categorization of constructed wetlands

Constructed wetlands are named according to their purpose. Hence, since they are mainly used for secondary or tertiary wastewater treatment, in this regard they are referred to as constructed treatment wetlands (CTW) whereas CWs designed for both wastewater treatment and other benefits e.g. wildlife habitat and recreational purpose are usually called enhancement wetlands (US. EPA, 2000). The former, which is the focus of this chapter, is mainly categorized as either free water surface (FWS) or subsurface flow (SSF) systems depending on the type of flow (Kadlec & Knight, 1996).

1.2.1.1 Free water surface constructed wetlands

Free water surface CWs are a typical representation of natural wetlands, marsh or swamp; since they are normally designed as one or more shallow basins, lined with an impermeable barrier (usually clay or geo-textile) to prevent seepage to fragile ground waters and a submerged soil layer to support the roots of the selected emergent macrophyte species. The CW is then flooded with pretreated wastewater from one side to a depth of 0.1-0.45 m above the ground level onto which floating, submerged or emergent macrophytes are rooted; the treated wastewater is then allowed to flow out from the opposite side (U.S. EPA, 2000; Vymazal, 2007).

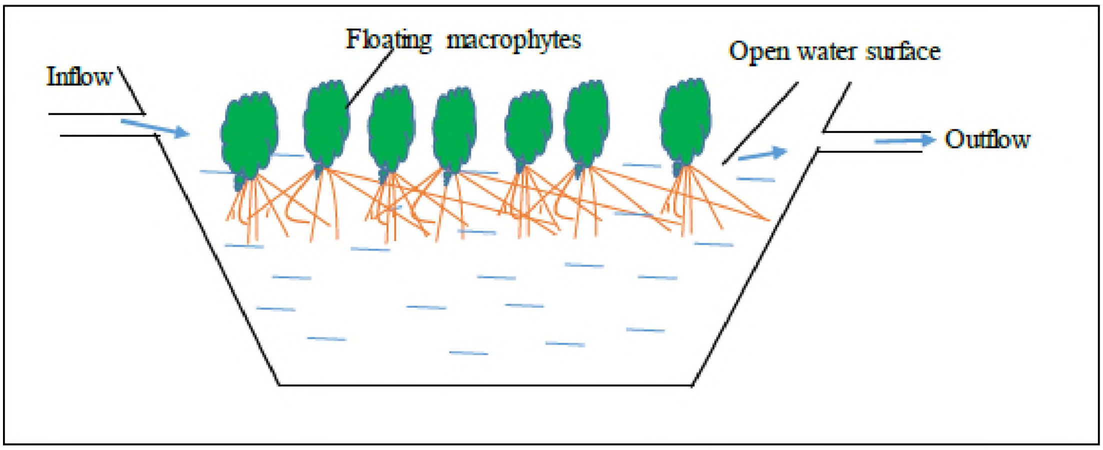

Figure 1-2. Diagrammatic representation of a free water surface flow constructed wetland. Image modified from Vymazal. (2007).

Figure 1-3. A picture of a FWS CW designed to treated storm water in Western Australia, Perth. The picture was taken during a field trip at the 2015 SWM conference in Perth, Australia.

Since wastewater in this type of CWs is exposed to the atmosphere, like WSPs, besides not meeting the effluent standards particularly for organic matter and TSS removal due to algae production within the system (US. EPA, 2000), FWS CWs encourage breeding of mosquitoes and other insect vectors. Because of these reasons, this category of CWs is not widely used for wastewater treatment especially in tropical and subtropical regions (Kayombo et al., 1998).

1.2.1.2 Subsurface flow constructed wetlands

To reduce the negative health and environmental impacts associated with FWS CWs, SSF CWs were developed and are the most widely accepted CW wastewater treatment systems around the world (Vymazal et al, 1998). Thus, SSF CWs will be discussed in greater details throughout this chapter.

Although the design and construction considerations of SSF CWs are similar to those of FWS CWs, in a SSF CW, the basin is filled with treatment media (usually sand or gravel) onto which the emergent macrophytes are rooted. Among the emergent wetland macrophytes, *Phragmites sp* (reeds), *Typha sp* (cattail), *Scirpus sp* (bulrushes) and *Cyperus sp* are the commonly used species (Okurut, 2000). The CW is then fed with pretreated wastewater, which is kept below the surface of the treatment media. While SSF CWs require less land than FWS CWs, the major limitation of this wastewater treatment technology is the higher capital cost requirement in comparison to FWS CWs attributed to cost of the treatment media (Kadlec & Knight, 1996). Even though clogging is reported as the most frequently encountered operational problem in SSF CWs (Calheiros et al., 2009; Nivala & Rousseau, 2009) and may result in poor effluent quality, this can be avoided by subjecting the wastewater to adequate pretreatment before feeding into the CW. According to Machibya & Mwanuzi (2006), a properly designed and constructed primary treatment unit for a CW should be able to remove up to 60 % of the influent BOD load at 20 ^{o}C. Additionally, operating the system at the designed surface loading rate could be important in achieving the expected effluent quality. Subsurface flow CWs are further differentiated into horizontal subsurface flow (HSSF) and vertical subsurface flow (VSSF) CWs.

A HSSF CW is operated so that pretreated wastewater "*continuously*" flows horizontally below ground through the substrate and the treated wastewater is collected on the opposite side (Brix, 1994) as shown in Figure 1-4.

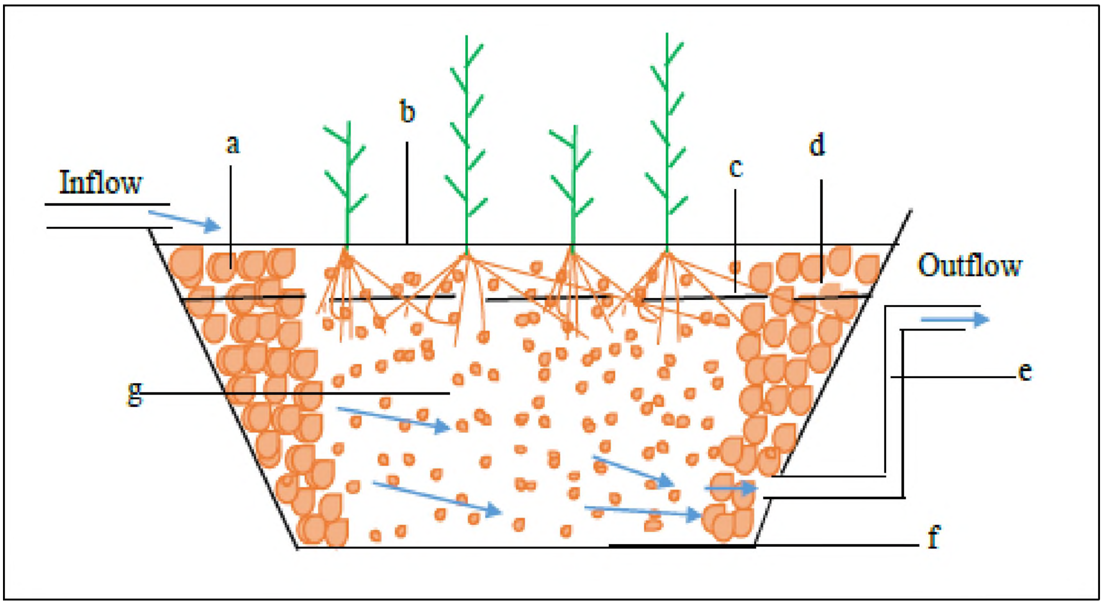

Figure 1-4. Diagrammatic representation of a typical HSSF CW. Distribution (a) and collection zones (d) filled with large gravel; treatment media level in the wetland (b); water level in the filtration bed (c); outlet structure (e) for maintaining the water level in the wetland; impermeable liner (f) and filtration zone (g), mainly gravel. Image modified from Vymazal (2007).

As opposed to HSSF CW, according to Vymazal (2008), the most common method of operating a VSSF CW is feeding it "*intermittently*" with pretreated wastewater through a network of pipes onto the entire surface of the wetland from above using a mechanical dosing system, allowing it to flow vertically through the treatment medium with discharge at the base as illustrated in Figure 1-5.

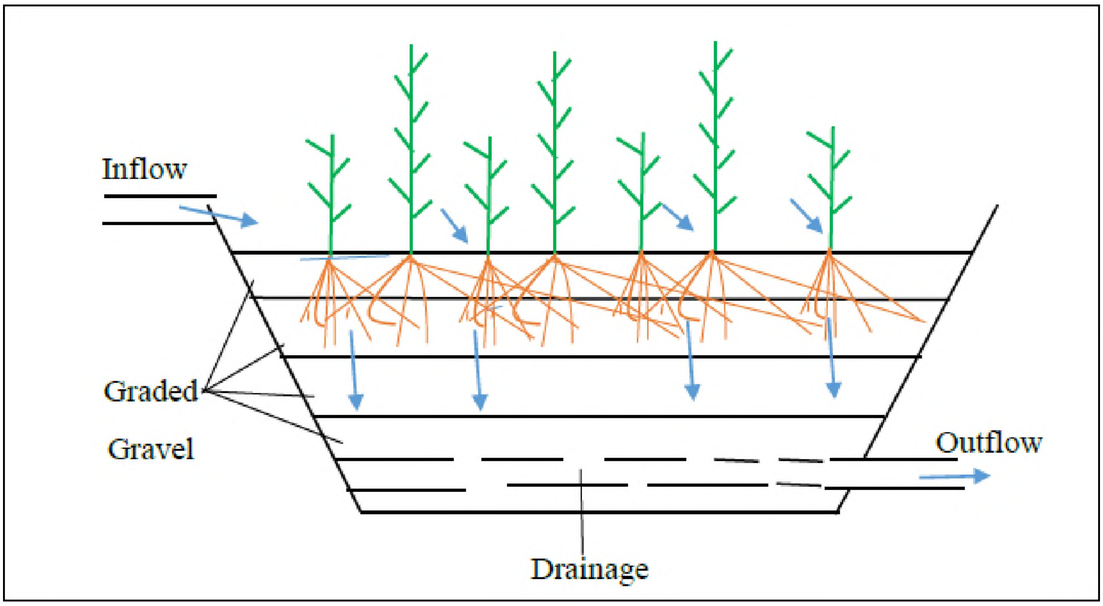

Figure 1-5. Diagrammatic representation of a VSSF CW. Gravel of different sizes is arranged in layers vertically in VSSF CWs. Image modified from Vymazal (2007).

However, "*batch*" operation of these systems was also reported (Brix & Arias, 2005). Batch feeding differs slightly from intermittent feeding. During batch feeding, the CW is fed with wastewater in pulses for a specific time and then allowed to drain completely until the next dose is applied (Caselles-Osorio & Garcia, 2007) while during intermittent feeding the CW does not completely drain before the next dose is applied (Knowles et al., 2011). It is hypothesized that batch or intermittent feeding of the CWs leads to good oxygen transfer within the bed and therefore increased ability to nitrify (Molle et al., 2008).

The limitations of operating a CW using batch or intermittent feeding is that in addition to requiring a steady energy supply, which may not be available especially in rural areas of most developing countries, the dosing system requires sophisticated mechanical equipment that may not be locally available. Additionally, since inflow wastewater is distributed through a network of pipes, there may be high risks of pipe blockage due to fluctuation in pressure because of changes in flow, which is common for any wastewater treatment system. Therefore, in addition to the general maintenance of CWs including checking the flow rate, cleaning the inlet and outlet pipes, harvesting vegetation and removing weeds among others, VSSF CWs may require additional regular O & M and in particular, checking the distribution pipes for any blockages and unblocking them if necessary. This makes operating VSSF CWs tedious and time consuming. In contrast, due to continuous feeding, HSSF CWs require little or zero energy in cases where gravity allows flow. Additionally, since continuous operation does not involve a dosing system, use of mechanical equipment is minimized hence, this mode of feeding a CW is simpler to operate and maintain (Faulwetter et al., 2009). Due to hypotheses put forward regarding nitrogen removal from SSF CWs and, that HSSF and VSSF CWs have limited ability of achieving nitrification and denitrification respectively while operating independently (Vymazal, 2005), led to the development of hybrid systems. In hybrid CWs, the advantages of various systems are combined to complement each other, thereby achieving higher treatment efficiency than a single CW, particularly for nitrogen (Sayadi et al., 2012; Vymazal, 2013; Zhang et al., 2015).

Initially, hybrid systems were introduced in the 1960s (Vymazal, 2008). Worldwide use of hybrid systems increased during the late 1990s and early 2000s, generally due to the stricter discharge limits for NH_4^+-N (Vymazal, 2008; Vymazal, 2013). The first hybrids were "*multi-stage*" consisting of several trains of VSSF systems typically planted with *Phragmites australis*, and fed with pre-treated wastewater for 1 to 2 days and allowed to dry for 4 to 8 hours. The VSSF were followed by 2 or 3 trains of HSSF systems and usually planted with multi-species of emergent plants comprising of *Iris*, *Cartex*, *Typha*, *Schoenoplectus*, *Soarganum* and *Acorus* (Vymazal, 2008). The essence of the alignment is that, in the VSSF system, oxidation of ammonia (nitrification) takes place and in the HSSF, due to the anaerobic conditions that prevail, and nitrates produced from the former would be reduced to molecular nitrogen (denitrification) (Tuncsiper, 2009). However, multi-stage hybrid CWs have not gained much interest in wastewater treatment. Therefore, currently, single stage hybrids, particularly a combination of VSSF and HSSF CWs are the most frequently used systems in wastewater treatment (Vymazal, 2008; Vymazal, 2013; Zhang et al., 2015). Although hybrid systems are currently used in many European countries including France, The United Kingdom, Norway, Slovenia, Austria, and Ireland (Vymazal, 2013), there is not a single report on the full-scale application of hybrid systems in tropical and subtropical countries probably due to deficiency in information regarding the performance of these systems operating in these climatic regions.

1.2.2 Performance of Subsurface flow constructed wetlands

This section presents the performance of SSF CWs treating domestic wastewater reported in literature.

1.2.2.1 BOD, COD, TSS and TP removal

The quality of treated water in terms of BOD, COD, TSS and TP from different types of SSF CWs reported in literature is summarised in Figure 1-6 below. Results from Figure 1-6 agree with Vymazal (2005) who stated that SSF systems are very efficient where compliance with BOD, COD and TSS removal standards are required.

Although the surveyed literature, clearly shows that there is limited information on the performance of these systems, the mean effluent quality for BOD, COD and TSS from all the SSF CWs (Figure 1-6), is generally lower than the effluent discharge limits for most of the countries in sub-Saharan Africa (Appendix 1) i.e. HSSF: TSS=27.2 ± 8.1 mg/L, BOD_5= 19.6 ± 5.4 mg/L and COD= 64.7 ± 16.4 mg/L, VSSF: TSS= 15.4 ± 5.6 mg/L, BOD_5= 22.7 ± 8.7 mg/L and COD= 76.5 ± 19.9 mg/L and V-H SSF: TSS=8.0 ± 1.8 mg/L, BOD_5= 19 ± 7.5 mg/L and COD=74.0 ± 5.7 mg/L.

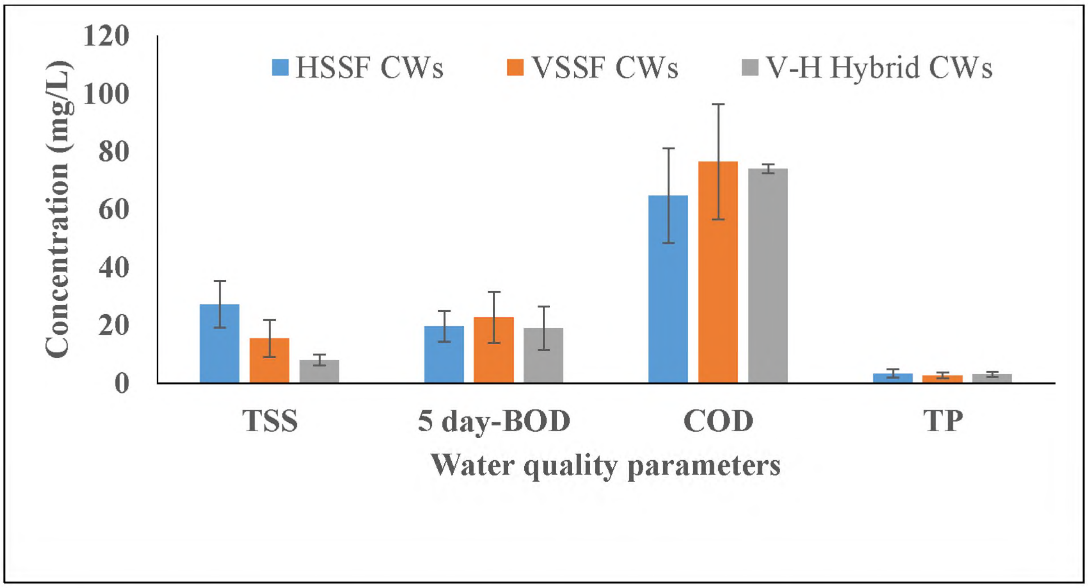

Figure 1-6. Effluent quality for TSS, BOD5, COD and TP from SSF CWs reported in literature. Data for VSSF and HSSF was summarized from Zhang et al. (2015) while that of V-H SSF hybrid CWs was from Vymazal (2013) and Zurita, & White (2014). For HSSF CWs, number of data (n)= 4, 5 and 6 for TSS, BOD_5 and COD respectively; for VSSF CWs, n= 2, 3 and 5 for TSS, BOD_5 and COD respectively while for V-H SSF hybrid CWs, n=8, 12 and 11 for BOD_5, COD and TSS respectively.

The high effluent quality from the surveyed SSF CWs is attributed to various removal pathways involved in organic matter and TSS removal. Besides aerobic and anaerobic degradation, filtration and sedimentation are responsible for removal of organic matter and TSS (Kadlec & Knight, 1996). Furthermore, results show that mean effluent TP from all the different SSF CWs investigated in this study are almost similar and these are 3.4 ± 1.4 mg/L, 2.7 ± 0.9 mg/L and 3.0 ± 0.8 mg/L for HSSF, VSSF and V-H SSF CWs respectively. These results are in support of Brix et al. (2000) who stated that phosphorus removal in VSSF CW is very comparable to that of HSSF CW. This is because the mechanisms of removal of TP are mainly physical and include adsorption to the substratum and plant root surface/ or precipitation with ions such as calcium, aluminum, and iron present in the rooting medium, and neither is influenced by oxygen concentration (Arias et al., 2001). Despite the low TP reported in the present study (Figure 1-6), other studies on SSF CWs for municipal wastewater treatment such as Lin et al. (2005) reported higher values for TP in the effluent than in the feed water. This is attributed to release of adsorbed phosphorus from the treatment media due to the anaerobic conditions that dominate SSF CW (Cramlet & Turyk, 2002). However, the increase in TP in the effluent is not so great to exceed the General Authorization standards for discharge into the environment. Additionally, as previously mentioned this chapter, municipal wastewater contains low TP with a typical value of 6 mg/L. Therefore, since most countries in sub-Saharan Africa have TP effluent discharge limits higher than raw sewage (Appendix I), this implies that despite the increase in the effluent, removal of TP from SSF CWs is not a challenge.

1.2.2.2 Nitrogen species

Studies claim that there is a problem for systems operating independently either as HSSF or VSSF to achieve effluent standards for nitrogen (Molle et al., 2008; Vymazal, 2013) probably due to the existing theories underlying the effect of mode of feeding the CWs on nitrogen removal. Firstly, Stein et al. (2003) claims that continuous feeding of HSSF CWs and keeping water below the bed restricts opportunities for contact between air and water resulting into limited transfer of oxygen in the system. Consequently, HSSF CWs are characterized as being largely anaerobic/anoxic systems, which limits nitrification (Langergraber, 2008). Additionally, studies such as that by Brix & Arias (2005) hypothesize that batch/intermittent feeding of VSSF results into good oxygen transfer and hence nitrification. Such a claim resulted into a conclusion that VSSF CWs show high removal for NH_4^+-N but poor removal for NO_3^--N (Molle et al., 2008).

To investigate the above theories, the effluent quality of inorganic nitrogen species from SSF CWs (treating domestic wastewater) reported in literature was evaluated and the results are summarized in Figure 1-7 below. Results show that there are very few data about NO_3^--N effluent quality from SSF CWs. However, by comparing independent systems, analysis of the available data shows that the mean NO_3^--N effluent quality from VSSF CWs (0.4 ± 0.1 mg/L), is surprisingly much lower than the mean NO_3^--N effluent quality from HSSF CWs (0.7 ± 0.0 mg/L). However, due to the limited data, it is not possible to conduct a statistical test to ascertain if the mean NO_3^--N effluent quality from the CWs is significantly different.

Even though nitrate is produced in VSSF CWs as claimed by Molle et al. (2008), amounts are below the effluent discharge limit for NO_3^--N for all the presented environmental bodies in in sub-Saharan Africa (Table 1-1). However, since there is insufficient data on NO_3^--N effluent quality from VSSF CWs reported in literature, it is again difficult to conclude that VSSF CWs are poor at nitrification.

Again, Figure 1-7 shows that the mean NH_4^+-N effluent quality from HSSF CWs (9.9 ± 3.8 mg/L) is better than from VSSF CWs (17.2 ± 5.4 mg/L). The mean NH_4^+-N effluent quality from either HSSF or VSSF CWs does not meet the effluent discharge limits for all the regulating environmental bodies in sub-Saharan Africa (Appendix 1). Although a significant difference between the mean effluent NH_4^+-N values from HSSF and VSSF CWs was not found (p=0.33), the results obtained imply that, even though HSSF CWs are characterized by anaerobic conditions (Langergraber, 2008), they perform better than VSSF CWs at removing NH_4^+-N.

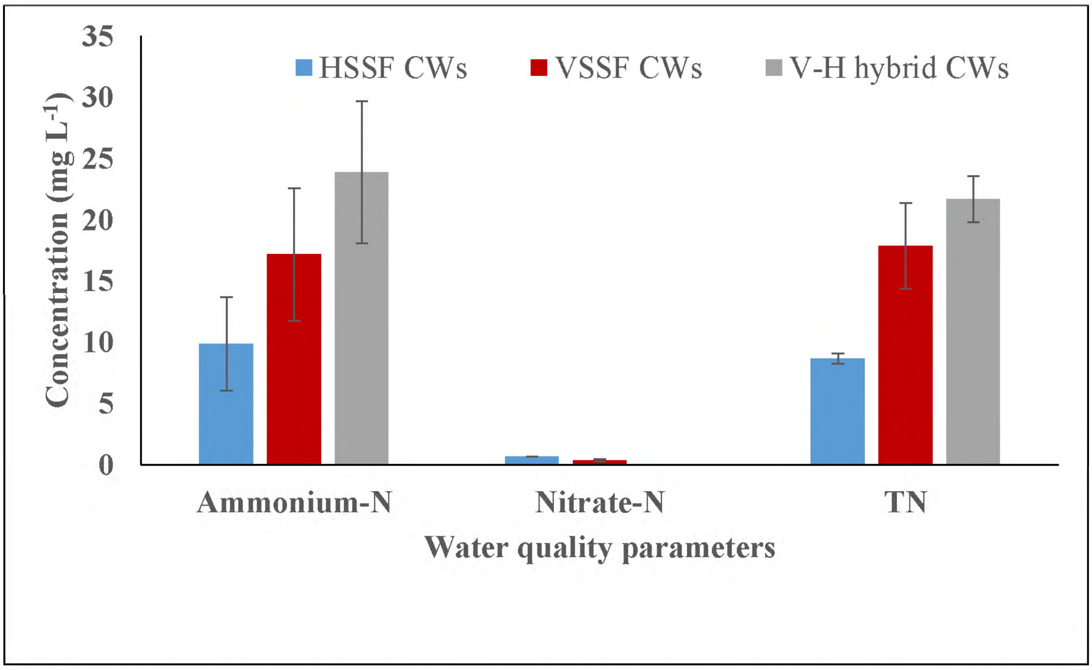

Figure 1-7. Effluent quality for nitrogen species from the different types of constructed wetlands. Data for VSSF and HSSF CW was summarized from Zhang et al. (2015) while for V-H hybrid CWs was from Vymazal (2013). Number of data (n) for HSSF CWs=5, 1 and 2 for NH_4^+-N, NO_3^--N and TN respectively; VSSF, n=3, 2 and 6 for NH_4^+-N, NO_3^--N and TN respectively and V-H SSF hybrid CWs, n= 11 and 7 NH_4^+-N and TN respectively.

Furthermore, while V-H hybrid CWs are reported for their potential in achieving higher treatment efficiency than a single CW particularly for nitrogen (Vymazal, 2013), a literature survey was conducted to investigate if this theory holds for SSF CWs treating domestic/municipal sewage. Firstly, the survey revealed that there is a paucity of data regarding the performance of V-H hybrid systems on the African continent. Out of the sixteen V-H hybrid CWs reported in literature (Table 1-6), only three were reported from Africa and these were from one country (Tunisia) which is characterized by Mediterranean climate. Again unexpectedly, on comparing the effluent quality from Figure 1-7, HSSF CWs revealed a better nitrogen effluent quality than V-H hybrid CWs. The mean effluent quality from V-H hybrid CWs were 17.3 ± 4.7 mg/L and 21.7 ± 1.9 mg/L for NH_4^+-N and TN respectively. Although a t-test showed no significant difference between NH_4^+-N effluent quality from HSSF and V-H

SSF CWs ($p=0.25$), TN effluent quality from the two systems was significantly different ($p=0.00$).

In summary, this section has shown that NH_4^+-N is the most monitored and reported inorganic nitrogen species from SSF CW treating domestic wastewater probably because it is the most abundant inorganic nitrogen species in wastewater (Vymazal, 2007) and the most preferred nitrogen form for most algae species for uptake and assimilation since it is more reduced energetically than either NO_3^- or NO_2^- (Kadlec & Knight, 1996; Rücket & Giani, 2004). Thus, it is the most important nitrogen species responsible for eutrophication. Additionally, although HSSF CWs reported in literature show a better NH_4^+-N effluent quality than either VSSF or V-H hybrid CWs, all the CWs produced an effluent quality that does not comply with the effluent discharge limits for all environmental regulating bodies in sub-Saharan Africa. The following section surveys the factors that could be responsible for the poor nitrogen effluent quality from SSF CWs.

Table 1-6. Effluent quality and performance of single stage V-H hybrid constructed wetlands for sewage treatment from 2000-2014. Data was modified from Vymazal (2013) and Zurita & White (2014).

Example by country	Water Quality Parameter (mg/L)						Macrophyte	Area (m^2)	Reference
	BOD_5	COD	TSS	TP	TN	NH_4^+-N			
Belgium[a]							*P. australis*	2250	Lesage et al. (2007), VMM (2006)
Effluent quality	5.8	43	5	2.9	27	*i*			
% Removal	92	81	9	45	43	*i*			
Belgium[a]							*P. australis*	2250	Lesage et al. (2006), VMM (2006)
Effluent quality	4	47	4.8	3.4	*i*	*i*			
% Removal	92	81	95	32	*i*	*i*			
Belgium[a]							*P. australis*	1080	Lesage et al. (2006), VMM (2006)
Effluent quality	9	49	4.3	3.4	26	*i*			
% Removal	96	90	98	47	53	*i*			
Belgium[a]							*P. australis*	660	Lesage et al. (2006), VMM (2006)
Effluent quality	10.3	57	15	4.3	23	*i*			
% Removal	93	84	87	38	60	*i*			
Estonia[a]							*P. australis*	432	Öövel et al. (2007)
Effluent quality	5.5	*i*	5.8	0.4	19	9.1			
% Removal	94	*i*	87	91	70	84			
Tunisia[b]							*Typha sp., P. australis*	1.8	Abidi et al. (2009)
Effluent quality	30	134	18	7.2	*i*				
% Removal	93	90	98	77	*i*				
Tunisia[b]							*Typha sp., P. australis*	1.8	Keffala & Ghrabi (2005)
Effluent quality	*i*	*i*	*i*	*i*	*i*	30			
% Removal	*i*	*i*	*i*	*i*	*i*	19			

Tunisia[b]							*P. australis. Typha sp*	327	Kouki et al. (2009)
Effluent quality	*i*	*i*	*i*	*i*	*i*	*i*			
% Removal	93	89	98	72	*i*	*i*			
Spain[b]							*P. australis, Scirpus sp*	0.88	Herrera Melián et al. (2010)
Effluent quality	24	71	3.6	*i*	*i*	11			
% Removal	85	74	95	*i*	*i*	91			
Spain[a]							*Typha latifolia*	450	Vera et al. (2010)
Effluent quality	66	172	16.2	8.8	26	40			
% Removal	84	77	95	35	43	51			
China[c]							*P. australis*	3716	Zhai et al. (2011)
Effluent quality	*i*	21	3.2	0.4	*i*	2.2			
% Removal	*i*	84	97	85	*i*	80			
China[c]							*Cyperus alternifolius*	1400	Zhai et al. (2011)
Effluent quality	*i*	26	7.2	0.9	*i*	5.3			
% Removal	*i*	90	85	77	*i*	84			
China[c]							*Cyperus alternifolius*	4459	Zhai et al. (2011)
Effluent quality	*i*	28	1.6	0.6	14	6.2			
% Removal	*i*	84	99	68	65	72			
Italy[c]							*P. australis*	6.75	Foladori et al. (2012)
Effluent quality	*i*	36	*i*	0.2	17	11.4			
% Removal	*i*	94	*i*	98	78	80			
Brazil[a]							*Typha sp., Zizaniopsisbo nariensis*	110	Phillipi et al. (2010)
Effluent quality	*i*	29	*i*	4	*i*	5.6			
% Removal	*i*	95	*i*	69	*i*	89			

Mexico[a]							*Zantedeschia aethiopica*	3.66	Zurita & white (2014)
Effluent quality	*i*	*i*	*i*	12	102	19			
% Removal	*i*	*i*	*i*	0	26	85			

Additional information:

-Letters a, b and c represent the Design/operational parameter considered during the study. a=Not provided, b=COD or BOD was the major operational parameter and c=design based on population equivalent.

1.2.3 Factors influencing the performance of subsurface flow constructed wetlands

There is an array of literature describing the factors influencing the performance of SSF CWs in general including design and construction, mode of operation, hydraulic and organic loading, media type and size, macrophyte, hydrology and influence of environmental variables (Kantawanichkul & Somprasert, 2004; Korkusuz, 2004; UN-Habitant, 2008). This section however, presents a detailed discussion of design, mode of operation and macrophyte as factors that may influence nitrogen removal from SSF CWs.

1.2.3.1 Design and size determination

Proper design is a major consideration for successful performance of a CW (USEPA, 2000). Figure 1-8 shows the configuration of well-designed CW for municipal/domestic wastewater treatment proposed in literature (Steiner & Freeman, 1989; UN-Habitat, 2008). The major component thus, include: preliminary and primary treatment units and the CW itself. A CW comprises of substrate/treatment media, vegetation, and micro-organisms that interact to remove pollutants from wastewater (Qasaimeh et al., 2015) while the preliminary treatment unit includes the screen and grit chambers that are necessary to remove heavy solids like plastic bags and bottles, rags, sand, gravel etc.

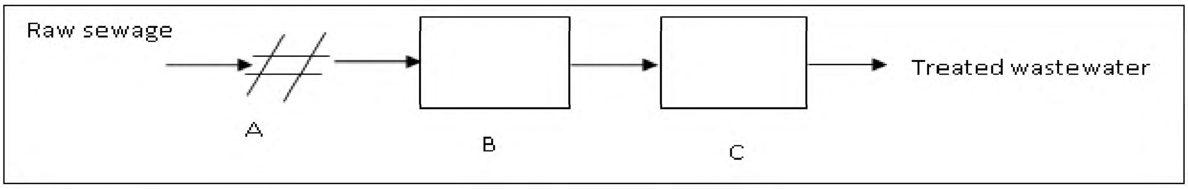

Figure 1-8. Process flow for a constructed wetland system designed for domestic wastewater treatment. Key: A=preliminary treatment unit: screen and grit chamber to remove heavy solids, B=primary treatment unit: septic tank, anaerobic baffled reactor (for individual households) or primary facultative pond and C=constructed wetland for secondary treatment (UN-Habitat, 2008).

In contrast, the primary treatment unit is responsible for reducing TSS and organic load through physical operation mainly sedimentation (UN-Habitant, 2008), and may require installation of an Imhoff tank or septic tank for individual households (Korkusuz, 2004) or a primary sedimentation tank or stabilization pond(s) for small communities (Kayombo et al., 1998; Abis & Mara, 2005). As mentioned in the introduction of this chapter, the primary treatment unit particularly involving the use of WSPs is associated with a problem of sludge accumulation and may require desludging from time to time. However, in the CW system, the primary stabilization pond can be upgraded into an advanced facultative pond (AFP). The AFP is constructed by integrating a fermentation pit into a primary facultative pond aimed at speeding up anaerobic digestion, this combination has an added advantage of methane production for use in energy derivation (Rose et al., 2002). A well-planned and constructed primary treatment facility for a CW system must remove up to 60 % of the influent BOD at 20 ^oC (Machibya & Mwanuzi, 2006) and it is likely, that addition of a fermentation pit will generate methane equivalent to 30 % of the influent organic carbon (Green et al., 1995). Most importantly, following the current process flow proposed for a CW (UN-Habitat, 2008), utilizing the AFP in the CW system can "*completely*" solve the problem of sludge production thereby reducing the maintenance operations while producing a reliable effluent quality.

Removal of pollutants from CWs is dependent not only on efficient removal of organic matter and suspended solids from the primary treatment unit, but also on optimum design. The latter, is to attain better nutrient removal while mitigating operational problems. Hence, the optimum performance of a CW depends on using an appropriate area for a given organic load since smaller areas for large flows result in lower treatment efficiency. As a consequence of this requirement, different methods have been proposed for sizing the effective area of a SSF CW including, the population equivalent (PE) method, pollutant loading method, and non-mechanistic models (Tousignant et al., 1999; Un-Habitat, 2008).

Among the methods used for sizing CWs, non-mechanistic models have been widely used for estimating the surface area required for efficient performance of a SSF CW (Vymazal, 2005) and therefore will be discussed in more details. Different models for sizing a SSF CW have been proposed by many authors. Literature shows that, earlier designers employed the model below that was first proposed by Kickuth (1977).

$$A_s=Q_{in}(\ln c_{in}-\ln c_o)/(K_{BOD}) \qquad \text{(i)}$$

Where: A_s= Surface area of the bed (m^2), Q_{in}= Average flow rate into the wetland (m^3/d), C_{in}= influent BOD (mg/L), C_o=expected effluent BOD (mg/L) and K_{BOD}=the rate constant (m/d).

However, there is controversy over the rate constant (K_{BOD}) used in the area estimation. Earlier studies reported a K_{BOD} value of 0.19 m/d (Kickuth, 1977). However, Vymazal (2005) opposes this figure stating that it is too large thus, results in a very small area of wetland and so lowers the performance of the system. Based on Experience, Vymazal (2005) suggests that the K_{BOD} value is much lower than that estimated by Kickuth (1977). Later, studies such as Schierup et al. (1990) and Cooper (1990) reported K_{BOD} values of 0.083 m/d and 0.067-0.1 m/d respectively which agrees with Vymazal (2005) although the latter stresses that a K_{BOD} value of 0.1 m/d (36.5 m/yr) is adequate; hence, the surface area for any wetland is about 5 m^2/PE. Nevertheless, the K_{BOD} value reported in literature is generated from studies conducted from temperate regions where temperature variations are high. Since K_{BOD} value is temperature dependent, it is not known if this value still holds for tropical and subtropical regions with minimal temperature range. Development in research led to Reeds et al. (1995) model shown below. Indeed, this model is an advancement of the Kickuth (1977) model. The former assumed a first order decay, plug flow model for all pollutants, including BOD, TSS, TP, TN, Org-N, NH_4^+-N, NO_x-N, and faecal coliforms.

$$A_s=Q_{in}(\ln C_{in}-\ln C_o)/K_T y n \quad \text{(ii)}$$

Where: A_s= treatment area of the CW (m^2), Q_{in}=mean flow rate into the CW (m^3/d), C_{in}= mean influent BOD_5 concentration (mg/L), C_o= expected effluent BOD_5 concentration from the CW (mg/L), K_T= rate constant corresponding to water temperature in the constructed wetland in (d^{-1}); but K_T=$K_R\theta_R^{(Tw-20)}$, where: K_{20} is the rate constant at 20 °C reference temperature (d^{-1}), Tw is the wetland temperature (°C), θ is the temperature coefficient for the rate constant. The first order kinetic constant values at 20 °C (K_{20}) and temperature coefficient (θ) are pollutant removal dependent. y= depth of the system (m) and n= porosity expressed as a decimal fraction. The first order kinetic constant values at 20 °C (K_R) and temperature coefficient, θ_R, are pollutant dependent (Reeds et al., 1995). Table 1-7 summarizes the values of K_R and θ_R at 20 °C for subsurface flow CWs.

Table 1-7. Temperature coefficients for rate constant for design equations proposed by Reed et al. (1995).

Constants	BOD removal	NH_4^+-N removal (nitrification)^y	NO_3^--N removal (denitrification)^y	Pathogen removal
K_R	1.104	K_{NH}^z	1.00	2.6
θ_R	1.06	1.048	1.15	1.19

Additional notes:
The rate coefficients are applicable for temperatures higher than 1 °C.
^{y}Nitrification/ denitrification cannot proceed at temperatures below 0 °C.
$^zK_{NH}=0.01854+0.3922(rz)^{2.6077}$ (iii)
Where K_{NH}= SSF nitrification rate (d^{-1}) and rz= depth of the bed occupied by the root zone (%, expressed as decimal).

Most CW designers have ignored the use of nitrogen as a major parameter in the design of SSF CWs probably due to lack of information about nitrogen in the earlier model proposed by Kickuth (1977). However, the advantage of using Reeds et al. (1995) model is that it gives a designer a chance to decide on the parameter to use depending on the treatment objective of the CW. Therefore, since the major objective of V-H SSF CWs is nitrogen removal (Vymazal, 2013), it is suggested that design of such a system should be based using nitrogen as a design parameter and BOD as has been the case in previous studies e.g. Keffala & Ghrabi (2005), Abidi et al. (2009), Herrera Melián et al. (2010) etc. Thus, using BOD as a design or an operational parameter in studies reported in literature could have resulted into a smaller area required for nitrogen removal hence the poor effluent quality observed with V-H SSF CWs in section 1.2.2.2. As mentioned earlier (Section 1.1), Vymazal (2005) stated that using BOD results in under estimation of the area; hence using BOD as a design parameter for CWs is appropriate for organic matter and TSS removal but not suitable for nutrient removal.

However, the limitation of using the Reeds et al. (1995) model is that since the depth of the bed occupied by the root zone (rz) is not defined, the CW designer always works with an assumption which may result in over or underestimating the depth thus, resulting into small or large area for the CW.

1.2.3.2 Mode of feeding

The different mode of operating CWs, the theories underling these modes of feeding proposed by various authors, advantages as well as their limitations were elaborated (Section 1.2.1). Literature shows that the mode of operation of a CW greatly influences the redox potential and consequently the performance of the CW (Faulwetter et al., 2009). Thus, intermittent feeding is claimed to show a high redox potential due to high oxygen transfer into the system and therefore high nitrification potential (Molle et al., 2008). However, according to the surveyed data (Section 1.2.2), it was revealed that continuously fed HSSF CWs produced a better effluent quality regarding the various nitrogen species than V-H SSF hybrid CWs in which the VSSF CW is are routinely fed intermittently and purported to be highly aerobic. Furthermore, literature shows that a continuously fed V-H SSF hybrid CWs produced a better effluent quality regarding NH_4^+-N than intermittently fed hybrid systems. For instance, continuous operation of a V-H hybrid CW planted with *P. australis* (in the VSSF CW) and *Scirpus sp* (in the HSSF CW) at HLR of 0.4 m/d with influent NH_4^+-N concentration of 122 mg/L revealed an effluent NH_4^+-N concentration of 11 mg/L (91 % removal). On contrary, intermittent operation of a hybrid CW planted with *Zantedeschia aethiopica* at HLR of 0.28 m/d with influent NH_4^+-N concentration of 128.2 mg/L revealed an effluent NH_4^+-N concentration of 19 mg/L (85 % removal). Furthermore, Keffala & Ghrabi (2005) reported effluent NH_4^+-N concentration of 30 mg/L (19 % removal) from an intermittently operated V-H SSF hybrid system planted with

Typha sp (in the VSSF CW) and *P. australis* (in the HSSF CW). The HLR and influent NH_4^+-N concentration were 0.24 m/d and 37 mg/L respectively.

The better NH_4^+-N effluent quality reported by Melian et al. (2010) than either Keffala & Ghrabi (2005) or Zurita & White (2015) could be attributed partly to the difference in the mode of feeding of these systems. Various mechanisms are reported to be responsible for nitrogen removal from CWs including nitrification, denitrification, microbial uptake and volatilization to mention a few (Kadlec & Knight, 1996; Vymazal, 2007; Faulwetter et al., 2009).

Literature reports that nitrification and denitrification are the major nitrogen removal mechanisms (Stefanakis et al., 2014). However, according to the available information (Section, 1.2.2.2), denitrification seems to be limited in the V-H SSF hybrid CWs due to the adequate NO_3^- produced in the VSSF CWs hence, resulting into the poor NH_4^+-N effluent quality observed in intermittently fed V-H SSF hybrid CWs. Therefore, since nitrogen removal from SSF CWs via plant uptake is considered negligible (Vymazal, 2007), it is can be proposed that microbial uptake is the major nitrogen removal pathway in V-H SSF hybrid CWs. CWs are reported to harbour a variety of microorganisms including bacteria, archea, protozoa, fungi and algae which are localized in the biofilm (Nadell et al., 2009) found on attached surfaces e.g. gravel and plant roots. Hence, the potential of CWs to remove pollutants from wastewater depends on the time of interaction between wastewater and the treatment media (Kantawanichkul et al. 2009). Even though the HRT required for effective nitrogen removal from SSF CWs is not documented, for other parameters e.g., typical HRT ranging from 2 to 5 d for organic matter removal have been reported for HSSF CWs (Wang et al., 2010) compared to 0.5 d for VSSF CWs (Abdelhakeem et al, 2016). Thus, a longer HRT due to continuous feeding is likely than in intermittently fed V-H hybrid CWs and therefore a better NH_4^+-N effluent quality reported by Melian et al. (2010) than either Zurita & White (2014) or Keffa & Ghrabi (2005). Furthermore, section 1.2.2.2 showed that continuously fed HSSF produce a better NH_4^+-N effluent quality than intermittently fed VSSF or V-H SSF hybrid CWs which could still be linked to the longer HRT achieved with continuous than intermittent feeding which allows for the interaction of the biofilm with wastewater for improved nitrogen removal. Therefore, it can be suggested that continuous feeding should be preferred to intermittent feeding if efficient removal of nitrogen from CW is to be achieved.

1.2.3.3 Macrophyte species

Various wetland macrophytes are available for use in CWs (Table 1-6) and a review by Brisson & Chazarenc (2009) on the effect of macrophyte species selection on pollutant removal in SSF CWs revealed that macrophyte species selection does not influence effluent quality. However, based on results from studies by Keffala & Ghrabi (2005), Herrera Melián et al. (2010) and Zurita & White (2014), it can be argued that if nitrogen is the target pollutant for removal especially in hybrid CWs, it could be important to pay more attention to the macrophyte species used than the mode of feeding. Hence, the difference in the NH_4^+-N effluent quality reported by Keffala & Ghrabi (2005), Herrera Melián et al. (2010) and Zurita & White (2014) based on the macrophyte species used can be explained in two ways:

(i) Wetland plants have been reported to transfer oxygen to the root system for oxidation of ammonium (Brix, 1994). Therefore, different plants might show differences in oxygen release into the rhizosphere. To date, there is little information regarding oxygen release rates for the various plants used in CWs, yet use of this trait in selecting species for use in CWs for a target pollutant especially nitrogen may be very important. Among the macrophytes used, *P. australis* is the most studied. Using different methods, several studies have reported oxygen release for *Phragmites* spp. Armstrong et al. (1990), Brix (1990) and Gries et al. (1990) for instance reported oxygen release rates of up to 5-12 g/m^2/d, 0.02 g/m^2/d and 1-2 g/m^2/d respectively. More recently, axial O_2 profiles revealed that O_2 was released at a rate of 0.21 μmol O_2/cm^2 (root surface area)/h in the apical region of *Phragmites* roots (Okabe et al., 2012). In this study, O_2 release was associated with regions of the root rich in *Nitrosomonas*-like ammonium oxidizing bacteria and *Nitrospira*-like nitrogen oxidizing bacteria, implicating *Phragmites* root biofilms in nitrification. Unfortunately, there appears to be no published information regarding oxygen release rates in roots of *Z. aethiopica* used by Zurita & White 2014). An earlier study on the use of this ornamental species in HSSF CWs by Belmont & Metcalfe (2003) however, showed considerable reduction in the influent ammonium concentration indicating that *Z. aethiopica* has a positive effect on ammonium removal. While SSF CWs are known to be anaerobic/anoxic, Belmont & Metcalfe (2003) observed increase in effluent oxygen concentration, which was assumed to be linked to high oxygen release rates of roots of *Z. aethiopica*. Although still to be established, *Z. aethiopica* could have higher oxygen release rate than *P. australis*, which contributed to the significant difference in ammonium removal reported by Keffala & Ghrabi (2005) and Zurita & White (2014).

(ii) In another study, Bezbaruah & Zhang (2004) reported lower oxygen release rates (in the range 0.00021-0.00155 $g/m^2/d$ and 0.00083-0.00288 $g/m^2/d$ for brown and white roots of *Scirpus* spp respectively that was used by Herrera Melián et al. (2010) than have been reported for *Phragmites* spp. The lower NH_4^+-N concentration reported by Herrera Melián et al. (2010) for a continuously operated V-H SSF hybrid CWs compared to the intermittently operated V-H SSF hybrid CWs reported by Keffala & Ghrabi (2005) and Zurita & White (2014) could be related to differences in secondary metabolite and exudate production by roots of the macrophyte and induction of extracellular polymeric substances (EPSs). Indeed, *P. australis* has been reported to contain high concentrations of the alkaloid N,Ndimethyltryptamine in the rhizomes (Wasel et al., 1985; Khan et al., 2012). During the treatment process, these alkaloids, or similar secondary metabolites, may impact EPS and biofilm formation to effect changes in nutrient abstraction. For example, proteins secreted by *Bacillus amyloliquefaciens* FZB42, a root-associated plant growth promoting rhizobacterium, are involved in nutrient utilization and transport, the induction of systemic resistance against plant pathogens, and as a trigger of plant growth (Kierul et al., 2015). In this study, it can therefore be hypothesized that the alkaloid concentration in the three different macrophytes follows the order *P. australis* > *Zantedeschia aethiopica* > *Scirpus* spp.

Clearly, further study is needed to investigate in more detail the biochemistry and physiology of different CW macrophyte species to determine the contribution of each to nutrient abstraction/utilization. In this way, CWs together with the appropriate macrophytes can be designed and implemented to more easily meet specific treatment objectives.

1.2.4 Suitability of macrophyte for application in constructed wetlands

Aquatic macrophytes form one of the major components of CWs (Lee & Scholz, 2007). While previous reports claimed that, macrophytes used in CWs are not vital and that the gravel bed itself rather than the macrophytes is responsible for pollutant reduction (Mara, 2003), recent research has confirmed that nutrient removal from wastewater is better in cells with plants than in adjacent cells without plants (Senzia et al., 2003; Mburu et al., 2013). Despite the fact that the role of macrophyte in nutrient uptake is considered negligible particularly in SSF CWs, their influence in relation to wastewater treatment processes is linked more to the physical effects provided by plant roots (Vymazal, 2007). Some of the advantages of macrophyte in CWs according to Gersberg et al. (1986) and Brix (1994) are outlined below.

- They increase surface area for attachment of microbial populations, which mediate more of the nutrient transformation processes in CWs than nutrient uptake by plants, which seems to be of a quantitative importance particularly in low loaded systems.
- They mediate transfer of oxygen to rhizosphere thus creating aerobic conditions around the root zone, which is conducive for the colonization of nitrifying bacteria that convert ammonia to nitrate.
- They provide a conducive environment for physical filtration of suspended solids, particulate nutrients and organic matter.
- Insulate the CW beds against ice during winter.
- Macrophytes have extra site-specific benefits by providing habitat for wildlife and making wastewater treatment aesthetically pleasant.

As mentioned earlier, there are various macrophytes available for use in CWs (Section 1.2.1.2). However, among the wetlands macrophytes, many authors recommend *P. australis* as most suitable macrophytes for use in CWs. The wide distribution, rapid growth, high potential productivity; deep rhizome and root system and climatic tolerance and are some of the reasons for its worldwide application in CWs (Lüderitz et al., 2001; UN-Habitat, 2008).

Among the common macrophytes used in CWs, *P. australis* is the most widely distributed species, found on all continents except the Antarctica (Jose et al., 2013). The worldwide distribution of this species is attributed to its ability to inhabit different environments including soils with varying pH, salinity, fertility and texture (Srivastava et al., 2014). The ability of *P. australis* to inhabit various environments is due to its association with arbuscular mycorrhizal fungi (AMF) (Wang et al., 2015). These fungi are able to establish a symbiotic relationship with the host plant; the AMF provides the host plant with water and nutrients in return for organic carbon from the plant. Thus, according to Miransari (2014), AMFs have been used to alleviate different stresses on plant growth including salinity, drought, acidity, flooding and heavy metals contamination. Therefore, due to this unique characteristic, *P. australis* could be a suitable macrophyte for different bioremediation applications.

Furthermore, the asexual reproduction behaviour of the plant results into high productivity potential hence longer survival of the rhizomes within the reed bed even during adverse conditions of drought and winter (Capon & Reid, 2016). For this reason, *P. australis* is widely selected for use in CWs since planting of the species is possible all year round (Brix & Arias 2005). Besides *P. austalis*, different types of macrophytes are available for use in CWs but they differ in their ability to uptake nitrogen (Zhang et al., 2007). Table 1-8 shows nutrient uptake capabilities of the different macrophytes commonly used in CWs. According to Brix (1994), among the emergent macrophytes commonly used in SSF CWs, *P. australis* has the highest nitrogen uptake capability of 2500 kg/ha/yr, which is two times more than either *Cyperus papyrus* or *Typha latifolia*.

Table 1-8. Nutrient Uptake capabilities of a number of emergent, free-floating, and submerged macrophytes. Data obtained from Brix (1994).

Macrophytes	**Uptake capabilities (kg/ha/yr)**	
	Nitrogen	**Phosphorus**
Phragmites australis	2500	120
Eichhonia crassipes	2400	350
Cyperus papyrus	1100	50
Typha latifloia	1000	180
Pistia stratiodes	900	40
Potanogeton pectinatus	500	40
Ceratophylum demersum	100	10

The high nitrogen uptake of the plant could be associated with its deep root system. In a study of significance of rooting depth in mine plants, Kohzu et al. (2003) reported a root depth of *P. australis* ranging from 0.05-2 m. These authors observed that the deep root system had implications for nitrogen uptake. There was a tendency for plants having a deeper root system to exhibit higher stable isotope ratio of nitrogen (δ^{15} N) value. Deep-rooted *P. australis* depleted more inorganic nitrogen than other plants under investigation suggesting that deep-rooted plants absorb more nitrogen from deep peat layers. While the deep root system makes *P. australis* more competitive than other plants in nutrient uptake, this may have implications for high water uptake. Literature reports that the evapotranspiration (ET) rates for common reed flactuate between 4.7 to 12.4 mm/d depending on climatic conditions yet, in low fed systems such extremely high evapotranspiration rates may exceed the influent flow resulting into a zero discharge (Holcová et al., 2009). Although there is insufficient literature on the ET rates of the different wetland macrophyte species, *P. australis* ET rates are lower than for some macrophyte species. Kyambadde et al. (2005) for instance reported ET rate of 24.5 mm/d from *Cyperus* *papyrus* CW system in Uganda. *P. australis* may therefore be a better macrophyte for use in CWs than *C. papyrus* especially in instances where wastewater reuse is critical.

Lastly, *P. australis* has a strong influence on the microbial populations involved in pollutant removal by CWs. A recent study for instance reported many more species of bacteria involved in total nitrogen cycle in *P. australis* than in *Typha angustifolia* L. roots (Li et al., 2013). Nevertheless, it is likely that that microbial mediated processes that remove nitrogen from SSF CWs e.g. denitrification and anaerobic ammonia oxidation (ANAMMOX) proceed at slower rate than those leading to nitrogen accumulation e.g. nitrogen fixation, mineralization/ammonification and nitrate ammonification and hence resulting in poor nitrogen removal from SSF CWs observed in Section 1.2.2.2. Thus, future studies should aim at optimizing nitrogen removal processes such increasing plant density in the system that add oxygen for the nitrification process and consequently increasing NO_3^--N for the denitrification process.

1.2.5 Nitrogen removal process

After organic carbon, nitrogen removal is the second most important pollutant for remediation from CWs treating domestic wastewater (Vymazal, 2007). The major processes accounting for nitrogen removal in CW are physico-chemical and biological processes. While physico-chemical processes such as filtration and sedimentation may depend on the maturity of the bed, biological processes such as nitrification and denitrification depend on temperature, oxygen availability and pH (Kadlec & Knight, 1996; US.EPA, 2000). Wetland pH is correlated with calcium content of water (pH of 7 = 20 mg/L Ca); thus, CW waters usually have pH of around 6-8 (Reeds et al., 1995). According to Tong & Sikora (1995), oxygen may enter CWs via the influent water, plants release and atmospheric diffusion.

There are various processes responsible for nitrogen removal and transformation in CWs described in literature (Kadlec & Knight, 1996; Vymazal, 2007; Faulwetter et al., 2009). Table 1-9 presents a summary of the nitrogen transformation and removal pathways and their magnitude on nitrogen removal in SSF CWs. Nevertheless, not all these processes essentially remove nitrogen from wastewater. Processes that eventually reduce nitrogen from CWs are limited only to ammonia volatilization, denitrification, plant uptake (with biomass harvesting), ammonia adsorption, ANAMMOX and organic nitrogen burial; while other processes such as mineralization, nitrate ammonification and nitrification “only” convert nitrogen into various nitrogen species but do not in fact remove nitrogen from the wastewater (Vymazal, 2007). Among the nitrogen transformation and removal mechanisms, studies report that nitrification and denitrification are the major important nitrogen transformation and removal pathways from SSF CWs (Vymazal, 2007; Stefanakis et al., 2014). Therefore, these processes are discussed in more details.

Table 1-9. Potential magnification of nitrogen transformation processes/mechanisms in subsurface flow constructed wetlands. Table obtained from Vyamazal (2007).

Process	HSSF	VSSF
Volatization	**0**	**0**
Ammonification	++++	++++
Nitrification	+	+++++
Nitrate ammonification	??	??
Denitrification	**+++++**	**+**
Nitrogen fixation	??	??
Microbial uptake	++	++
Plant uptake[y]	**++**	**++**
Ammonia adsorption	+	+
Organic nitrogen burial	++	+
ANAMMOX	??	??

Key: +++++=very high, ++++=high, +++=medium, ++=low, +=very low, 0=Zero, ??=not known.

Additional notes:

i) Processes that ultimately remove total nitrogen from wastewater are indicated in bold,

ii) [y]=with multiple harvesting.

Nitrification is defined as the biological conversion of NH_4^+ to NO_3^- with NO_2^- as an intermediate in the reaction sequence. On contrary, denitrification is a four-reaction steps converting NO_3^- to N_2 via intermediaries (NO_3^- to NO_2^- to NO to N_2O to N_2) (Vymazal, 2007). While the major product of denitrification is N_2, release of gaseous like NO and N_2O may also occur (Faulwetter et al., 2009).

Although heterotrophic nitrification occurs and is recognised to be significant, nitrification has been typically associated with chemoautotrophic bacteria (Paul & Clark, 1996). Thus, nitrifying bacteria derive energy through oxidation of NH_4^+ and or NO_2^- and carbondioxide is used as a carbon source to synthesize new cell (Vymazal, 2007). The major factors influencing nitrification in CWs are DO, inorganic carbon source, NH_4^+-N concentration, temperature, microbial populations temperature, pH Value and alkalinity of water (Vymzal, 1995). However, the denitrifiers make use of NO_3^- as the terminal electron acceptor under anaerobic environments and in the presence of carbon source to derive energy for new cells synthesis (Lee et al., 2009). Therefore, the major factors that may have an influence on denitrification are absence of oxygen, NO_3^--N concentration, carbon source and presence of the heterotrophic denitrifiers among others (Vymazal, 1995). Denitrifying bacteria are categorised into heterotrophs and autotrophs (Rijn et al., 2006). To date, the heterotrophic denitrification process is primarily involved in wastewater treatment, although autotrophic denitrification has lately been studied (Kim et al., 2004).

However, the removal of nitrogen from SSF CWs treating domestic wastewater with a low carbon/nitrogen (C/N) ratio can often be limited since organic carbon is a limiting factor for denitrification (Lee et al., 2009). Furthermore, lack of DO concentration in wastewater is one of the factors that limits nitrification in SSF CWs (Ouellet-Plamondon et al., 2006). Although nitrification occurs at low DO concentrations, the reaction rate is significantly lower when the DO concentration falls below 2 mg/L (Tang et al., 2009). Plants improve nitrification in CWs through supply of oxygen in the rhizosphere (Brix, 1994; Faulwetter et al., 2009). However, Wang et al. (2015) proposed that root oxygen release is far less than the amount required to support the nitrification process. Therefore, to achieve effective nitrification especially in V-H hybrid CWs, artificial aeration is suggested as a means of creating an aerobic environment in the VSSF CW (Pan et al., 2012) while Tchobanoglous et al. (2003) recommend a requirement of an extra carbon source for effective denitrification. Both suggested options may seem to increase wastewater treatment costs and thus, may not be affordable in the developing world.

However, advancement in biotechnology shows that combining various wastewater treatment systems may be crucial for effective nitrogen removal from wastewater. For instance, replacing VSSF CWs with high rate algal oxidation ponds (HRAOPs) in the V-H hybrid CW system, commonly referred to as the HRAOP-HSSF CW hybrid system has proved a very promising technology for efficient nitrogen removal from wastewater (Ding et al., 2016). The HRAOP is the secondary treatment unit in the advanced integrated wastewater pond system (AIWPS®) or integrated algae pond system (IAPS) (Oswald et al., 1994; Green et al., 1996, Mambo, 2016). Although a HRAOPs requires an external energy source to drive the paddle wheel for it to effectively function, the energy demand of this system is very low compared to that of conventional WWTPs (Craggs et al., 2014). In a HRAOP-HSSF CW hybrid system, the HRAOP replaces the VSSF CW with two major objectives: 1) the high algal productivity of HRAOPs results into algal debris and thus, increased carbon source for denitrifying bacteria and 2) photosynthetic algal productivity increases DO concentration for improved nitrification (Ding et al., 2016). Thus, with increase in DO, HRAOPs show high nitrification rates and denitrification is accomplished in the HSSF resulting into effective nitrogen removal from wastewater (Ding et al., 2016). Although the HRAOP-HSSF hybrid CW is a potential biotechnology for municipal wastewater treatment, there are limited pilot scale studies reported in literature and no reports on full-scale application of this system. Pilot-scale studies on the performance and associated costs of HRAOP-HSSF CW hybrid system from various climatic regions are highly recommended to provide adequate information about these systems especially in areas where strict effluent nitrogen standards exist.

1.2.6 Alternative biotechnological application of subsurface flow constructed wetlands

Since the introduction of SSF CWs in the middle of the past century, they have been mainly reported as a promising technology for treatment of wastewater of various type including municipal sewage (Vymazal, 2008; Vymazal, 2010). However, one of the challenges of treating municipal wastewater in SSF CWs is attaining effluent standards for discharge to the environment particularly regarding NH_4^+-N concentration (Section 1.2.2.2). As previously mentioned (Section 1.2.3.1), this is mainly attributed to design limitation since SSF CWs are designed using BOD rather than NH_4^+-N which underestimates the surface area requirement for nutrients. However, other studies such as Lee et al. (2009) point it to the low C/N ratio of NH_4^+-N-rich wastewater including domestic/municipal sewage since organic carbon is a limiting factor for denitrification. Hence, due to the low C/N ratio of most wastewaters, conventionally, additional organic carbon dozing is required (Du et al., 2016). The limitations of this operation strategy include: i) increase in the operational costs i.e. to purchase a carbon source and energy for the dozing process; and, ii) requirement of skilled personnel and sophisticated equipment whose spare parts may not be locally available especially in the developing world.

Thus, it can be proposed that, the low C/N ratio of wastewater and the resulting poor water quality pertaining NH_4^+-N concentration obtained from SSF CWs (Section 1.2.2.2) could be improved by replacing gravel that is traditionally employed as support media for macrophytes with solid waste materials such as discard coal that is highly carboneous. Although experimental data is highly required to ascertain the feasibility of the use of discard coal in SSF CWs as an alternative treatment media to gravel, it can be suggested that this application may be a more economically and environmentally friendly option to dozing of wastewater treatment systems with extra organic carbon since discard coal may readily provide a cheap source of organic carbon. Although not yet ascertained, carboneous discard coal could play an important role in influencing denitrification and therefore improving nitrogen removal which is always a problem is gravel-based SSF CW.

Besides, discard coal is as an industrial by-product with potential environmental impacts in coal mining countries including Australia, India, Brazil and South Africa to mention a few. Waste coal dump pollution arises when pyrite (iron disulphide) a major component of waste coal is exposed to oxygenated water (rainwater), thus, undergoing oxidation and generating sulphuric acid as shown in equation iv below.

$$FeS_2 + 7.5O_2 + 7H_2O \longrightarrow 2Fe\,(OH)_3 + 4H_2SO_4 \qquad \text{(iv)}$$

The negative impacts of discard coal dumps to the environment and public health are discussed as follows. Firstly, decantation of the highly acidic medium into the ground increases solubility of certain heavy metals including Uranium (U), Mercury (Hg), Lead (Pb), Arsenic (As) etc, which may leach into the ground polluting ground and surface water sources (Swetti and Geetha, 2013). Metals like mercury inform of methylmercury have been reported to accumulate in seafood e.g. fish and shellfish and consumption of these contaminated foodstuffs by humans has been reported to cause toxic effects on the nervous, digestive and immune systems, lungs, kidneys, skin and eyes (WHO, 2017). Secondly, waste coal contains various elements such as Manganese (Mn), Iron (Fe), Magnesium (Mg), Calcium (Ca), Sulphur (S) etc. (Appendix 14) some of which are detrimental to both aquatic and human health when they leach out of the waste coal and find their way into the environment. For instance, water decanting from waste coal dumps and discharging into surface waterways is usually "hard" attributed to high concentration of dissolved minerals particularly Ca and Mg and to less extent Fe that can be a nuisance due to build-up of minerals in water distribution pipes and poor scum formation with soap and/or detergent. Lastly but not least, Fe^{2+} is precipitated as Fe^{3+}, which usually settles out as brown insoluble precipitate. This is usually a menace to drinking water supplies since it stains laundry. The presence of Fe may also encourage growth of iron bacteria some of which are hazardous to human health (WHO, 2003).

To mitigate any detrimental environmental impacts arising from discard coal dumps, several bioremediation alternatives are in place. In South Africa, the traditional method involves covering the dump with about 30-100 cm layer of top soil, fertilizers are applied, and the top soil is seeded with leguminous grass species as a biocatalyst (Rethma & Tanner, 1995; Cowan et al., 2016). This is referred to as dry land bioremediation or phytoremediation. The major drawbacks of this technique include: i) in cases where the topsoil is not locally available, it must be sourced and transported to the site under rehabilitation, ii) from the area where the top soil is extracted, there is likely to be heavy environmental damage including erosion and sedimentation of particulate solids into water ways. Even after successful plant establishment and colonisation, this approach does not necessarily result into breakdown of the underlying waste coal and indeed, may only mask and delay future environmental catastrophes (Cowan et al., 2016).

Thus, one approach with a potential to mitigate environmental degradation resulting from discard coal dumps is to treat discard coal as CWs. Besides providing a carbon source necessary for improving nitrogen removal, the use of discard coal as a support media in SSF CWs may help to alleviate the negative impacts associated with its disposal into the environment. The reason is that CW macrophytes have been reported to buffer pH (Vymazal & Kröpfelová, 2008) hence are likely to buffer highly acidic pH that is expected from discard coal based CWs to near neutral. Additionally, CW macrophytes have also been reported to uptake and accumulate metals in form of biomass (Rousseau, 2005; Dhir, 2013; Zingelwa & Wooldrigde, 2016). While information about metal concentration from discard coal based CWs is scarce, it can be hypothesized that the use of discard coal as a treatment media in SSF CWs may generate an effluent with reduced metal concentration. Clearly, this suggests that research into the application of discard coal as a support media in SSF CWs to ascertain the water quality from these systems in comparison to gravel based SSF CWs is highly recommended.

1.2.7 Cost aspects of subsurface flow constructed wetlands

The costs associated with establishing a wastewater treatment system is an important consideration mainly in developing countries where wastewater treatment is under-prioritized which is important for policy and decision makers to decide on the most cost-effective wastewater treatment technology for implementation. The economics of CWs entails two distinctive categories: The initial investment/capital costs and operation and maintenance costs. The major costs associated with a wastewater treatment system are capital costs (Rousseau et al., 2008, Stefanakis et al., 2014) and for SSF CWs these are:

- Land procurement (where land is not available),
- Landscape grading, site investigation and facility design,
- Lining to prevent groundwater contamination,
- Treatment media (e.g. sand or gravel), labour,

- Vegetation and planting,
- Hydraulic equipment (pumps, valves, pressure systems etc.) and pipes; such as distribution pipes, drainage pipes and aeration pipes.

Nevertheless, after life span of the CW, the land can be freely made available for other uses hence this cost is sometimes excluded from the balance. Capital costs are particularly dependent on the local conditions including such as soil type, groundwater table height, terrain slope, distance from settlement, discharge criteria, climate and land accessibility. The economy of scale is another important factor with larger wetlands being quite cheaper per PE or m^3 of wastewater treated (Rousseau et al., 2008). Additionally, the economic status and parameters vary from one country to the other and over time, thus a corresponding change and variability in the estimation of costs as shown below.

Table 1-10. Examples of capital cost for some of the subsurface flow constructed wetlands designed for BOD removal.

Type of SSF CW	Costs/PE	Unit	Country	Reference
HSSF	102.2	€	Kenya	Mburu et al. (2013)
HSSF	300	$	Uganda	Okurut (2000)
VSSF	310.85	€	Greece	Tsihrintzis et al. (2007)
VSSF	507	€	Belgium	Rousseau (2005)
Not defined	42	$	Nicaragua	Platzer et al. (2002)

While a major drawback of SSF CWs is the cost of the treatment media, the CW technology is regarded as a cheaper natural wastewater treatment system in terms of operation and maintenance than ASS. The most common treatment media used in CWs is gravel of varying size. While major drawback of using gravel in CW is its low adsorption capacity for phosphorus (Rhue & Harris, 1999), on the economic point of view, the cost of gravel has always been a major expense for the CWs accounting for about 30-50% of the total investment costs (Masi & Bresciani, 2016). In a survey of the capital costs for the SSF and FWS CWs, US. EPA (1993) reported an average capital costs for the SSF system of ~ $ 200,000/ha in comparison to $ 50,000/ha for FWS system. The difference in the capital costs of the two systems was attributed mainly to the cost of procuring the gravel media, hauling it to the site and placing it. Whilst there is little information on the costs of establishing SSF CWs, there is also a need to explore the potential of application of other locally available treatment media that can be used as an alternative to gravel in CWs to lower the capital costs associated with SSF CWs.

1.3 Aims and objectives

This chapter provided an overview of CW use in domestic wastewater treatment. The following conclusions can be drawn:

- Subsurface flow CWs are the widely adopted systems for municipal/domestic wastewater treatment due to their environmental benefits.
- While there is limited information about the quality of treated water from SSF CWs operating in tropical and subtropical regions, a literature survey revealed that all SSF CWs produce a high effluent quality regarding organic matter, TSS and particulate nutrients. However, HSSF CW produced a better effluent quality concerning NH_4^+-N than either VSSF or V-H SSF hybrid CW.
- All the surveyed SSF CWs produced an effluent quality that met the effluent discharge standards for most environmental regulatory bodies in sub-Saharan Africa with regards to TSS, COD, BOD, TP but not for NH_4^+-N. In view of this information, the main aim of this thesis is therefore to study the performance of a V-H SSF hybrid CW designed using NH_4^+-N as a design parameter.

The specific objectives include:

i) Design and construct a V-H SSF hybrid CW for using NH_4^+-N as a major parameter.

ii) Operate, monitor and evaluate the effluent quality from a V-H SSF hybrid CW.

iii) Investigate the potential of using discard coal as an alternative treatment media to gravel in a HSSF CW.

iv) Evaluate the use of a V-H SSF hybrid CW and a series of HRAOP to supplant the oxidation pond component of a dysfunctional WSP system.

Chapter 2 Design and construction of a vertical horizontal subsurface flow hybrid constructed wetland

2.1 Introduction

Worldwide, environmental governing bodies set strict effluent discharge limits for wastewater treatment plants for release of nitrogen into fragile surface watercourses that are susceptible to eutrophication. In South Africa for instance these are ≤ 15 mg/L and ≤ 6 mg/L for NO_3^--N and NH_4^+-N respectively (DWS, 2013).

However, a detailed evaluation of published studies (Chapter 1) on the performance of V-H hybrid SSF CWs showed that these systems do not necessarily meet the NH_4^+-N effluent discharge limit into surface water courses for most of the environmental regulating bodies in sub-Saharan Africa (Appendix 1). It was concluded that the poor NH_4^+-N effluent quality was mainly due to the design of SSF CWs, which has traditionally been based on BOD as the target pollutant rather than NH_4^+-N (Chapter 1). Since a major objective of V-H SSF hybrid CWs is to optimize nitrogen removal and considering the theory that the design of CWs should be based on the treatment objective (Reeds et al., 1995), the primary aim of this work was to design and construct a V-H SSF hybrid CW based on nitrogen as a design parameter rather than BOD.

In addition, CWs are essentially a downstream treatment process and usually configured following primary treatment. Although several primary treatment processes options are available (e.g. septic tank, anaerobic baffle reactor, anaerobic ponds, etc.), the current project was confined to an advanced facultative pond (AFP), a component of the pilot-scale integrated algal pond system (IAPS). Thus, part of the exercise was to determine whether a V-H SSF hybrid CW could be used to supplant the high rate algae oxidation ponds (HRAOPs) of the IAPS to achieve a final effluent of a similar or better quality and the same time, reduce the overall footprint of the wastewater treatment process. This was considered important as WSPs are widely adopted sewage treatment processes in South Africa and other South African countries. Furthermore, many of these WSP systems are dysfunctional or overloaded and IAPS has been mooted as a technology suitable for conversion of WSP systems to increase capacity of the performance without incurring unnecessary costs. Thus, the aim of the work described in this chapter is design and construction of V-H SSF hybrid CW using nitrogen as a design parameter for implementation after an AFP.

2.2 Material and Methods

2.2.1 Hybrid constructed wetlands design

Design of the V-H SSF hybrid CW involved two stages: (1) estimation of the surface area required for the hybrid CWs; and, (2) estimation of cell dimension/configuration of the V-H SSF hybrid CW (Figure 2-1).

2.2.1.1 Estimation of surface area for hybrid CW

Effective pollutant removal from CWs depends on using an effective area for a given flow. The total area of the subsurface flow bed was estimated using Reeds et al. (1995) first order kinetic model (Chapter 1, equation ii) by employing NH_4^+-N as the critical parameter rather than BOD since BOD as design parameter under-estimates the area requirement for nutrient removal (Vymazal, 2005). NH_4^+-N was employed in the calculation due to the following reasons: among the nitrogenous compounds viz: NO_3^-, NO_2^-, NH_4^+, urea (CH_4N_2O), amino acids (R-CH(NH2)-COOH) etc. (Fauwetter et al., 2009), the most important inorganic nitrogen species in wastewater are NO_3^-, NO_2^- and NH_4^+. However, among these nitrogen species, NH_4^+ is the most abundant (Vymazal, 2007) and the most preferred nitrogen form for most algae species for uptake and assimilation since it is more reduced energetically than either NO_3^- or NO_2^- (Kadlec & Knight, 1996; Rücket & Giani, 2004). Thus, it is proposed that it could be the most important nitrogen species responsible for eutrophication.

A summary of parameters employed in the calculation is given below.

- A conservative design inflow rate (Q) of 0.1 m^3/d was considered.
- The average temperature of the coldest month (July), T_w was taken as 17 ^oC (Rhodes University, online metrological data, 2014).
- Influent NH_4^+-N concentration (C_{in}) of 12.1 mg/L, which corresponds to the average effluent NH_4^+-N concentration from the primary treatment unit (AFP) was obtained from literature (Rose et al., 2002).
- Effluent NH_4^+-N concentration (C_{out}) of 3 mg/L was employed (South Africa Water Act, 1998).

- The depth of the bed occupied by the root zone (rz) of 95 % (0.95) was assumed since roots of the macrophyte used (i.e. *P. australis*) can penetrate to a depth of about 0.4 m (Reeds et al., 1995).
- Porosity n, of treatment media was determined using the direct method of porosity determination (https://en.wikipedia.org/wiki/Porosity#Measuring_porosity).

As a custom, gravel was chosen as a treatment media for the hybrid CWs due to its local availability. Firstly, a field study was conducted to find out the availability of gravel and, 5 kg of 14 mm were obtained from Amatola Quarry Products, Tempe Farm, Grahamstown for porosity determination.

In the laboratory, the gravel was washed to eliminate any fine particles. For bulk volume determination (V_b), a 1000 mL beaker was filled with gravel to a depth *h*. The height of the gravel in the beaker was recorded (Appendix 2). V_b was then determined by calculation using the expression;

$$V_b=\pi r^2 h \tag{v}$$

Where: V_b= bulk volume (cm^3), π= 3.14, r^2=radius of the beaker (cm) and *h*= height of the gravel in the beaker (cm).

To establish the pore volume (V_p) of the substrate, the weight in g of the beaker was recorded. Tap water was then poured into the beaker until it just covered the surface of the gravel and the weight in g was recorded and V_p, which corresponds to the amount of water needed (1g=1 cm^3) was calculated according the expression below;

$$V_p=W_2-W_1 \tag{vi}$$

Where: V_p=pore volume (cm^3), W_2=Weight of the beaker + gravel + tap water (g) and W_1= weight in g of the beaker + gravel (g).

Then porosity was calculated using the equation:

$$n=V_p/V_b \tag{vii}$$

Where: n= porosity, V_p=pore volume (cm^3) and V_b=bulk volume (cm^3).

A summary of the parameters employed in the calculations is given in Appendix 3; and by employing equation vii, n=0.43 was derived.

Finally, by employing equation ii (i.e. $A_s=Q_{in}(lnC_{in}-lnC_o)/K_Tyn$) and values of parameters previously outlined in section 2.2.1.1 of this chapter, the total area of the SSF CW was estimated as 3 m^2. While adopting the theory that the area of the VSSF is half that of the HSSF (Tousignant et al., 1999), the surface area of the HSSF and VSSF CWs of 2 and 1 m^2 respectively were derived.

Additionally, to investigate the theory that using BOD as a design parameter for SSF CWs under-estimates the area requirement for nutrient removal (Vymazal, 2005), the area requirement of the hybrid CW using BOD as a design parameter was also estimated by employing Reeds et al. (1995) model according to the following description. Although BOD is widely used as a design parameter for SSF CWs, in South Africa, BOD is not among the parameters stipulated in the DWS (2013) effluent discharge standards. Therefore, it is rarely monitored. In the present study however, BOD concentration after primary treatment (C_i) was first estimated from COD concentration (308 mg/L), which was borrowed from Wells (2005) using a general rule that the ratio of BOD:COD in domestic wastewater is 0.5. Also, since South Africa doesn't have BOD effluent discharge standard, a BOD concentration of 30 mg/L from one of the countries in Southern Africa (Zimbabwe) was considered (Nhapi, 2004).

A summary of parameters employed in the calculation is given below.

- A conservative design inflow rate (Q), average temperature of the coldest month, treatment depth y and porosity n of the treatment media are similar to those previously described while using NH_4^+-N as a design parameter.
- Influent BOD concentration (C_{in}) of 154 mg/L.
- Effluent BOD concentration (C_{out}) of 30 mg/L was employed. K_R and θ_R of 1.104 and 1.06 were considered (Reeds et al., 1995).

Thus, by employing Reeds et al. (1995) model, a surface area of ~ 1 m^2 was estimated.

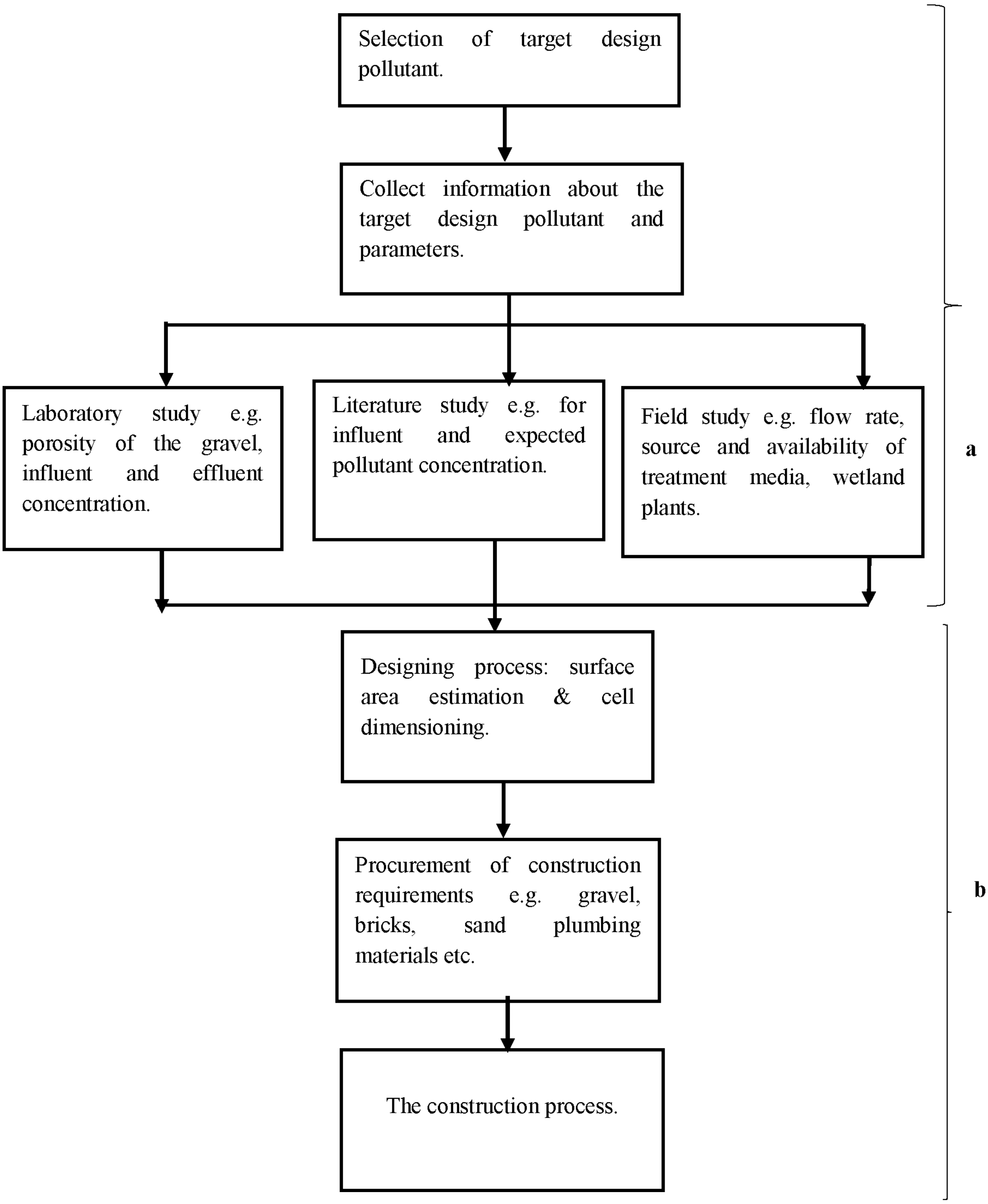

Figure 2-1. A conceptual model for design and construction of a subsurface flow constructed wetland. Design process (a) and construction process (b).

2.2.1.2 Hybrid CW cell dimensions/configuration

Aspect ratio, which refers to the ratio of length to width, is an important design consideration especially for HSSF CWs. A unit increase in aspect ratio has a corresponding increase in hydraulic retention time (García et al., 2004) and therefore improved performance. While an aspect ratio ranging 0.25:1 to 5:1 is considered sufficient for HSSF CWs (Wood, 1995), in this study, an aspect ratio of 3:1 was taken into consideration resulting into ~2 m and ~1 m for length and width respectively while adopting a total bed depth of 0.55 m. However, since there is no literature regarding the aspect ratio requirement for VSSF CWs, it was assumed that this was not necessary so, the length, width and total bed depths of the VSSF CWs were 1 m each. Table 2-1 gives a summary of the design considerations for the V-H SSF hybrid CWs.

Table 2-1. Summary of design aspects for the V-H SSF hybrid CWs.

Parameter	Hybrid CWs	VSSF CWs	HSSF CWs
Design HLR (mm/d)	33		
Influent NH_4^+-N loading rate ($g/m^2/d$)	0.4		
HRT (d)	3		
Length (m)		1	2
Width (m)		1	1
Total depth (m)		1	0.55
Treatment depth (m)		0.7	0.4
Slope (%)		1	1

2.2.2 Gravel requirement for the hybrid CWs

Different gravel sizes were used in the V-H SSF hybrid CWs and these were chosen based on the specification by the UN- Habitat (2008) with little modifications based on the local availability at the quarry. Initially, the amount of gravel required for the V-H SSF hybrid CWs was calculated to determine the quantity that would be purchased, which was calculated based on the bulk volume of the CW compartment that would be occupied by the specified gravel size. Details of the calculations of the quantity of gravel for the V-H SSF hybrid CWs is given in Appendix 3 while Table 2-2 gives a summary of the quantity of different gravel sizes employed in the hybrid CWs. In total, 4 m^3 of gravel of different sizes described in the Table 2-2 were purchased from Amatola Quarry Products, Tempe Farm, Grahamstown.

Table 2-2. Gravel requirement for the V-H SSF hybrid constructed wetlands

Item no.	Gravel size	Quantity (m^3)
1	Dust (up to 8 mm)	1.5
2	14 mm	1.5
3	Scalping (22-54 mm)	1

2.2.3 Construction of the hybrid CW

The designed pilot scale V-H SSF hybrid CWs were constructed at the Institute for Environmental Biotechnology (EBRU) Rhodes University, in Eastern cape, South Africa (33° 19’ 07” South, 26° 33’ 25” East) in July 2014. Before construction, a suitable site for construction of the hybrid CW was selected on the basis that water from the primary treatment unit, the advanced facultative pond (AFP), flowed into the V-H SSF hybrid CWs by gravity.

The V-H SSF hybrid CWs comprising of VSSF and HSSF cells was constructed in duplicate using the design specifications given in Table 2-1. For the VSSF CW section, plastic containers (1 m^3) with outlets to allow flow of water into the HSSF CWs were used. The HSSF CW components (2×1×0.55 m) were constructed of concrete with a bottom slope of about 1% to allow for drainage and the cells lined with polyvinyl chloride (PVC) fabric (0.2 mm thickness) to avoid seepage (Figure 2-2a).

Figure 2-2. Construction aspects of V-H SSF hybrid CW. Lining the HSSF CW with PVC (a) hybrid CW filled with gravel (b)

Gravel previously described was used as media for the V-H SSF hybrid CWs (Figure 2-2b). Again, gravel arrangement in the V-H SSF hybrid CWs was according to the specification by the UN-habitat (2008) with little modification based the wetland size. Prior to filling the CWs, gravel was sorted and washed to eliminate fine particles and residual organic matter that might cause clogging in the system.

As recommended, in the VSSF, gravel was arranged vertically in layers covering a depth of 0.7 m (Figure 2-3). From this, the bottom and surface layers were 0.15 m and 0.05 m respectively and comprised gravel of particle size 22-58 mm while the support layer (0.05 m) comprised gravel of 14 mm. The treatment layer (0.45 m) was of fine gravel (1-8 mm).

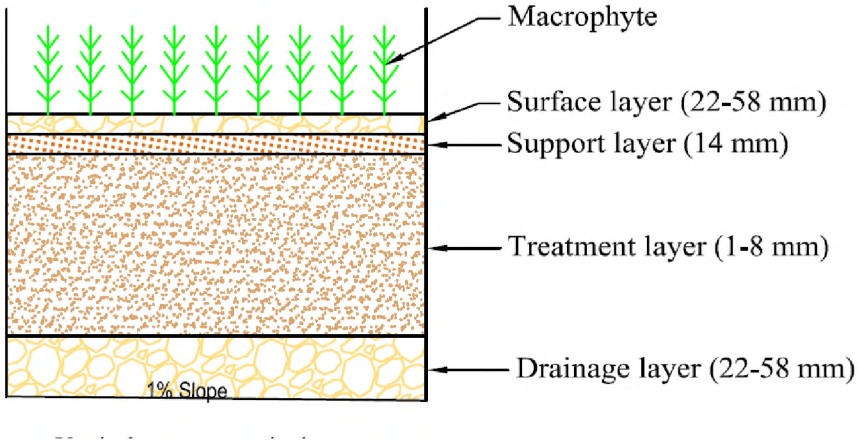

Figure 2-3. Diagrammatic representation of the vertical arrangement of gravel in the VSSF CWs.

On the other hand, while it is recommended that the media in the inlet and outlet zones of the HSSF CW should range from 40-80 mm in diameter to minimize clogging and ensure uniform distribution of wastewater, in this study, for that purpose, gravel ranging from 22 to 54 mm was used. Furthermore, due to the small size of the HSSF CWs, it was not possible to adopt the inlet and outlet zone lengths of 0.75 m each as recommended. Therefore, the inlet and outlet zones covered a total length of 0.4 m, reason being that; it was considered impractical for the two zones to cover a total length of 1.5 m as this would result in a small treatment zone. Finally, the treatment layer (0.4 m deep) was filled with gravel of particle size 14 mm since a recommendation of 5-20 mm is given and this covered a length of 1.6 m. An outlet standpipe was installed to keep wastewater at about 0.3 m below the bed surface as shown in Figure 2-4 below.

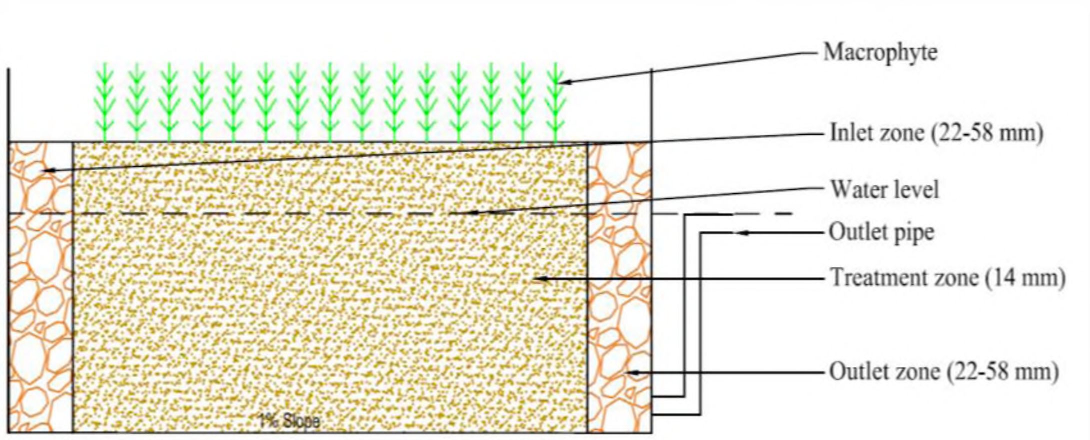

Figure 2-4. Diagrammatic representation of the arrangement of gravel in the HSSF CWs.

Artificial aeration can improve the flow pattern to mitigate clogging and oxygenation level of the rhizosphere especially when temperatures decline during winter (Munoz et al., 2006; Yan & Xu, 2014). Thus, for this purpose, aeration pipes (40 mm id) drilled with holes (5 mm diameter) were installed in the hybrid CWs at 9 pipes/m^2. Due to fluctuations in the flow rate at the AFP outflow, two buffer tanks (250 L each) were installed before the VSSF CWs to ensure constant influent flow rate. These tanks were designed with overflows to maintain a fixed water level and the outlet pipes, fitted with valves to control the flow rate, were positioned to ensure the required flow of 0.1 m^3/d (100l/d) into the CWs. Thus, the pilot scale V-H SSF hybrid CW employed in the present study was designed to treat wastewater of 1 PE.

Contrary to the recommendation to uniformly distribute the influent wastewater in the VSSF CWs via a network of pipes (Stefanakis et al., 2014); a single-entry point was eventually used. This was due to constant clogging of the inlet that was initially installed. PVC pipes (20 mm i.d.), perforated with holes (5 mm diameter) were used to deliver wastewater from the VSSF CWs into the HSSF CWs by evenly distributing the flow along the width of the cells to permit plug flow and minimize short circuiting (Tousignant et al., 1999).

In this study, *P. australis* was the macrophyte of choice. A literature review (Chapter 1) revealed that among the common macrophytes used in CWs, *P. australis* has the highest nitrogen uptake potential and that this species harbours most species of bacteria involved in total nitrogen cycle.

Based on this information and availability of *P. australis* locally, each V-H SSF hybrid CW was flooded with AFP effluent and planted with uniform shoots of *P. australis* (9 plants/m^2) that were collected from the banks of a nearby river.

2.3. Results

Table 2.3 below presents the estimated surface area for the SSF CW at various NH_4^+-N effluent discharge limits and considering the NH_4^+-N wastewater composition after primary treatment of 12.1 mg/L (Rose et al., 2002) while Figure 2-5 presents a completed fully constructed V-H SSF hybrid CW with duplicate cells.

Table 2-3. Estimated surface area for the V-H SSF CW using an influent NH_4^+-N concentration of 12.1 mg/L at different NH_4^+-N effluent discharge limits.

NH_4^+-N discharge limit (mg/L)	1.0	2.0	3.0[a]	4.0.	5.0.	6.0[b]
Estimated area (m^2)	4.0	3.0	**3.0**	2.0	2.0	**1.0**

*Estimated surface area for SSF CW using BOD is 1 mg/L. [a] and [b] correspond to the DWS effluent discharge limit for NH_4^+-N in 1998 and 2013 respectively.

Results revealed that initially, the surface area of 3 m^2 estimated for the V-H SSF hybrid CWs using the 1998 NH_4^+-N discharge limit, is three times bigger than that would be required while using BOD as a design parameter. However, as shown Table 2-3 above, with an increase in the NH_4^+-N discharge limit from 3 mg/L in 1998 to 6 mg/L in 2013, the surface area estimated using NH_4^+-N becomes equal to that while using BOD as a design parameter.

Figure 2-5 below shows a V-H SSF hybrid CW immediately after planting and following establishment of *P. australis* prior in to initiation and optimization of performance evaluation studies. After planting, *P. australis* established successfully in the system.

Figure 2-5. Fully constructed pilot-scale V-H SSF hybrid CWs immediately after planting (a) and 3 months after establishment of *P. australis* (b).

2.3 Summary

The summary of the findings from the design and construction of the V-H SSF CWs are as follows:

- The area estimated for the V-H SSF hybrid CW using NH_4^+-N as a design parameter of 3 m^2 is three times bigger than that would be required while using BOD as a design parameter.
- However, the surface area estimated using NH_4^+-N becomes equal to that while using BOD with an increase in the NH_4^+-N discharge limit from 3 mg/L in 1998 to 6 mg/L in 2013.

Chapter 3 Performance evaluation of a vertical-horizontal subsurface flow hybrid constructed wetland

3.1 Introduction

Worldwide, point source discharge of partially or untreated municipal sewage is a major cause of surface water pollution, eutrophication and deterioration of aquatic ecosystems (Mustapha & Getso, 2014). In addition, disease outbreaks including cholera and diarrhoea are common occurrences (Corcoran et al., 2010). The South African Department of Water and Sanitation (DWS) has established effluent discharge standards for the release of treated water into watercourses and for WWTPs. These are: 75 mg/L, 25 mg/L, 6 mg /L, 15 mg/L, 10 mg/L and 1000 CFU/100 mL for COD, TSS, NH_4^+-N, NO_3^--N + NO_2^--N, PO_4^{3-}-P and faecal coliforms (FC) respectively (Republic of South Africa Government Gazette No. 36820, 2013).

To meet these stringent discharge standards especially for nitrogen, studies have focused on combining different types of WWT technologies and of late, V-H SSF hybrid CWs have been proposed as a promising wastewater treatment technology especially for nitrogen removal from domestic sewage (Vymazal, 2013). As reported earlier in chapter 1, it is hypothesized that intermittent operation of a VSSF CW results in oxygen transfer into the bed allowing conversion of ammonium to nitrate (nitrification) (Brix & Arias, 2005). However, Vymazal (2007) and Molle et al. (2008) state that continuous feeding and maintaining the water level below the treatment media in the HSSF CW limits contact between air and water leading in limited transfer of oxygen into the system thus, resulting in anoxic/anaerobic conditions that are responsible for converting nitrate to nitrogen gas (denitrification). This results in reduced total nitrogen in the final effluent.

Although earlier studies reported that V-H SSF hybrid CWs are more efficient in ammonia, organic matter and TSS removal than single stage HSSF or VSSF CWs (Vymazal, 2013), Chapter 1 showed that single stage CWs, particularly HSSF, were better than V-H SSF hybrid CWs at ammonium and organic matter removal. This was attributed to continuous feeding of HSSF CWs, which results in longer HRT allowing enough contact time for the wastewater to interact with the biofilm to facilitate efficient pollutant removal. In contrast, in the V-H SSF hybrid CW system, the VSSF CW is normally operated intermittently, and this therefore results in overall short HRT and therefore lower pollutant removal than HSSF CWs. It was therefore concluded that continuous feeding could be the most appropriate mode of feeding CWs. Although intermittent feeding is proposed as an appropriate mode of operation for the vertical flow systems and to achieve fully nitrified conditions (Brix & Arias, 2005), dosing of the system with wastewater requires energy input which may not be economically feasible especially in developing countries where energy supply is erratic and unreliable.

The study described in this chapter was carried out to determine the quality of treated water from a V-H SSF hybrid CW designed using NH_4^+-N as a target parameter, which were operated continuously for a period of two growing seasons. The major aim of this study was therefore to establish whether water quality from a V-H SSF hybrid CW designed using NH_4^+-N as a parameter meets the General Authorizations for discharge to the environment. Results are discussed in terms of the importance of parameter selection in the design of CWs especially if nitrogen is a critical parameter for remediation.

3.2 Material and methods

3.2.1 Pilot-scale hybrid constructed wetland

The pilot-scale V-H SSF hybrid CWs designed and constructed in parallel as described in Chapter 2 were used. The V-H SSF hybrid CWs were located at Institute for Environmental Biotechnology Rhodes University (EBRU) which is situated within the premises of Belmont Valley WWTP, in Grahamstown, Eastern cape, South Africa (33° 19' 07" South, 26° 33' 25" East). The mean meteorological data of Grahamstown during the study period is summarized in Table 3-1.

Table 3-1. Mean ± SE meteorological data for Grahamstown recorded from February 2015-July 2016.

Air temp (°C)	Precipitation (mm/d)	Humidity (%)	Wind speed (Km/hr)
15.9 ± 0.8	1.2 ± 0.2	71.4 ± 2.5	12.8 ± 0.9

Number of data (n) for all parameters is 16. Data was obtained from https://www.wunderground.com.

The systems were operated simultaneously by feeding effluent from an advanced facultative pond (AFP) into the CWs via buffer tanks at a rate 0.1 m^3/d. Continuous feeding was allowed to proceed for 5 months after planting of the macrophyte *P. australis*, to allow for adequate equilibration. The process flow for the pilot-scale hybrid CW and colonisation of the substrate media by *P. australis* 5 months after planting are shown in Figure 3-1.

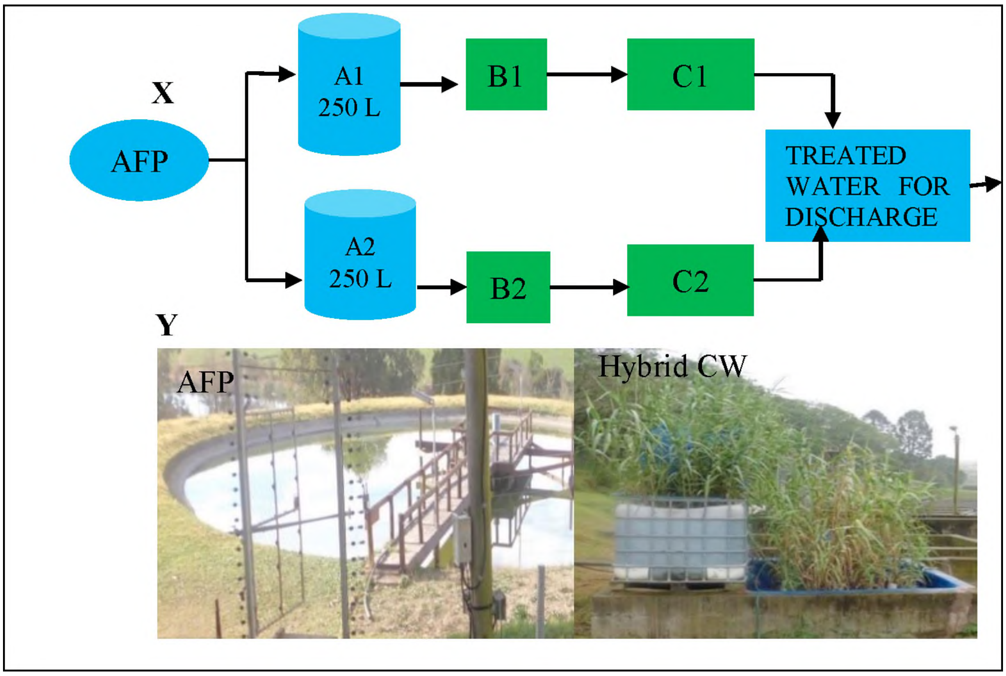

Figure 3-1. Process flow for the pilot scale hybrid CW (X) and image showing V-H SSF hybrid CWs containing *P. australis* 5 months after establishment (bottom left). A1 and A2=holding tanks, B1 and B2=VSSF CWs and C1 and C2=HSSF CW.

3.2.2 Operation and monitoring of the hybrid CW

Operation and monitoring of the V-H SSF hybrid CWs was carried over two growing seasons and in two phases.

Phase 1 or system initiation was carried out from February 2015 to July 2015. During system initiation, partially treated wastewater from the AFP flowed continuously under gravity into the V-H SSF hybrid CWs. A mean HLR of 56.0 ± 3.8 mm/d was determined, yielding an approximate theoretical HRT of ~2 d (Table 3-3). Following initiation phase (i.e. the first growing season, July 2015), the above ground biomass was harvested. The V-H hybrid SSF CWs were then rested for two months to allow plants to re-establish before recommencing performance monitoring in October 2015.

Phase 2 or system optimisation (i.e. from October 2015 to July 2016) was aimed at demonstrating improvement in the performance of the hybrid CWs by varying the operational parameters (i.e. HLR, HRT and influent NH_4^+-N loading rate) in order to further improve the water quality. During phase 2, the V-H SSF hybrid CWs were further operated under two sub phases and these were:

(i) A high NH_4^+-N loading rate phase (HP) from October 2015 to March 2016, where the V-H SSF hybrid CWs were fed with AFP effluent but the HLR was reduced to a mean 39.0 ± 1.2 mm/d, to increase the HRT from 2 to ~3 d with a mean NH_4^+-N loading rate of 1.09 ± 0.09 g/m²/d (Tables 3-2 & 3-3) and,

(ii) A low NH_4^+-N loading phase (LP) from April to July 2016, where the HLR was maintained at 37.0 ± 0.3 mm/d with a HRT of ~3 d. The effluent from the AFP was diluted to achieve an influent NH_4^+-N concentration of 14.8 ± 1.0 mg/L close to the design value of 12.1 mg/L to yield a mean influent NH_4^+-N loading rate of 0.55 ± 0.03 g/m²/d (Tables 3-2 & 3-3).

Table 3-2 provides a summary of the V-H SSF hybrid CWs hydraulics and operational parameters while Table 3-3 presents the physico-chemical characteristic of wastewater fed into the hybrid CWs during the different operational phases.

Table 3-2. Hydraulic and operational parameters of the V-H SSF hybrid CWs during system initiation and optimization.

Parameter (unit)	System Initiation (Feb-Jul 2015)	System Optimisation (Oct 2015-Jul 2016) HP (Oct 2015-Mar 2016)	LP (Apr -Jul 2016)
Inflow rate (HLR) (mm/d)	56.0 ± 3.8, (6)	39.0 ± 1.2, (5)	37.0 ± 0.3, (8)
Outflow rate (mm/d)	35.0 ± 6.9, (6)	18.0 ± 3.2, (5)	13.0 ± 3.4, (8)
HRT (d)	2	3	3
Precipitation (mm/d)	1.4 ± 0.4, (6)	1.2 ± 0.4, (5)	1.2 ± 0.4, (8)
Estimated ET (mm/d)	22.0 ± 6.4, (6)	22.0 ± 4.6, (5)	25.0 ± 3.0, (8)
NH_4^+-N loading rate (g/m²/d)	1.17 ± 0.33, (6)	1.09 ± 0.09, (5)	0.55 ±0.03, (8)

Mean value ± SE are provided.

Additional information: Evapotranspiration rates of the V-H hybrid SSF CWs were estimated using the inflow and outflow rates together with precipitation data (Appendices 8 & 9) according to the expression below as recommended by Headley et al. (2012).

$$ET= (Q_{in}+P-Q_o) \quad \text{(viii)}$$

Where: ET=evapotranspiration (mm/d), Q_{in}= inflow rate (mm/d), P=precipitation (mm/d) and Q_o=outflow rate (mm/d).

Table 3-3. Physico-chemical composition of AFP water fed into the V-H SSF hybrid CWs during system initiation and optimization.

Parameter (unit)	System Initiation (Feb-Jul 2015)	System Optimisation (Oct 2015-Jul 2016) HP (Oct 2015-Mar 2016)	LP (Apr -Jul 2016)
Water temp (°C)	18.8 ± 2.3, (6)	21.6 ± 0.7, (5)	17.8 ± 1.2, (8)
pH	7.1 ± 0.1, (6)	7.2 ± 0.1, (5)	7.5 ± 0.1, (8)
DO (mg/L)	5.1 ± 0.9, (2)	2.8 ± 0.3, (5)	3.5 ± 0.3, (8)
EC (mS/m)	124 ± 2.7, (6)	116 ± 2.4, (5)	103 ± 5.5, (8)
NH_4^+-N (mg/L)	21.7 ± 5.8, (6)	30.5 ± 3.5, (5)	14.8 ± 1.0, (8)
NO_3^--N + NO_2^--N (mg/L)	0.3 ± 0.0, (6)	0.3 ± 0.0, (5)	0.2 ± 0.0, (8)
PO_4^{3-}-P (mg/L)	5.1 ± 0.6, (6)	6.8 ± 0.4, (5)	2.9 ± 0.4, (8)
COD (mg/L)	91.7 ± 12.6, (6)	154.5 ± 18.5, (5)	145.2 ± 26.5, (8)
TSS (mg/L)	120.2 ± 23.8, (6)	109 ± 22.3, (5)	70.5 ± 9.7, (8)
FC (CFU/100 mL)	5.3×10^5, (6)	4.1×10^5, (5)	4.1×10^5, (4)

Mean values ± SE are provided. Numbers in the parenthesis represent number of data used to generate the means.

3.2.3 Water sampling and analysis

The V-H SSF hybrid CWs were monitored monthly during the initiation phase and HP However, during LP, the V-H SSF hybrid CWs were monitored biweekly. Sampling points were at the outlet of the buffer tanks (A_1 and A_2) and, outlet of the HSSF CWs (C_1 and C_2) as shown in Figure 3-1. Flow rates into and out of the hybrid CWs were determined at the time of sampling using the volumetric method (Tabolt-Smith, 2013). Parameters measured on-site included pH (HANNA HI 8424 model pH meter, HANNA instruments, Singapore), EC (EC Testr® 11 dual range 68×546,501 m, Eutech Instruments, Singapore), and DO and temperature (DO meter, model: 8602 CE).

Grab samples (500 mL) were collected from sampling points A_1 and A_2 and from C_1 and C_2 (see Figure 3-1) at 8:00, 12:00 and 16:00 for NH_4^+-N, NO_3^--N, NO_2^--N, PO_4^{3-}-P, COD, TSS and FC analysis (*E. coli* was used as the indicator of faecal pollution). Prior to sampling, glass bottles were acid washed and autoclaved for 15 min at 121 °C, rinsed with wastewater from the sampling points, and water samples collected and transferred immediately to the laboratory for analysis.

Chromocult agar (prepared as indicated by manufacturers' instructions) was used for the enumeration of *E. coli* by spread plate analysis of 0.1 mL of the diluted sample. The plates were incubated for 24 h at 37 °C and enumeration was based on colony forming units per 100 mL (CFU/100 mL).

Residual water from each collected sample was filtered immediately for nutrient and COD analysis while TSS was analysed from the unfiltered samples. Both filtered and unfiltered samples were stored at 4 °C if samples were to be analysed the following day. PO_4^{3-}-P, NH_4^+-N, NO_3^--N, NO_2^--N, COD and TSS were be analyzed following standard methods (APHA, 1998) and included analysis of PO_4^{3-}-P using the ammonium molybdate method, NH_4^+-N using the phenol-hypochlorite method, NO_3^--N using the salicylate method and NO_2^--N using the sulphanil acid method. Soluble COD was analyzed using a COD test kit purchased from Merck Chemical Company, Damstadt Germany. TSS was analysed using gravimetric determination of weight of suspended solids retained using Whatman membrane filter discs of pore size 0.45 µm that were pre-oven dried at 103-105 °C overnight. A defined volume of water was filtered and the filter together with the residue dried for 3 h at 95 ± 5 °C until a constant weight was obtained. The TSS concentration in the water sample was then calculated using the expression below.

$$TSS = ((W_i - W_o) \times 10^6)/V \qquad \text{(ix)}$$

Where TSS=Total suspended solids concentration (mg/L), W_o=weight of pre-oven dried filters (g), W_i=weight of filters + residue after drying (g) and V=volume of water sample filtered (mL).

3.2.4 Statistical analysis and data presentation

Data were analysed using Excel (Microsoft Office Version 7, Microsoft, USA), and Sigma Plot Version 8, (Systat Software Inc., USA). Firstly, data was tested for normality; then subjected to t-test and non-parametric tests as required with the level of significance, determined at 0.05.

Influent and effluent concentration data is presented as mean concentration for the two hybrid CWs but separately for the two operational phases. Efficiency of the V-H hybrid SSF CWs was computed based on mass balance calculations according to the expression below.

$$E = ((L_{in} - L_{out})/Lin) \times 100 \qquad \text{(x)}$$

Where: E is the removal efficiency expressed as a percentage (%), L_{in} and L_{out} are the influent and effluent loading rate ($g/m^2/d$); computed according to the equation below.

$$L = (C \times Q)/A \qquad \text{(xi)}$$

Where: L=mean loading rate into or out the V-H SSF hybrid CW (g/m^2/d), C=mean pollutant concentration (mg/L i.e. 1 mg/L=1 g/m^3), Q=mean flow rate into or out CWs (m^3/d) during a given operational phase and A=surface area of the V-H SSF hybrid CW (m^2).

3.3 Results

This section presents results of the performance of the V-H SSF hybrid CWs during the monitoring period (i.e. from February 2015 to July 2016). The results are presented in two phases; Phase 1 and phase 2. Phase 1 deals exclusively with the initiation phase whereas phase 2 is devoted to the optimization phase.

3.3.1 Phase 1: Overall performance of the hybrid CWs during the initiation phase

Initiation of the V-H SSF hybrid CW system was an exploratory phase that was carried out from February-July 2015 whose aim was to determine the optimal operating conditions so that performance can be determined accurately.

Table 3-4 presents a summary of the mean physico-chemical characteristic of the AFP water quality after treatment by the V-H SSF hybrid CWs and overall efficiency of the system (presented as mass removal rate) during the initiation phase while Appendix 9 provides performance details of the monthly data during the study period.

Table 3-4. Performance summary for the V-H SSF hybrid CWs during the initiation phase compared with the DWS (2013) discharge limit.

Parameter (units)	AFP effluent	hybrid CWs effluent	Removal rate (%)	DWS (2013) discharge limits
Water temperature (°C)	18.8 ± 2.3, (6)	17.5 ± 2.1, (6)	N/A	N/P
pH	7.1 ± 0.1, (6)	7.0 ± 0.1, (6)	N/A	5.5-9.5
DO (mg/L)	5.1 ± 0.1, (2)	2.6 ± 0.3, (2)	N/A	>2
EC (mS/m)	124 ± 2.7, (6)	111 ± 4.2, (6)	N/A	70 mS/m above intake to A maximum of 150 mS/m
NH_4^+-N (mg/L)	21.7 ± 5.8, (6)	**11.9 ± 2.7, (6)**	62.4	6
NO_3^--N + NO_2^--N (mg/L)	0.3 ± 0.0, (6)	0.3 ± 0.1, (6)	50.0	15
COD (mg/L)	91.7 ± 12.6, (6)	55 ± 10.8, (6)	60.0	75
TSS (mg/L)	120.1 ± 23.8, (6)	14.0 ± 2.6, (6)	93.5	25
PO_4^{3-}-P (mg/L)	5.1 ± 0.6, (6)	7.1 ± 1.2, (6)	7.1	10
FC (CFU/100 mL)	5.3×10^5, (6)	**1500 ± 550, (6)**	99.8	1000

Mean ± SE and number of data used to calculate the means (values in the parentheses) are provided. Bold values do not meet the DWS (2013) general authorization limit for discharge into a surface water resource. N/A means not applicable.

As shown in Table 3-4 above, V-H SSF hybrid CWs achieved high removal rate for all the monitored parameters except PO_4^{3-}-P. All the monitored parameters met the DWS (2013) authorization for discharge of the treated wastewater into a surface water resource except NH_4^+-N and FC.

The performance of CWs is influenced by various physico-chemical, operational and hydrological factors. Table 3-5 presents a summary of the results from regression analysis of the factors that influenced removal of NH_4^+-N and FC from the V-H SSF hybrid CWs. As illustrated in Table 3-5, the major factor that greatly influenced NH_4^+-N removal from the V-H SSF hybrid CWs was influent NH_4^+-N loading rate.

Table 3-5. Summary of regression analysis for the various parameters that influenced NH_4^+-N and FC removal from the V-H SSF hybrid CWs.

	NH_4^+-N
Parameter	
Influent NH_4^+-N LR	y=0.74x-0.14, R^2=0.93, sig F=0.00
HLR	y=0.01x + 0.13, R^2=0.03, sig F= 0.76
H_2O temperature	y= -0.07x + 1.99, R^2=0.38, sig F=0.19
ET	y=0.01x + 0.87, R^2=0.03, sig F=0.76
	FC
Influent NH_4^+-N LR	y=1.00x-11.21, R^2=0.99, sig F=0.00
HLR	y=585.01-3010.52, R^2=0.16, sig F=0.43
H_2O temperature	y=-768.6 x + 43979.8, R^2=0.10, sig F=0.54
ET	y=-92.35x + 31577.58, R^2=0.01, sig F=0.84

Number of data used to generate the equation=6; LR means loading rate.

3.3.2 Phase 2: Overall performance of the hybrid CWs during the optimization phase

The optimization phase aimed at investigating the effect increasing HRT and varying influent NH_4^+-N loading rate on the of performance of the V-H SSF hybrid CWs.

Thus, Table 3-6 presents a summary of the mean physico-chemical characteristic of the AFP water quality after treatment by the V-H SSF hybrid CWs and overall removal rate of the systems compared with the DWS (2013) discharge limit during the optimisation phase (i.e. HP and LP). Details of the monthly performance data during the two operational phases is provided in Appendix 9.

According to Table 3-6, optimization of the operational parameters, yielded interesting results. Initially, reducing only the HLR from 56.0 ± 3.8 mm/d to 39.0 ± 1.2 mm/d while increasing the HRT from ~2 d during the initiation phase to ~3 d during HP (Table 3-2) generally improved the performance of the V-H SSF hybrid CWs respectively.

Table 3-6. Performance summary for the V-H SSF hybrid CWs during HP and LP compared with the DWS (2013) discharge limits.

Parameters /details	AFP effluent	hybrid CW effluent	Removal rate (%)	DWS (2013) effluent discharge limits
Water temperature				N/P
HP	21.6 ± 0.7, (5)	21.5 ± 1.3, (5)	N/A	
LP	17.8 ± 1.2, (8)	16.8 ± 1.1, (8)	N/A	
DO				>2
HP	2.8 ± 0.3, (5)	2.2 ± 0.2, (5)	N/A	
LP	3.5 ± 0.3, (8)	2.9 ± 0.2 (8)	N/A	
pH				5.5-9.5
HP	7.2 ± 0.1, (5)	7.0 ± 0.1, (5)	N/A	
LP	7.5 ± 0.1, (8)	7.3 ± 0.0, (8)	N/A	
EC				70 mS/m above intake to a maximum of 150 mS/m
HP	116 ± 2.4, (5)	122 ± 5.6, (5)	N/A	
LP	103 ± 5.5, (8)	122 ± 7.0, (8)	N/A	
NH_4^+-N				
HP	30.5 ± 3.5, (5)	**15.9 ± 4.4, (5)**	72.5	6
LP	14.8 ± 1.0, (8)	3.0 ± 0.7, (8)	92.7	
NO_3^--N+ NO_2^--N				
HP	0.3 ± 0.0, (5)	0.2 ± 0.0, (4)	100	15
LP	0.2 ± 0.0, (8)	0.2 ± 0.0, (8)	100	
COD				
HP	154.2 ± 18.5, (5)	**105 ± 7.5, (5)**	66.8	75
LP	145.2 ± 26.5, (8)	**125 ± 21.0, (8)**	71.1	
TSS				
HP	109 ± 22.3, (5)	7.0 ± 1.0, (5)	97.0	25
LP	70.5 ± 9.7, (8)	6.0 ± 0.9, (8)	97.3	
PO_4^{3-}-P				
HP	6.8 ± 0.4, (5)	9.2 ± 1.6, (5)	56.4	10
LP	2.9 ± 0.4, (8)	2.8 ± 0.4, (7)	63.6	
FC				
HP	4.1×10^5, (5)	0 ± 0.0, (6)	100	1000
LP	**4.1** $\times 10^5$, (8)	**1500, (4)**	99.8	

Mean ± SE and number of data used to compute the means (values in parentheses) are provided. All parameters are in mg/L except water temperature in °C, EC in mS/m and faecal coliforms in CFU/100 mL. Bold values do not meet the DWS effluent discharge standards. N/A and N/P mean not applicable and not provided respectively.

Thus, higher removal rates for all water quality parameters were recorded at longer HRT during HP (Table 3-6) than at shorter HRT during the initiation phase (Table 3-4). During the HP FC in the treated water significantly decreased hence meeting the DWS (2013) effluent discharge limit together with NO_3^--N + NO_2^--N, TSS, PO_4^{3-}-P. Unexpectedly, NH_4^+-N and COD concentrations failed to meet the DWS (2013) effluent discharge limit of 6 and 75 mg/L

Nevertheless, diluting the AFP effluent to achieve a mean influent NH_4^+-N concentration of 14.8 mg/L and maintaining the HLR of 37.0 ± 0.3 mm/d during the LP resulted in a more reduced mean influent NH_4^+-N loading rate than that recorded either during the initiation phase or HP (Table 3-2). Therefore, operating the V-H SSF hybrid CWs at lower NH_4^+-N loading rate while maintaining the HRT ~3 d during the LP than either the initiation phase or the HP further improved the performance of the V-H SSF hybrid CWs. The removal rates for all parameters further increased and were higher than those recorded during HP except for FC, which slightly reduced, achieving similar removal rate like during the initiation phase. During LP, all parameters met the effluent discharge limits for the DWS (2013) except COD and FC (Table 3-6).

Table 3-7 provided a summary of the results from the regression analysis about the factors that influenced the performance of the V-H SSF hybrid CWs with specific attention to NH_4^+-N, COD and FC.

Table 3-7. Summary of regression analysis for various parameters that influenced removal of NH_4^+-N, COD and FC from the V-H SSF hybrid CWs.

Operational phase	**HP**	**LP**
Parameter	**NH_4^+-N**	
Influent NH_4^+-N LR	$y=0.07x + 0.71$, $R^2=0.01$, sig F=0.86	$y=0.79x + 0.08$, $R^2=0.98$, sig F=0.02
HLR	$y=0.04x + 0.57$, $R^2=0.52$, sig F= 0.17	$y=0.03x – 0.45$, $R^2=0.87$, sig F=0.07
H_2O temperature	$y= 0.04x + 0.01$, $R^2=0.21$, sig F=0.0.44	$y=-0.01x +0.72$, $R^2=0.86$, sig F=0.07
ET	$y=0.01x + 0.59$, $R^2=0.53$, sig F=0.16	$y=-0.01x + 0.71$, $R^2=0.86$, sig F=0.07
	COD	
Influent NH_4^+-N LR	$y=1.27x -3.40$, $R^2=0.94$, sig F=0.00	$y=0.92x -1.21$, $R^2=0.97$, sig F=0.02
HLR	$y=0.57x – 18.46$, $R^2=0.48$, sig F=0.20	$y=-0.85x + 35.77$, $R^2=0.36$, sig F=0.40
H_2O temperature	$y=0.73x – 11.91$, $R^2=0.29$, sig F=0.35	$y=0.46x - 4.38$, $R^2=0.52$, sig F=0.28
ET	$y=0.12x + 1.17$, $R^2=0.30$, sig F=0.34	$y=0.34x + 4.88$, $R^2=0.68$, sig F=0.18
	FC	
Influent NH_4^+-N LR	$y=1.00x – 4.96$, $R^2=1$, sig F=0.00	$y=1.00x + 49.23$, $R^2=1$, sig F=0.00
HLR	$y=1552.71x – 44213.80$, $R^2=0.21$, sig F=0.43	$y=4425.19x-149077$, $R^2=0.83$, sig F=0.09
H_2O temperature	$y=3646.23x – 62727.30$, $R^2=0.44$, sig F=0.22	$y=-2118.91x + 55742.97$, $R^2=0.93$, sig F=0.03
ET	$y=117.37x + 13472.65$, $R^2=0.02$, sig F=0.83	$y=-1401.11x + 53946.87$, $R^2=0.98$, sig F=0.01

Results show that generally, the influent NH_4^+-N loading rate, HLR, water temperature and ET strongly influenced NH_4^+-N, COD and FC removal rate during HP than LP.

3.4 Summary

The summary of findings from the treatment performance of the V-H SSF hybrid CWs are as follows:

- During the initiation phase, the V-H SSF hybrid CWs achieved high removal rates for NH_4^+-N (62.4 %), NO_3^--N + NO_2^--N (50 %), COD (60 %), TSS (93.5) % and FC (99.8 %) but low removal rate for PO_4^{3-}-P (7.1 %). Evaluation of the effluent quality revealed that all the monitored water quality parameters met the DWS (2013) effluent discharge limit except NH_4^+-N and faecal coliforms.
- The optimization phase revealed that:
  (i) Increasing the HRT from ~2 d to ~3 d during HP only improved the removal rate of the V-H SSF hybrid CWs. Thus, higher removal rates for all water quality parameters were recorded at longer HRT during HP i.e. NH_4^+-N (72.5 %), NO_3^--N + NO_2^--N (100 %), COD (66.8 %), TSS (97 %), PO_4^{3-}-P (56.4 %) and faecal coliforms (100 %) than at shorter HRT during the initiation phase. However, despite the increase in removal rate, all the other parameters met the DWS (2013) effluent discharge limit except NH_4^+-N and COD.
  (ii) Maintaining the HRT of the V-H SSF hybrid CWs at HRT ~3 d while reducing the influent NH_4^+-N loading rate from 1.09 g/m^2/d during HP to 0.55 g/m^2/d during the LP further improved both removal rate and effluent quality of the monitored parameters. Thus, all parameters met the effluent discharge limits for the DWS (2013) except COD and FC.

Chapter 4 Use of discard coal as a substrate for constructed wetlands: a conceptual study

4.1 Introduction

Presently, the coal mining industry is the most important driver of the South Africa's economy with annual production of 250 million tonnes, over 70 % of which is utilized locally, mostly for electricity and synthetic fuels production. Beneficiated coal is also exported (61 million tonnes) which generates a large foreign income exchange for South Africa (North et al., 2015). However, the export market demands coal of a high calorific value, thus to achieve this requirement, raw coals must be treated/washed resulting into generation of 3 different coal wastes viz: duff, discard and slurry coal. Whereas the middle calorific quality is being used for internal power and heat production by power generation companies such as ESKOM (LHV~18-30MJ/kg), lower calorific coals have been dumped as discard coals although this is changing. Nowadays, washing and sorting have been optimized to higher efficiencies, so that the resulting discard to be dumped is of calorific values often lower than ~5-6 MJ kg^{-1}. Even so, as at 2001, discard coal and slurry were being produced at annual rates of 42.5 million tonnes and 11.2 million tonnes respectively while Swanepoel (2008) reported an annual production rate of 16 million tonnes for duff coal. Thus, annual discard production in South Africa increased from 43.6 million tonnes in 1985 to 66.2 million tonnes by 2001 (National Inventory Discard and Duff coal, 2001).

The total area covered by discard and slurry disposal amounts to 40,011 ha and most is in the Witbank coalfields. The largest at 394 ha, is a mined out opencast area. If left unattended, coal discard dumps and slurry ponds are major contributors to atmospheric pollution, contamination of surface and ground water by acid leachate run off, erosion and sedimentation of particulates into adjacent rivers and dams, spontaneous combustion, and landslides (Truter et al., 2009). To solve the problems arising from waste coal piles, several bioremediation alternatives are in place with, dry land bioremediation/phytoremediation being the most established approach. In South Africa for instance, waste coal piles are limed, covered with 30-100 cm layer of topsoil, fertilizers are applied, and the topsoil seeded with grass species as a biocatalyst (Rethma & Tanner 1995; Cowan et al., 2016). The drawbacks of this technology include: 1) in cases, where the topsoil is not locally available, it must be sourced and transported to the site under rehabilitation; and, 2) from the area where the top soil is extracted, there is likely to be heavy environmental damage including erosion and sedimentation of particulate solids into the water ways. Even after successful plant establishment, this approach does not necessarily result in breakdown of the underlying waste coal and indeed, may only mask and delay future environmental catastrophes (Cowan et al., 2016). Nevertheless, remediation of such large quantities of solid waste to mitigate environmental impacts should occur *in situ*, be passive, and if possible, lead to formation of carbon rich soil-like material. As stated by Sourkova et al. (2005), rehabilitation of opencast spoil and discard dumps should focus on two aspects: 1) transformation of the carbonaceous waste to a soil-like material through abiotic and biotic weathering, and 2) successful vegetation cladding of the discard dump.

One approach with potential to mitigate environmental degradation resulting from stockpiled discard is to consider discard dumps as non-vegetated constructed wetland (CWs) requiring only water, nutrients, and a suitable macrophyte as biocatalyst. As opposed to dryland bioremediation, treating waste coal dumps as CWs would appear to be a more economical and environmentally sound option as no soil excavation and transportation of topsoil is necessary. Additionally, replacing gravel with discard coal as treatment media would be important in reducing the capital costs associated with SSF CWs.

Thus, this chapter describes an investigation carried to determine the potential of using discard coal as an alternative treatment media in CWs. The intention was to: 1) ascertain whether discard coal supports growth and proliferation of the wetland macrophyte *P. australis*; 2) ascertain whether coal is decomposed within the CWs; and, 3) examine the quality of the treated water from CWs. Results are discussed in terms of the potential of discard coal-containing CWs to mitigate pollution and the suitability of discard coal as a treatment media for use in CWs for domestic wastewater treatment.

4.2 Materials and methods

4.2.1 Experimental set-up

To determine whether discard coal can be used as treatment media in CWs, laboratory-scale HSSF CWs were established by packing containers (0.4 × 0.3 × 0.25) either with coal discard or gravel of particle size ~14 mm to a depth of 0.2 m. The summary of proximate and selected elemental composition of discard coal that was used as treatment media is shown in Table 4.1. The small-scale HSSF CWs were positioned outdoors and maintained under ambient conditions at the Belmont Valley WWTW, Grahamstown, South Africa (33° 19’ 07” South, 26° 33’ 25” East). At the beginning of November 2015, two gravel- and two discard coal-containing HSSF CWs were each planted with four similar sized shoots of *P. australis* obtained from the banks of a nearby river and immediately irrigated with water. Two types of influent water were used: 1) nutrient poor tap water and, 2) nutrient rich water from the advanced facultative pond (AFP). Table 4.2 gives a summary of the physico-chemical properties of each influent water type used during the study period.

Table 4-1. Composition of discard coal used in the laboratory scale HSSF CWs.

	Weight (%)	
Element	**Gravel**	**Discard coal**
Proximate composition:		
Water content	-	1.6 ± 0.2, (4)
Ash content	-	23.5 ± 0.1, (4)
Volatile matter	-	19.7 ± 2.5 (4)
Fixed carbon	-	55.1
Elemental composition:		
Al	2.62 ± 0.19, (3)	2.3 ± 1.5, (3)
C	5.60 ± 0.62, (3)	68.8 ± 5.2, (3)
Cl	-	0.14 ± 0.0 (3)
Fe	-	1.61 ± 0.9, (3)
H	-	2.7 ± 0.1, (3)
Mg	-	0.36 ± 0.2, (3)
N	-	1.6 ± 0.2, (3)
O	59.14 ± 0.32, (3)	39.04 ± 1.6, (3)
P	-	3.72 ± 1.7, (3)
S	-	0.8 ± 0.4, (3)

Mean values ± SE are provided. The numbers in the parentheses represent number of data n, used to generate the means.

Reservoirs (20 L) were positioned to continuously feed either tap water or AFP water at a rate of 43 L/d to both gravel- and discard coal-containing HSSF CWs. Water within the HSSF CWs was maintained 0.15 m below the surface of the treatment media using an outlet pipe (Figure 4.1).

Table 4-2. Physico-chemical characteristics of water fed into the gravel- and discard coal-containing HSSF CWs.

Parameter	Tap water	AFP water
pH	7.2 ± 0.1, (5)	7.2 ± 0.1, (5)
DO (mg/L)	4.7 ± 0.4, (5)	3.8 ± 0.5, (5)
EC (mS/m)	52.3 ± 13.4, (4)	134.3 ± 4.6, (4)
NH_4^+-N (mg/L)	1.3 ± 0.4, (4)	28.8 ± 6.0, (5)
PO_4^{3-}-P (mg/L)	0.4 ± 0.1, (5)	6.7 ± 0.6, (5)
SO_4^{2-} (mg/L)	9.3 ± 3.3, (4)	30.7 ± 5.7, (4)
Cl (mg/L)	101.0 ± 30.6, (5)	228.0 ± 9.8, (4)
Mg (mg/L)	10.6 ± 3.3, (5)	19.4 ± 0.6, (5)
Fe (mg/L)	0.1 ± 0.0, (5)	0.2 ± 0.0, (5)
Al (mg/L)	0.0, (5)	0.0, (5)

Mean values ± SE are provided. The numbers in the parentheses represent number of data n, used to generate the means.

Figure 4-1. Image of the experimental set-up that was used in the study from right to left: discard coal + tap water, discard coal + AFP water, gravel + tap water and gravel + AFP effluent.

4.2.2 Sampling and laboratory analysis

This commenced immediately after planting in November 2015 and involved monitoring plant growth and water quality parameters before and after treatment through the CWs.

4.2.2.1 Plant growth assessment

Plant growth was assessed by determining: (1) the shoot density and (2) the biomass. Shoot density was determined by counting the number of shoots in each experimental set-up at the end of every month from November 2015 to July 2016. The monthly shoot density was then used to estimate the shoot density per m^2, which was calculated as the ratio of shoot density at the end of every month to the area of the HSSF CW.

Plant biomass was determined at the end of July. All the above ground plant material from each experimental CW was harvested, sun dried (1 week) prior to oven drying at 60 °C until a constant weight was obtained after which the mass determined and expressed as dry weight. Plant biomass was then estimated as ratio of plant dry weight to area of the HSSF CW.

Chlorophyll fluorescence was used to assess the health status of *P. australis* during the experimental period using a plant efficiency analyser (Hansatech Model, Hansatech instruments Ltd, The United Kingdom and was carried out monthly from May-July 2016. Chlorophyll fluorescence, presented as Fv/Fm, was analysed following dark phase adaptation (30 sec) of 10 different leaves per measurement.

4.2.2.2 Proximate analysis and ultimate analysis

Proximate analysis included determination of water content and ash content, volatile matter and fixed carbon carried out on discard coal samples before and after treatment in the CWs (end of the experimental period) and these are expressed as weight percentage (wt %). For each of the parameters, duplicate initial samples (1 g) were considered. Moisture content was determined after exposure of discard coal and gravel samples (1 g) to 105 °C for 1 h and the relative loss of mass reported as percentage moisture. Ash content was determined after combustion of coal discard in a muffle furnace (Gallenkamp Model, Gallenkamp Muffle Furnace Co., London) at 815 °C. Firstly, the samples were heated by increasing the temperature to 400 °C for over 30 min, then to 815 °C for a further 30 min, followed isothermally for 2 h and the residue, reported as percentage ash. Volatile matter of coal discard was quantified after heating to 910 °C for 7 min and the weight loss reported as percentage volatile matter.

The fixed carbon content of coal discard was calculated according to the expression below:

$$\% \text{ fixed carbon} = 100 - (\% \text{ moisture} + \% \text{ volatile matter} + \% \text{ ash}) \qquad \text{equation (xii)}$$

Following manufacturers' instructions, elemental analysis of discard coal was carried out using an elemental analyser (Elementar vario microcube, Elementar UK Ltd).

4.2.2.3 Water quality sampling and analysis

Monitoring of the HSSF CWs for water quality parameter commenced in March 2016 and were monitored monthly until July 2016. Sampling points were located at the outflow of the reservoirs and the HSSF CWs. Parameters measured on-site included pH, DO, temperature and EC measured using instruments previously described in chapter 3. Chemical parameters monitored included NH_4^+-N, PO_4^{3-}-P, Cl, Fe and Al as per the general standards (Republic of South Africa Government Gazette No. 36820, 2013). NO_3^--N and NO_2^--N were not analysed since their concentration in domestic wastewater effluent from HSSF CWs is very low (< 5 mg/L) (Chapter 1, Chapter 3, Mburu et al., 2013). However, SO_4^{2-} and Mg were also included due to their known effects on the environmental (WHO, 2003; Strigul et al., 2005) despite not being included in the DWS effluent guidelines.

For analysis of chemical parameters, duplicate grab water samples (500 mL) were collected at monthly intervals before and after treatment. One set of samples was used for analysis of NH_4^+-N and PO_4^{3-}-P using standard methods previously described in Chapter 3. The remaining samples were placed in a cool box and transported to the ISO 17025 accredited laboratories of BEMLAB pty, Strand, South Africa within 48 hours for analysis of Al, SO_4^{2-}, Fe, Mg and Cl.

4.2.3 Statistical analysis and data presentation.

The data were analysed using Excel (Microsoft Office Version 7, Microsoft, USA), and Sigma Plot Version 8, (Systat Software Inc., USA). One-way ANOVA was used to determine the difference in the mean values between the different treatments; with the level of significance determined at 0.05.

All the monitored parameters are presented as mean values over a given monitoring period but separately for the 4 experimental set-ups.

4.3 Results

4.3.1 Plant growth assessment

Figure 4-2 and Table 4-3 show the minimum, maximum and mean *P. australis* shoot density recruitment on discard coal and gravel-containing HSSF CWs respectively, that were fed with tap water and advanced facultative pond (AFP) effluent.

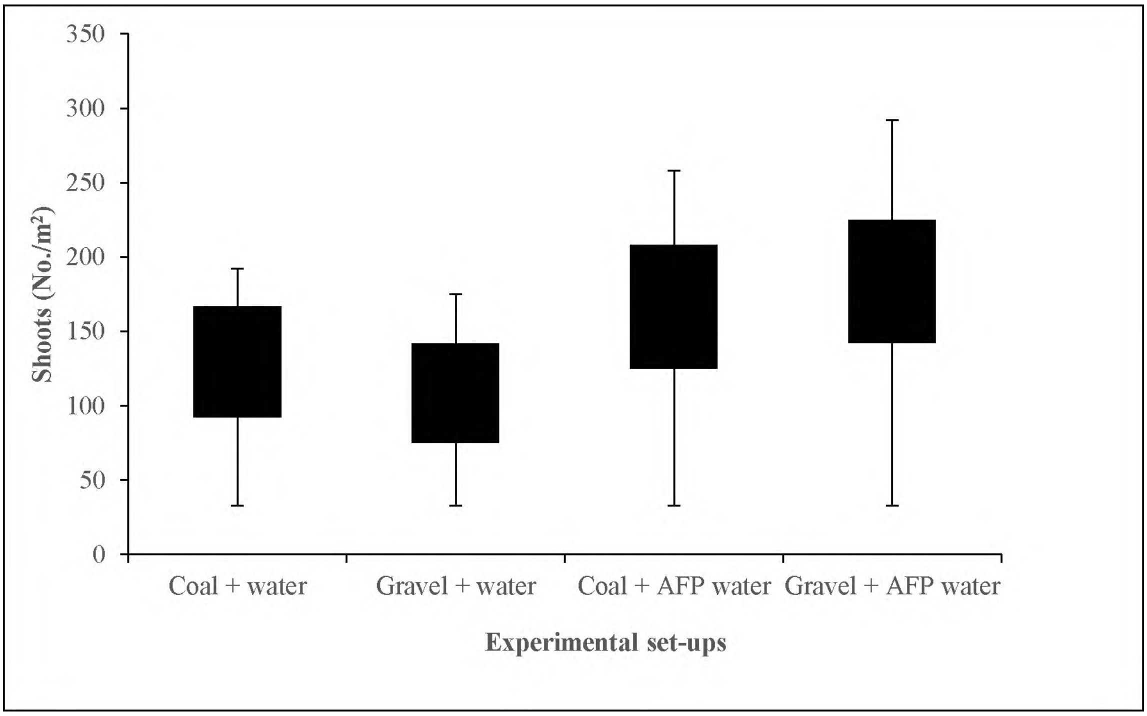

Figure 4-2. Recruitment of *P. australis* shoot discard in coal- and gravel-containing HSSF CWs fed with tap and AFP water.

Table 4-3. Recruitment of *P. australis* shoots on discard coal- and gravel-containing HSSF CWs fed with tap water and AFP effluent over a 9 months monitoring period.

Experimental set-up	Coal + H_2O	Gravel + H_2O	Coal + AFP H_2O	Gravel + AFP H_2O
Shoot density (No./m^2)	124 ± 19.8 (9)	109 ± 17.3, (9)	158 ± 25.6 (9)	178 ± 28.9, (9)

Mean ± SE and number of data, n used to calculate the means (in the parentheses) are represented.

Generally, *P. australis* in discard coal and gravel-containing HSSF CWs fed with tap water revealed lower shoot density than coal and gravel CWs fed with AFP effluent. In discard coal and gravel-containing HSSF CWs fed with tap water, the shoot density of *P. australis* increased from 33 shoots/m^2 in November 2015 to a maximum of 192 shoots/m^2 and 175 shoots/m^2 respectively (Figure 4-3). Although the mean shoot density in the coal based HSSF CW (124 ± 19.8 shoots/m^2) was greater than that of the gravel based HSSF CW (109 ± 17.3 shoots/m^2) (Table 4-3), ANOVA test showed that these values were not significantly different (p=0.58). However, in the AFP water fed coal and gravel HSSF CWs, the shoots density increased from 33 shoots/m^2 to 258 shoots/m^2 and 292 shoots/m^2 in the coal and gravel-containing HSSF CWs respectively by the end of the experimental period (Figure 4-3). The mean shoot density was 158 ± 25.6 shoots/m^2 and 178 ± 28.9 shoots/m^2 in the coal and gravel containing HSSF CWs respectively (Table 4-3). Even though this time, the gravel-containing HSSF CW registered higher mean shoot density (178 ± 28.9 shoots/m^2) than the coal-containing HSSF CW (158 ± 25.6 shoots/m^2), these values are not significantly difference ($p=0.62$).

Table 4-4 gives a summary of the above ground plant biomass that was estimated from the four experimental set-ups. In tap water fed HSSF CWs, *P. australis* on gravel containing HSSF CW produced less above ground biomass (1080 g/m^2) than *P. australis* on coal HSSF CW. Nonetheless, despite the lower plant shoot density in the coal than gravel HSSF CW fed with AFP water (Figure 4-3, Table 4-3), the coal HSSF CW recorded more biomass (7580 g/m^2) than the gravel containing HSSF CW (3670 g/m^2).

Table 4-4. *P. australis* biomass harvested from waste coal- and gravel-containing HSSF CWs at the end of August 2015.

Experimental set-up	Coal + H_2O	Gravel + H_2O	Coal + AFP effluent	Gravel + AFP effluent
Plant Biomass (g/m^2)	1500	1080	7580	3670

Chlorophyll fluorescence was used to assess the health status of *P. australis* plants and Figure 4-3 shows the mean Fv/Fm values of *P. australis* in the four treatments. In tap water fed coal and gravel-containing HSSF CWs fed, chlorophyll fluorescence of *P. australis* leaves was 0.68 ± 0.03 and 0.72 ± 0.02 respectively while in AFP water fed coal and gravel-containing HSSF CWs, mean values of 0.75 ± 0.01 and 0.72 ± 0.01 respectively were recorded. Considering treatments receiving the same influent feed, the ANOVA test showed that these values are not significantly different (coal and gravel HSSF CWs fed with tap water: $p=0.20$; coal and gravel fed with AFP water, $p=0.18$).

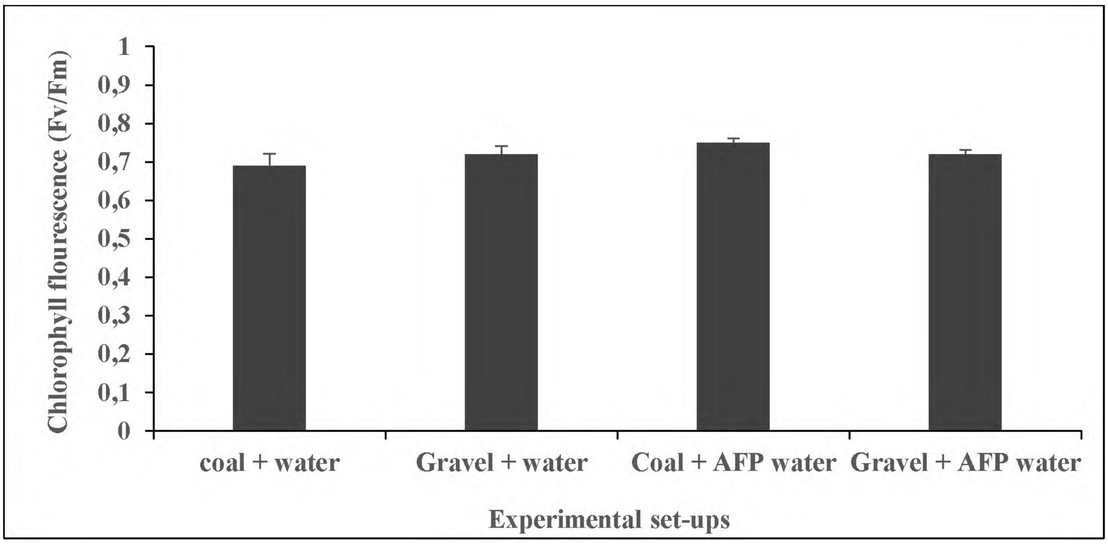

Figure 4-3. Mean ± SE Fv/Fm values for leaves of *P. australis* from coal- and gravel-containing HSSF CWs fed with tap and AFP water.

4.3.2 Proximate and ultimate analysis data

Results obtained from proximate and ultimate analysis of discard coal at the end of the experimental period are presented in Table 4-5.

Table 4-5. Proximate and ultimate analysis of discard coal composition before the start and end of the experimental study period.

Parameter	Discard coal at the start (wt %)	Discard-coal at the end	
		Discard coal + tap water (wt %)	Discard coal + AFP water (wt %)
Proximate analysis:			
Water content	1.6 ± 0.2, (4)	0.5 ± 0.0, (4)	0.5 ± 0.2, (4)
Ash content	23.5 ± 0.1, (4)	29.6 ± 0.5, (4)	27.7 ± 0.4, (4)
Volatile matter	19.7 ± 2.5, (4)	27.7 ± 0.1, (4)	25.7 ± 1.6, (4)
Fixed carbon	55.1	42.2	46.1
Elemental analysis:			
C	68.8 ± 5.2, (3)	32.8 ± 2.3, (3)	61.0 ± 5.9, (2)
H	2.7 ± 0.1, (3)	1.0 ± 0.5, (3)	2.7 ± 0.5, (2)
N	1.6 ± 0.2, (3)	0.7 ± 0.1, (3)	1.4 ± 0.2, (2)
S	0.8 ± 0.4, (3)	0.6 ± 0.1, (3)	0.5 ± 0.1, (2)

Mean ± SE and number data n, used to generate the means (values in parentheses) are provided

Proximate analysis showed that discard coal before the start of the experiment contained higher water content and fixed carbon (1.6 ± 0.2 % and 55.1 % respectively) than either discard coal from a HSSF CW fed with tap water (water content=0.5 ± 0.0 %; fixed carbon=42.2 %) or AFP water (water content=0.5 ± 0.2 %; fixed carbon=46.1 %) at the end of the experimental period (Tables 4-1 & 4-5). The ANOVA test showed that these values were significantly different (discard coal before vs discard coal from tap water fed HSSF CW at the end of the experiment: p=0.00; discard coal before vs discard coal from AFP water fed HSSF CW at the end of the experiment: p=0.00).

However, discard coal before the start of the experiment recorded lower ash content (23.5 ± 0.1 %) and volatile matter (19.7 ± 20.5 %) than discard coal at the end of the experiment from either tap water fed HSSF CW (ash content=29.6 ± 0.5 %; volatile matter=27.7 ± 0.1 %) or AFP water fed CW (ash content=27.7 ± 0.4 %; volatile matter=25.7 ± 1.6 %) (Tables 4-1 & 4-5). The ANOVA test showed that the ash content recorded from discard coal from tap and AFP water fed HSSF CWs at the end of the experiment were significantly higher than that recorded from discard coal before the start of the experiment ($p=0.00$). The ANOVA test showed that while the volatile matter from discard coal before the experiment was significantly lower than that from tap water fed HSSF CW at the end of the experiment ($p=0.024$), it was however not significantly different from discard coal from AFP water fed CW ($p=0.087$). The volatile matter from discard coal from tap and AFP water fed HSSF CWs were also not significantly different ($p=0.331$).

Ultimate analysis however showed that discard coal before the start of the experiment contained slightly higher elemental composition than discard coal at the end of the experiment from either tap or AFP water fed HSSF CWs (Tables 4-1 & 4-5).

4.3.3 Effluent quality from constructed wetlands

Table 4-6 provides a summary of the effluent water quality obtained from the HSSF CWs fed with tap water and AFP effluent. The DWS (2013) guidelines for discharge of effluent into the environment does not provide effluent discharge standards for some water quality parameters e.g. Cl^-, SO_4^{2-}, mg and Al.

However, in gravel and discard coal- containing HSSF CWs fed with tap water, all the other monitored water quality parameters (i.e. pH, EC, DO, NH_4^+-N, PO_4^{3-}-P and Fe) met the DWS (2013) effluent discharge limits. On the other hand, in gravel and discard coal- containing HSSF CWs fed with AFP effluent, all parameters (i.e. pH, EC, DO, PO_4^{3-}-P and Fe) met the DWS (2013) effluent discharge standards except NH_4^+-N. Nevertheless, as shown in Table 4-6 below, discard- containing HSSF CW fed with AFP effluent did much better on NH_4^+-N removal (28.8 mg/L to 7.9 mg/L) than gravel- containing HSSF CW (28.8 mg/L to 13.5 mg/L).

Table 4-6. Physico-chemical composition of tap and AFP water before and after treatment through the discard coal- and gravel-containing HSSF CWs.

Parameter (unit)	Tap water before treatment	Tap water after treatment		AFP water before treatment	AFP water after treatment		DWS (2013) discharge limit
		Discard coal + H_2O	Gravel + H_2O		Discard coal + AFP	Gravel + AFP	
pH	7.2 ± 0.1, (5)	7.2 ± 0.2, (5)	7.1 ± 0.1, (5)	7.2 ± 0.1 (5)	6.0 ± 0.1, (5)	6.9 ± 0.0, (5)	5.5-9.5
EC (mS/m)	52.3 ± 13.4, (4)	60.0 ± 13.1, (4)	54.3 ± 14.3, (4)	134.3 ± 4.6, (4)	135.5 ± 1.9, (5)	129.2 ± 5.9, (5)	70 above intake to a maximum of
DO (mg/L)	4.7 ± 0.4, (5)	4.2 ± 0.5, (5)	3.0 ± 0.5, (5)	3.8 ± 0.5, (5)	4.0 ± 0.7, (4)	3.3 ± 0.7, (4)	>2
NH_4^+-N (mg/L)	1.3 ± 0.4, (4)	1.3 ± 0.3, (5)	1.3 ± 0.3, (5)	28.8 ± 6.0, (5)	7.9 ± 2.4, (5)	13.5 ± 5.0, (5)	6
PO_4^{3-}-P (mg/L)	0.4 ± 0.1, (5)	0.6 ± 0.2, (5)	0.3 ± 0.0, (5)	6.7 ± 0.6, (5)	5.1 ± 1.1, (5)	5.8 ± 1.0, (5)	10
SO_4^{2-} (mg/L)	9.3 ± 3.3, (4)	12.2 ± 2.7, (5)	14.6 ± 4.7, (5)	30.7 ± 5.7, (5)	31.2 ± 4.2, (4)	24.5 ± 1.2, (4)	-
Cl^- (mg/L)	101.0 ± 30.6, (5)	97.0 ± 29.6, (5)	101.0 ± 27.9, (5)	228.0 ± 9.8, (4)	235.8 ± 6.3, (4)	247.4 ± 21.2, (4)	-
Mg (mg/L)	10.6 ± 3.3, (5)	12.2 ± 2.7, (5)	10.9 ± 3.1, (5)	19.4 ± 0.6, (5)	21.7 ± 0.7, (5)	20.3 ± 0.9, (5)	-
Fe (mg/L)	0.1 ± 0.0, (5)	0.1 ± 0.0, (4)	0.1 ± 0.0, (4)	0.2 ± 0.0, (5)	0.3 ± 0.1, (4)	0.2 ± 0.0, (4)	0.3
Al (mg/L)	0.0 ± 0.0, (5)	0.0 ± 0.0, (5)	0.0 ± 0.0, (5)	0.0 ± 0.0, (5)	0.0 ± 0.0, (5)	0.0 ± 0.0, (5)	-

Mean values ± SE are provided. The numbers in the parentheses represent number of data n, used to compute the means.

The NH_4^+-N concentration recorded after treatment by either gravel or discard coal-containing HSSF CWs fed with AFP effluent was however not significantly different from the 6 mg/L requirement for disposal of treated water from WWTP by the DWS (i.e. gravel- and discard coal-containing HSSF CWs, p=0.20 and 0.49 respectively).

4.4 Summary

This chapter investigated the use of discard coal as an alternative treatment media in CWs. The findings of the study are summarised as follows:

- Regardless of the treatment media, *P. australis* plants established successfully. However, although without a significant difference, better plant growth was recorded from discard coal-containing HSSF CWs fed with AFP effluent than either from gravel- containing CW fed with AFP effluent or gravel and coal HSSF CWs fed with tap water.
- The Fv/Fm value indicates the health status of *P. australis*, thus, the Fv/Fm value is from all the HSSF CWs was not significantly different.
- Proximate analysis of discard coal revealed: i) higher ash content in coal-containing HSSF CW fed with either tap or AFP water at the end of the experimental study than before application into HSSF CWs and ii) higher fixed carbon in discard coal before application in CWs than at the end of the experimental study. Additionally, elemental analysis revealed lower elemental composition from discard coal obtained from HSSF CWs at the end of the experiment than before introduction into the HSSF CWs.
- All HSSF CWs met the DWS (2013) effluent discharge limit for all the monitored water quality parameters except NH_4^+-N from gravel- and discard coal-containing HSSF CW fed with AFP effluent. However, generally, discard coal- did better on NH_4^+-N removal than gravel-containing HSSF CW.

Chapter 5 The use vertical-horizontal hybrid subsurface flow constructed wetland and the high rate algal to supplant the oxidation pond component of a dysfunctional and overloaded waste stabilization pond system

5.1 Introduction

Wastewater 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 communities (Wang et al., 2012). All implemented technologies must be ideally rigorous, ecologically sound and environmentally friendly (Oswald, 1991; Oswald, 1995) and WWTPs should, be able to withstand the overloading and require minimal maintenance over extended periods (Wallis et al., 2008; González et al., 2012). Thus, for a chosen technology to be considered suitable should, over the medium to long term, lower overall costs without sacrificing reliability and efficiency (Katukiza et al., 2012).

As already mentioned in (Chapter 1), municipal wastewater contains various compounds (Appendix 1) that contaminate water bodies. Thus, the Department of Water and Sanitation (DWS) mandates the remediation of all effluent streams prior to discharge (DWA, 2013). Discharge standards ensure that treated effluent from municipal (and industrial) WWTPs is not detrimental and/ or damaging to the receiving environment. In South Africa, low-cost municipal wastewater treatment technologies comprise of mainly waste stabilization ponds (WSP). These treatment works are small to medium (i.e. design capacity either of 500-2000 m^3/d or 2000-10,000 m^3/d) and approximately 70 % are located in urban areas. Unfortunately, the existing infrastructure is ageing and unable to cope with the sheer volumes generated within these urban areas and it is estimated that >50 % of waste water treatment works are either in disrepair, underperform or are overloaded thus, producing an effluent that does not meet national and international wastewater discharge standards (Mthembu et al., 2013). Population growth and migration are major contributors (Showers, 2002; Van Koppen, 2003). Furthermore, due to limited resources, waste management including wastewater treatment has been neglected in favour of other priorities (e.g. health, housing, education) preventing acquisition of new infrastructure and provision of associated municipal services (Wang et al., 2012). Consequently, municipalities in many southern African countries have little choice but to continue to discharge untreated or partially treated wastewater. Despite the fact that WSPs need to be upgraded due to the poor effluent quality generated, WSPs require a relatively large land footprint (Mara, 2003) which is a major limiting factor especially in many urban areas due to high population growth resulting in inadequate land or expansion of WSP systems.

Even in areas where land is readily available, WSPs would not be recommended due to the various disadvantages previously mentioned (Chapter 1). In recent years, two wastewater treatment systems have emerged namely: the V-H SSF hybrid CWs and IAPS. Despite their limited full-scale application, V-H SSF hybrid CWs and IAPS could be a better alternative to WSPs since they both produce a reliable effluent quality and minimise sludge handling (Oswald et al., 1994; Green et al., 1996; Rose et al., 2002; Herrera Melián et al., 2010; Craggs et al., 2014). While a lot of information has been given about V-H SSF hybrid CWs in the previous chapters, IAPS is an adaptation of a conventional WSP system and was developed to streamline wastewater treatment by exploiting the natural interaction between water, air algae and other microorganisms (Oswald 1995; Green et al., 1996; Craggs et al., 2004; Craggs, 2005). In South Africa, the IAPS was introduced about 20 years ago (Rose et al., 2002) and has been the subject of research over the past years. The IAPS treats wastewater with a series of four ponds namely: the advanced facultative pond, (AFP), high rate algal oxidation ponds (HRAOP), algal settling ponds (ASP) and a maturation pond (MP) Figure (5-1).

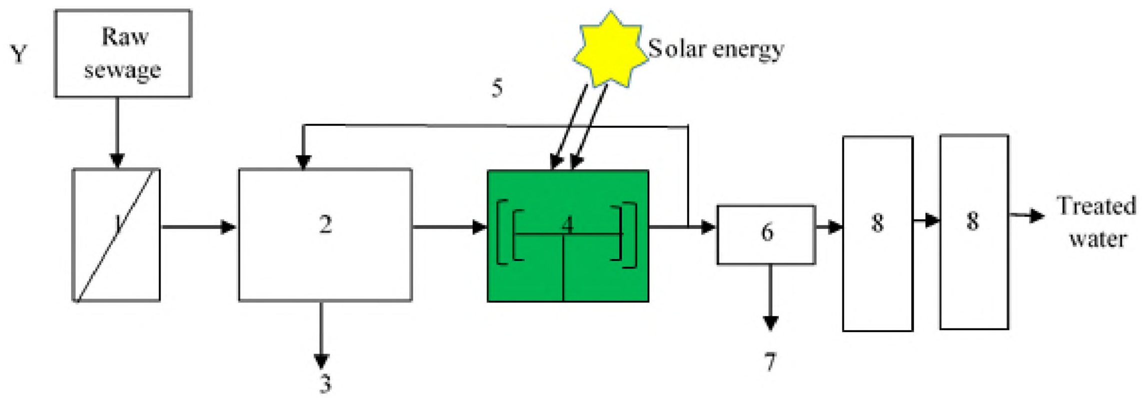

Figure 5-1. Process flow for a full-scale IAPS at St. Helena in California. Key: 1=screening and grit removal chamber, 2=advanced facultative pond (AFP), 3=methane utilization, 4=paddle wheel driven high rate algal oxidation pond (HRAOP), 5=internal circulation, 6=algae settling ponds (ASP), 7=algae utilization and 8=Maturation ponds. Image modified from US. EPA (2011).

Earlier studies reported that the IAPS generates clean water for recycle and reuse, methane-rich biogas, and biomass for high-value horticulture/agriculture and or/or as a renewable source of commodity products (Murphy and Allen, 2011; Craggs et al., 2014). In the IAPS, the HRAOPs are the major driver for reported high quality effluent. Unlike in the WSPs where stratification results in uninform distribution of DO in ponds, studies report that in the HRAOPs, the paddle wheel circulates water to full depth of pond hence, 1) preventing stratification, 2) keeps algae in suspension thus, exposing it to sunlight, which results in high rate algae production. The high rate algae production therefore, results in increased production of DO concentration, which is important DO dependent pollutant removal processes e.g., nitrification and degradation of organic matter (Green et al., 1996; Garcia et al., 2000) thus, generating a high-quality effluent.

Despite the purported high effluent quality from V-H SSF hybrid CW and HRAOPs, to date, there is no information comparing the two systems in terms of the quality of treated water, footprint and capital costs requirement. Thus, the aim of this chapter is to evaluate the use of a V-H SSF hybrid CW and a series of HRAOP to supplant the oxidation pond component of a dysfunctional and overloaded WSP system. The specific objectives included evaluation of the quality of treated water, footprint requirement and cost of implementation of the two systems for treatment of wastewater for the same population equivalent.

5.2 Material and methods

5.2.1 The V-H hybrid CW and integrated algae pond system

The pilot-scale V-H SSF hybrid CW previously described in Chapters 2 & 3 (Figure 5.1) and the demonstration IAPS located at the Belmont Valley wastewater treatment works (BWWTW) (33° 19' 07" South, 26° 33' 25" East) were used as reference for the water quality that is likely to be generated from full scale systems. The IAPS at Belmont Valley was designed by Prof Oswald (RIP) and since it was commissioned in 1996, it is operated and maintained by the Institute for Environmental Biotechnology (EBRU), Rhodes University (Rose et al., 2002). Wastewater for 500 PE (75 m^3/d) mainly of domestic origin is channelled into the demonstration IAPS for treatment. Various studies have been carried out on EBRU IAPS in the past years and details of the design, configuration, and operation of this system is described elsewhere (Rose et al., 2002; Mambo, 2016; Cowan et al., 2016). However, Figure 5-2 and Table 5-1 depicts the process flow and configuration of the IAPS respectively.

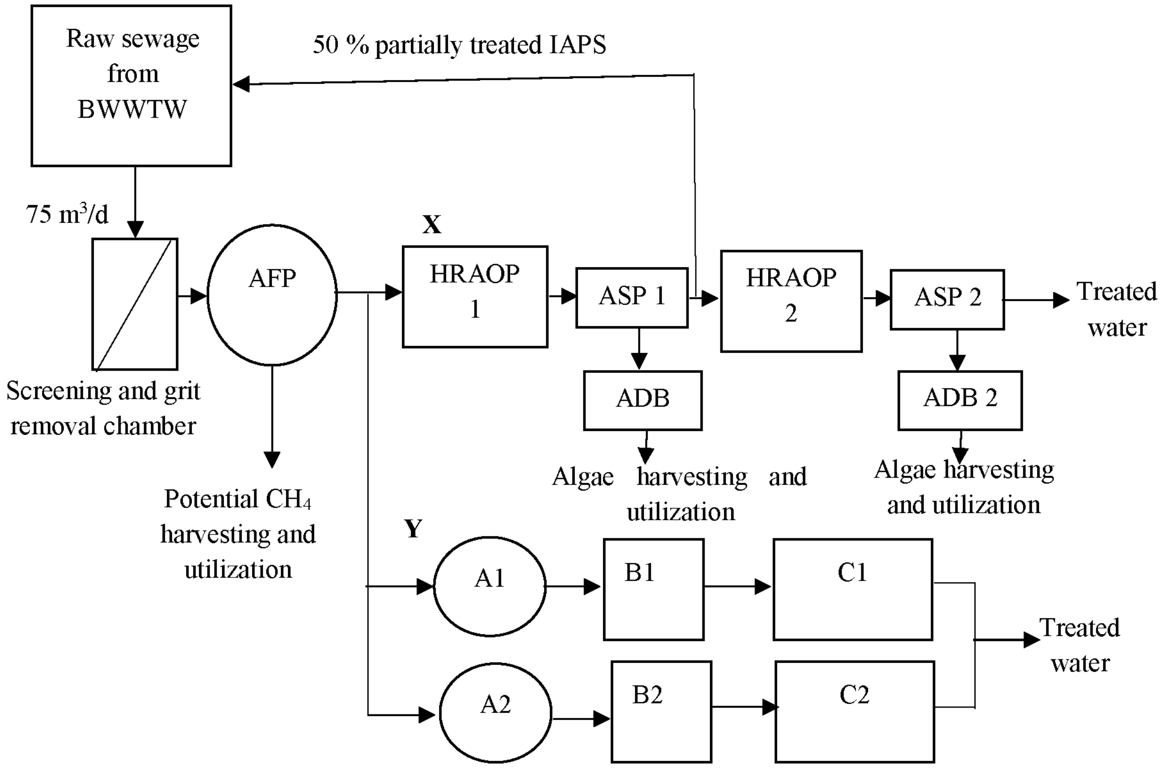

Figure 5-2. Process flow for the pilot scale V-H SSF hybrid CW (Y) and demonstration HRAOP (X) systems. Both systems received pretreated effluent from the advanced facultative pond (AFP)= primary facultative pond + fermentation pit. Key for the V-H hybrid CW system (Y): A 1 & 2= holding tanks, B 1 & 2= VSSF CWs and C 1 & 2=HSSF CWs. Key for the IAPS (X): HRAOP 1 & 2=first and second high rate algal oxidation ponds, ASP 1 & 2=first and second algae settling ponds, ADB 1 & 2=first and second algae drying beds.

Table 5-1. Summary of configuration of each treatment compartment in the IAPS at EBRU.

Parameter	PFP	FP	HRAP 1	HRAP 2	ASP 1	ASP 2	ADB
HRT (d)	20	3	2	4	0.5	0.5	-
Volume (m^3)	1500	225	150	150	19	19	13.75
Surface area (m^2)	840	50	500	500	12.5	12.5	22.9
Depth (m)	3	4.5	0.3	0.3	1.5	1.5	0.6

5.2.2 Sourcing of metadata

To evaluate the quality of the treated wastewater from the V-H SSF hybrid CWs and HRAOPs, metadata was sourced from Chapter 3 (i.e. HP of the V-H SSF hybrid CWs). Water quality data for the V-H SSF hybrid CWs from the HP was chosen in preference to the LP because this gives a clear comparison with the HRAOPs whose influent feed water was not diluted. However, data for the IAPS was obtained from EBRU during monitoring of the IAPS for the study periods 2002, 2006, 2012/2013 and 2013/2014 (Cowan et al., 2016).

5.2.3 Estimation of land area requirement

To evaluate the HRAOPs and V-H SSF hybrid CWs in terms of land footprint requirement, it was pertinent to compare the two systems treating wastewater of the same population equivalent. Thus, the EBRU IAPS was considered as a model with a treatment capacity of 75 m^3/d. For the HRAOP system, the land area occupied by the HRAOPs in the IAPS at EBRU was taken into consideration. However, since treated water from the HRAOPs is disposed into the environment after settling of algae in the ASPs and ADB are required for temporary storage of the harvested algae before the intended use (Figure 5-2), the area occupied by the ASPs and ADBs was also taken into consideration (Table 5-1). Furthermore, according to operation of the IAPS at EBRU, the second HRAOP treated 50 % of the effluent from the first HRAOP, which was attributed to design limitation of the system. According to the initial design of the IAPS (Figure 5-1), the 50 % partially treated water from the first HRAOP was supposed to be recirculated back into the AFP with a major intention of diluting the raw sewage while minimising foul odour (Green et al., 1996). Therefore, in this study, the area of the HRAOP system was estimated by considering that the HRAOPs treated 100 % effluent from the AFP, which implied an additional HRAOP and ADBs. In summary, the area requirement for the HRAOP was calculated by summing up of the area occupied by the 3 HRAOPs, 3 ASPs and 6 ADBs.

For comparison, the land area requirement for the V-H SSF hybrid CW treating 75 m^3/d was also estimated. Initially, the NH_4^+-N rate constant (K_{NH4+}) of the V-H SSF hybrid CWs described in Chapters 2 & 3 was estimated by employing equation i in Chapter 1. The parameters employed in the calculation are as follows:

- Surface area (A_s) of the treatment bed of 3 m^2
- A conservative design flow rate (Q) of 0.1 m^3/d
- Influent NH_4^+-N concentration (C_{in}) during the HP of 30.5 mg/L
- Effluent NH_4^+-N concentration (C_o) expected to be produced from the V-H SSF hybrid CWs of 6 mg/L.

Thus, by employing equation ii, the K_{NH4+} value of **0.05 m/d** was derived. Additionally, by employing equation i, the K_{NH4+} value obtained was then used to estimate the surface area of the V-H SSF hybrid CW that would be required to treat 75 m^3/d of wastewater to acceptable DWS NH_4^+-N discharge standards. The influent and effluent NH_4^+-N concentrations employed in the calculation were 30.5 mg/L and 6 mg/L respectively.

5.2.4 Estimation of capital costs

In South Africa, many WSP system are overloaded hence produce a poor effluent due to an ever-increasing population. An example is the Bedford wastewater treatment plant (BWWTP) located in Eastern Cape, South Africa. BWWTP was designed with a capacity of 560 m^3/d. However, since 2009 the system operates 59 % overcapacity. The population of Bedford is expected to increase to 23000 by the year 2025, with corresponding treatment capacity of 1750 m^3/d, which indicated a requirement for urgent upgrade. The Amathole District Municipality, in association with Mvula Trust thus, identified the IAPS technology, developed by EBRU, Rhodes University, as a potential alternative to the activated sludge process due to its relatively lower skill requirements and running costs. Therefore, following a tender document passed by Amathole District Municipality requiring upgrade of the BWWTP in 2009, UWP consulting, was contracted to provide a conceptual design based on the process design of the IAPS at EBRU and, as well as cost estimates for conversion of BWWTP into an IAPS (Appendix 11). Since the IAPS at BWWTP was designed and costed based on the process flow of IAPS at EBRU, therefore, cost estimates for implementing an AIPS at BWWTP were extrapolated to provide an estimated cost of implementing the IAPS treating wastewater for 500 PE (75 m^3/d) at EBRU. However, since the cost of the ADBs were not included in the BWWTP, these were estimated by extrapolation of costs provided by Dekker (2002).

On the other hand, the capital costs of installing a V-H SSF hybrid CWs treating wastewater of 500 PE (75 m^3/d) was estimated by extrapolation of the costs incurred to set-up the pilot scale V-H SSF hybrid CW described in Chapters 2 (Appendix 12). The costs of the two treatment systems were estimated based on undiscounted capital (Mburu et al., 2013). In both cases, only direct costs were considered. The 14 % value added tax (VAT) was excluded while a design period of 25 years for both systems was adopted. The cost of land was neglected with an assumption that the land area occupied by the WSPs should be sufficient to accommodate the proposed wastewater treatment system.

5.3 Results

5.3.1 Quality of treated water from the HRAOPs and V-H SSF hybrid CWs

Table 5-2 presents data for the quality of treated water generated from a pilot scale V-H SSF hybrid CW during the HP and the demonstration HRAOP. Results show that the quality of treated water from the V-H SSF hybrid CW met the DWS (2013) effluent discharge limit regarding all parameters except NH_4^+-N and COD. On contrary, the quality of treated water from the HRAOP system met the DWS (2013) effluent discharge limit regarding all parameters except TSS and FC.

Table 5-2. Physico-chemical composition of treated water from a pilot-scale V-H SSF hybrid CW and a HRAOP system at EBRU, Rhodes University.

Parameter (units)	General limit: Irrigation[A]	General limit: Discharge[A]	Quality of the treated water V-H hybrid CW	HRAOPS
pH	5.5-9.5	5.5-9.5	7.0 ± 0.1, (5)	9.6 ± 0.3, (4)
DO (mg/L)	>2	>2	2.2 ± 0.2, (5)	5.6 ± 0.1, (2)
EC (mS/m)	70 mS/m above intake to a Maximum of 150 mS/m	70 mS/m above intake to a Maximum of 150 mS/m	122.0 ± 5.6, (5)	110.0 ± 2.0, (2)
NH_4^+-N (mg/L)	6	6	15.9 ± 4.4, (5)	3.5 ± 1.2, (4)
NO_3^--N + NO_2^--N (mg/L)	15	15	0.2 ± 0.0, (5)	11.7 ± 3.2, (4)
PO_4^{3-}-P (mg/L)	10	10	9.2 ± 1.6, (5)	4.4 ± 0.6, (4)
COD (mg/L)	75	75	105 ± 7.5, (5)	65.8 ± 2.5, (4)
TSS (mg/L)	25	25	7.0 ± 1.0, (5)	47.3 ± 7.4, (4)
FC (CFU/100 mL)	1000	1000	<1000, (6)	>1000

Mean ± SE and number of data, n (in the parentheses) used to calculate the means are provided

[A] represents DWS (2013) Revision of general authorisations in terms of section 39 of the national water act (Republic of South Africa, *Water Act No. 3607, 1998*)

5.3.2 Land area requirement

Table 5-3 provides a summary of the estimated land area required for establishment of a V-H SSF hybrid CW and a series of HRAOP system after an AFP. Results show that even though for treatment of domestic wastewater for the same PE (500 PE; 75 m^3/d), a V-H SSF hybrid CW requires a larger area (2440 m^2) than HRAOPs (1620 m^2), a ratio of land area requirement for V-H SSF hybrid to HRAOP of 1.5, indicates that the area of the two systems is comparable.

Table 5-3. Estimated land area requirement of a V-H SSF hybrid CW and HRAOP for treatment of domestic wastewater of 500 PE (75 m^3/d).

Area	V-H Hybrid CW	HRAOP
Area for 500 PE (m^2)	2440	1620

5.3.3 Capital costs

Table 5-4 presents the estimated capital costs for establishing a V-H SSF hybrid CW and HRAOPs for treatment of domestic wastewater for 500 PE (75 m^3/d). However, Table 5-5 shows a summary of comparison of the estimated land area and capital cost requirement of the V-H SSF hybrid CW and HRAOP series based on PE considering a design period of 25 years. Installation of a V-H SSF hybrid CW requires extremely more capital (US $ 290300) than a

HRAOP system (US $ 35100). The capital costs requirement per PE/yr for the V-H SSF hybrid CW is therefore about US$ 23 compared to US$ 3 for HRAOPs (Table 5-5). However, replacing gravel with discard coal tremendously reduces the capital costs by 26.1 % (Table 5-4) resulting in capital costs of about 17 US $ PE/yr (Table 5-5).

Table 5-4. Estimated capital costs (rounded to the nearest 100) of a V-H SSF hybrid CW and HRAOPs with a treatment capacity of 75 m^3/d (based on design lifespan of 25 years)

Description	Cost (US $)	% cost
V-H SSF hybrid CW system:		
Construction materials	91700	31.6
Cost of gravel (incl. transport to the site)	75700	26.1
Plastic lining	121100	41.7
Macrophyte collection & planting	1800	0.6
Total cost	**290300**	**100**
HRAOP system:		
3 HRAOP costs (incl. mechanical/electrical installations)	27600	78.6
ASPs	6500	18.5
ADBs	1000	2.8
Total cost	**35100**	**100**

Table 5-5. Comparison of the land area and capital costs requirements of a V-H SSF hybrid CW and HRAOPs based on PE over a design lifespan of 25 years.

Cost	V-H SSF hybrid CW using gravel	V-H SSF hybrid CW using discard coal	HRAOPs
Area for 500 PE (m^2)	2440	2440	1620
Area/PE (m^2/PE)	4.9	4.9	3.2
Total capital for 500 (US $)	290300	214600	35100
Total capital costs/yr (US $/yr) for 500 PE	11600	8600	1400
Total capital cost/PE/yr (US $/PE/yr)	23	17	3

5.4 Summary

This chapter has evaluated the use of V-H SSF hybrid CWs and HRAOP systems for implementation after an AFP to supplant an overloaded and dysfunctional WSP system. Findings from the study show that:

- The quality of treated water from a V-H SSF hybrid CW meets the effluent discharge standards for all water quality parameters except NH_4^+-N and COD while HRAOPs meet the effluent discharge standards for all water quality parameters except TSS and FC.
- The land area requirement for the V-H SSF hybrid CW is comparable to that of HRAOPs for treatment of wastewater for the same PE.
- The capital costs required for implementing a V-H SSF hybrid CWs are notably higher than that required for HRAOPs for treatment of wastewater for the same PE.

Chapter 6 General discussion and conclusion

Treatment of wastewater using waste stabilization ponds (WSP) is a widespread technology that is in use in many developing countries. However, their use remains inappropriate especially in places where strict effluent discharge standards for nitrogen and TSS exist. As previously mentioned (Chapter 1), one of the major reasons contributing to excessive nitrogen levels in WSP effluent is stratification, which results in a very narrow zone for nitrification, since the aerobic zone is limited to the upper 50 cm (Baskaran et al., 1992) and inadequate floc formation required for attachment of nitrifying bacteria. As a result, it is claimed that populations of nitrifying bacteria in WSPs are inadequate to satisfactorily carry out the nitrification process (Zimmo et al., 2000; McLean et al., 2000; Babu, 2011). Furthermore, WSPs require a large land footprint for implementation. However, given the high population growth and urbanisation in most towns in sub-Saharan Africa, there is a need to explore other wastewater treatment technologies that are efficient and cost effective in terms of both land requirement and investment.

Subsurface flow constructed wetlands (SSF CWs) emerge amongst a few wastewater treatment options that have been studied and applied, mostly in temperate and to a minor extent in tropical and subtropical regions, especially in sub-Saharan Africa. While literature survey revealed that SSF CWs are efficient at removing TSS, organic matter and particulate nutrients from treated water, these systems produce an effluent that does not meet NH_4^+-N effluent discharge limits for all environmental regulating bodies in sub-Saharan Africa (Chapter 1). Chapter 1 further revealed that most SSF CWs reported in the literature are designed or operated based on BOD or COD removal as parameter. Therefore, use of BOD/COD as a design parameter in estimating the surface area for SSF CWs was suggested as a major factor contributing to the poor nitrogen removal efficiencies reported in literature since Vymazal (2005) claims that using BOD underestimates the surface area required for nutrient removal. Since nitrogen is reported to be an important design parameter for SSF CWs (Huang et al., 2000), the work presented in this thesis focuses on evaluating SSF CWs designed and operated using nitrogen as a major parameter. In addition, the contribution of the treatment media in SSF CWs was assessed following replacement of gravel with discard coal. The outcome of these studies was then used to determine the most appropriate technology for use in mitigating dysfunction and/or overloading of traditional WSP sewage treatment plants, which was achieved by comparing water quality, land footprint and capital costs of V-H SSF hybrid CWs with a HRAOP series.

Contrary to previous studies that employed BOD as design and or operational parameter for the V-H SSF hybrid CWs (Keffala & Grabi, 2005; Abidi et al., 2009; Kouki et al, 2009), this study employed NH_4^+- N rather than BOD to design V-H SSF hybrid CWs. Since the major objective of V-H SSF hybrid CWs is nitrogen removal, this is the first response to Reed et al. (1995) argument that the design of CWs should be based on the treatment objective. Until the present study was conducted, there was no information about the impact of using nitrogen as design parameter on the surface area and water quality derived for V-H SSF hybrid CWs.

Based on the concept by Vymazal (2005) that using BOD as a design parameter underestimates the area required for removal of nutrients, one can hypothesize that using nutrients e.g. NH_4^+-N to design SSF CWs results into a bigger area than using BOD and therefore a better treatment performance is expected for both NH_4^+-N and organic matter (BOD and COD). Nevertheless, the present study reveals that Vymazal (2005) concept is circumstantial. Initially, the larger surface area derived using NH_4^+-N than BOD as a design parameter for SSF CWs (Chapter 2; Table 2-3) agrees with Vymazal (2005) concept. Based on the non-mechanistic model proposed by Reeds et al. (1995), the estimated surface area for SSF CWs depends on various factors including effluent discharge limit, influent NH_4^+-N concentration and temperature to mention a few. Therefore, in South African context with specific reference to Grahamstown, Vymazals' concept may particularly hold for instance if the design is based on extremely low NH_4^+-N discharge limits such as 3 mg/L as revealed using the DWS (1998) NH_4^+-N discharge limit (Table 2-3). The present study further reveals that the NH_4^+-N concentration in the AFP effluent increases over time (Chapter 3, Table 3-3). For instance, the NH_4^+-N concentration recorded in the AFP effluent during the initiation phase and HP was found be significantly higher than that reported about a decade ago by Rose et al. (2002). This could be attributed to the aged AFP facility. According to Rose et al. (2002), the AFP was commissioned in 1996 and was designed for a life span of 15 years implying that it was supposed to be decommissioned in 2011. The implies that presently, the AFP is operating beyond its lifespan. Therefore, it is likely that the fermentation pit is filled up with sludge thus, releasing the mineralized organic materials in form of NH_4^+-N into the primary facultative pond hence an increase in NH_4^+-N concentration in the AFP effluent. This suggests that as the AFP ages, the strength of NH_4^+-N concentration in its effluent increases correspondingly. An increase in NH_4^+-N concentration in the AFP effluent over time as observed in the present study strongly affects the land footprint requirement of SSF CWs since the area requirement of SSF CWs is directly proportional to influent concentration (Reeds et al., 1995). For instance, the present study revealed that SSF

CW designs based on significantly high NH_4^+ N concentrations such as ≥ 15 mg/L intended to meet the DWS (2013) NH_4^+ N discharge standard of 6 mg/L require a surface area, which is bigger than that derived using BOD (Appendices 5 & 6). This further supports Vymazal (2005) concept. These results suggest that using NH_4^+-N as design parameter may be important in increasing the effective surface area required for the SSF CWs especially where significantly higher NH_4^+-N concentration than 15 mg/L from a primary treatment unit is expected.

However, as previously mentioned, Vymazal (2005) concept is conditional. Further analysis reveals that with an increase in the NH_4^+-N discharge limit from 3 mg/L in 1998 to 6 mg/L in 2013, the surface area for the SSF CW estimated using NH_4^+-N as a design parameter is equals to that while using BOD (Chapter 2), which reveals a contradiction with Vymazal (2005) concept. This suggests that the significance of using NH_4^+-N as a design parameter for SSF CWs may be important depending on the NH_4^+-N effluent discharge limit which vary from country to country. Therefore, in South Africa, basing on the DWS (2013) effluent discharge limit, it can be suggested that the choice of the design parameter for the SSF CW may not matter. In this case, one may choose to use either NH_4^+-N or BOD as a design parameter for SSF CWs since both parameters generate similar surface area. This may be the case 'only' where the primary treatment unit generates an effluent that is described as weak in terms of NH_4^+-N concentration such as that used in the present study as reported by Rose et al. (2002).

The V-H SSF hybrid CWs described in chapter 2 were further operated using NH_4^+-N as a major parameter and their performance was evaluated to determine if the final water quality meets the South African DWS General Authorizations for discharge to the environment for the selected physico-chemical parameters (Chapter 3).

Concerning nitrogen composition, NH_4^+-N and NO_3^--N + NO_2^--N are the common nitrogen species of environmental concern present in municipal wastewater. However, since NO_3^--N + NO_2^--N concentration in the AFP effluent and after treatment by the V-H SSF hybrid CWs was very low (i.e. <1 mg/L on average) during all operational phases (Table 3-4 & 3-6), the discussion concerning nitrogen water quality is mainly based on NH_4^+-N.

The present study reveals that the V-H SSF hybrid CWs designed and operated using NH_4^+-N as described in Chapters 2 & 3 failed to reduce NH_4^+-N to within the limits for discharge into the environment as specified by the DWS during the initiation phase, which was not expected.

This is because the large surface area derived using NH_4^+-N as design parameter was thought to be sufficient for NH_4^+-N removal. According to Wu et al. (2008), the larger surface is associated with longer hydraulic retention time (HRT) and therefore better treatment performance is likely. Since the main mechanisms responsible for nitrogen removal from SSF CWs are nitrification and denitrification that are microbial mediated (Cooper et al., 1996; Vymazal et al., 1998; Stefanakis et al., 2014), it can be reasoned that efficient removal of NH_4^+-N via these processes depends on the time of interaction between water and the treatment bed which depends on the HRT. However, the poor NH_4^+-N water quality from the V-H SSF hybrid CWs could be attributed to operation of the systems at high HLR. Due to the high fluctuation in ET rates (Appendix 7), the V-H SSF hybrid CWs were operated at significantly higher mean HLR than the deign value during the initiation phase. HLR has been reported to influence pollutant loading rate and HRT (Reeds et al., 1995; Tousignant et al., 1999). Furthermore, the performance of SSF CWs depend on proper HLR and HRT (Kantawanichkul & Somprasert, 2004; Toet et al., 2005; Xu et al., 2014). Therefore, operation of the V-H SSF hybrid CWs at higher HLR than the designed value reduced the theoretical HRT from the designed value of 3 d to 2 d thus affecting the removal of NH_4^+-N from the V-H SSF hybrid CWs during the initiation phase and therefore the observed poor treated water quality. Furthermore, organic loading rate affects the performance of CWs (Kadlec et al., 2000; López-Rivera et al., 2015; Andreo-Martínez et al., 2016). Thus, Tousignant et al. (1999) suggest that optimal performance of CWs depends on using an appropriate area for a given organic load. Organic load is a function of HLR and influent concentration of the parameter per unit area. Besides the high HLR, during the initiation phase, the AFP effluent recorded notably higher NH_4^+-N concentration than 12.1 mg/L reported by Rose et al. (2002) and used that was used to design the V-H SSF hybrid CWs employed in the present study by an approximate factor 2 which was not expected. The cause of the unexpected high NH_4^+-N concentration in the AFP effluent and its influence on NH_4^+-N concentration was previously discussed in this Chapter. A combination of operating the V-H SSF hybrid CWs at higher HLR and influent NH_4^+-N concentration therefore, resulted in notably higher influent NH_4^+-N loading rate than the designed value (Tables 2-1 & 3-2). In fact, an attempt made to estimate the surface area required to handle the influent NH_4^+-N loading rate applied to the V-H SSF hybrid CWs during the initiation revealed that a surface area of ~ 4 m^2 would be required to meet the DWS (2013) NH_4^+-N effluent discharge limit (Appendix 5). This suggests that the surface area of the V-H SSF hybrid CWs was insufficient to handle the high influent NH_4^+-N loading rate that was applied to the V-H SSF hybrid CWs and therefore resulting into poor NH_4^+-N removal.

Additionally, the poor NH_4^+-N removal from the V-H SSF hybrid CWs could be attributed to the effect of physico-chemical parameters. Nitrification one of the most important nitrogen transformation processes in CWs is an oxygen demanding process requiring 43 g of O_2 per g of ammonium oxidised (Schäfer et al., 1998). Thus, the mean influent oxygen concentration of 5.1 mg/L (0.29 g O_2/m^2/d) was insufficient for complete oxidation of 1.17 g/m^2/d of NH_4^+-N thus, resulting in the poor effluent quality. However, the observed increase in NO_3^--N + NO_2^--N in the VSSF (Appendix 10) indicated that nitrification was occurring, but the source of oxygen could have been mostly by atmospheric aeration since the influent pipe was held at about 0.3 m above the VSSF CW surface. Furthermore, the reduction of NO_3^--N + NO_2^--N from 3.3 ± 1.3 mg/L in the influent to 0.3 ± 0.0 mg/L in the effluent of the HSSF CWs showed that denitrification was occurring. Hence, nitrification and denitrification were among the nitrogen transformation and removal processes that accounted for the 62.4 % NH_4^+-N removal from the V-H SSF hybrid CWs.

Following a theory by Kwantawanichkul & Somprasert, 2004; Toet et al., 2005 that suggests that operating CWs at proper HLR and HRT significantly improves its performance, the optimisation phase was launched; initially with major aim of varying the operational parameters (HLR and HRT) during the HP to improve the performance of the V-H SSF hybrid CWs. During the HP, it was assumed that the influent NH_4^+-N concentration that was recorded during the initiation phase would reduce to within the designed value of the V-H SSF CWs due to increase in temperature experienced during spring and summer period. Thus, the reduction in HLR and influent NH_4^+-N concentration was expected to maintain the NH_4^+-N loading rate to within the design limits, which would result in improved performance of the system and therefore improved NH_4^+-N water quality. As expected, a reduction in HLR during the HP slightly improved the general performance of the V-H SSF hybrid CWs with regards to NH_4^+-N removal. Nevertheless, the water quality failed to meet the DWS (2013) NH_4^+-N effluent discharge limit. Thus, it is suggested that in addition to the low DO concentration in the AFP effluent that was insufficient for complete nitrification (Table 3-3), the poor NH_4^+-N water quality obtained during the HP is attributed to operating the V-H SSF hybrid CWs at a significantly higher influent NH_4^+-N loading rate than the designed value. This reason is related to López-Rivera et al. (2015) and Andreo-Martínez et al. (2016) who reported that organic loading rate affects the performance of CWs. The significantly high NH_4^+-N loading rate during HP was however, not expected since improvement in air temperature during spring and summer period was expected to increase the biological activity and therefore improvement in NH_4^+-N removal (Kayombo et al., 1998; Mara, 2005; Garcia et al., 2010). Consequently, the lower NH_4^+-N concentration in the AFP effluent would be expected during the HP than during the initiation phase. Thus, a combination of reduction in HLR and influent NH_4^+-N concentration was expected to result into lower NH_4^+-N loading rate during the HP than that recorded during the initiation phase. However, surprisingly, the NH_4^+-N concentration from the AFP during spring and summer increased above 12.1 mg/L reported in literature (Rose et al., 2002) and that recorded during the initiation phase (Table 3-3) hence resulting in unexpectedly higher NH_4^+-N loading rate than the design value and therefore accounting for the poor NH_4^+-N water quality as previously mentioned. The reason that accounted for the observed increase in the NH_4^+-N concentration in the AFP effluent during the HP is similar to that previously given during the initiation phase.

Despite the failure of the V-H SSF hybrid CWs to meet the DWS NH_4^+-N effluent discharge limit, the NH_4^+-N treated water quality obtained during the HP is comparable to 15 mg/L obtained by Herrera Melián et al. (2010) who operated their system continuously at high HLR using *P. australis* and *Scirpus sp* as macrophyte in a VSSF and HSSF CWs respectively but lower than 30 mg/L reported by Keffala & Ghrabi (2005) who operated their hybrid systems intermittently. In both cases, BOD was the major operational parameter. This suggests that even though operated at a significantly higher influent NH_4^+-N loading rate than the designed value, a continuously fed V-H SSF hybrid CW designed and operated using NH_4^+-N can produce an effluent comparable or even better than continuously or intermittently fed systems operated using BOD as a major parameter.

Even though a previous report by Vymazal (2005) proposes that using nutrients e.g. NH_4^+-N results into a larger treatment area for SSF than using BOD implying that a high-water quality regarding NH_4^+-N is likely, the present study reveals that this is not necessarily the case especially for highly loaded systems as observed during the HP. This is because, further analysis revealed that the significantly higher NH_4^+-N concentration recorded in the AFP effluent during the HP than the designed value would require similar surface area as that used in the present study (Appendix 6) but failed to meet the discharge limit for NH_4^+-N. This therefore implies that the failure of SSF CWs to meet effluent NH_4^+-N discharge standards is due to their inability to handle high influent NH_4^+-N loading rates. Since most pretreated sewage is characterised as medium in terms of NH_4^+-N concentration (Table 3-3, Mburu et al., 2012), it is proposed that the use of NH_4^+-N may not necessarily be a suitable design parameter for SSF CWs receiving pretreated wastewater with high NH_4^+-N concentration (i.e. characterised as medium or strong) especially where strict discharge limits for nitrogen exist.

Contrary to the HP, the V-H SSF hybrid CWs produced an effluent NH_4^+-N concentration that met the South African NH_4^+-N effluent discharge limit during the LP. This was attributed to maintaining the HLR and influent NH_4^+-N concentration and consequently the influent NH_4^+-N loading rate to within design values. Indeed, the NH_4^+-N effluent concentration obtained from the V-H SSF hybrid CWs is better than that reported from literature in which BOD was the major operational parameter (Keffala & Ghrabi, 2005; Herrera Melian et al., 2010). The ability of the V-H SSF hybrid CWs to meet the NH_4^+-N effluent discharge standard during the LP suggests that performance of SSF CWs not only depend on proper HLR and HRT as suggested by Toet et al. (2005) but also proper surface loading rate. Certainly, influent NH_4^+-N loading rate showed a strong positive linear relationship with NH_4^+-N removal (Table 3-7). Given the fact that DO concentration recorded in the AFP effluent during LP was insufficient for complete nitrification, it can be suggested that operating SSF CWs at proper surface loading rate could be important in influencing removal of nitrogen via other routes such ANAMMOX that is independent of DO. Besides, macrophytes have been documented to play an important role in pollutant removal from CWs (Brix, 1994; 1997). However, although the role of macrophytes in nutrient uptake is considered insignificant in SSF CWs (Vymazal, 2007), the potential of CWs designed to meet NH_4^+-N effluent discharge limit may depend on the choice the macrophyte. Among the unique characteristics of *P. australis* used in the present study is the high evapotranspiration (ET) rates. There is no information about ET rates of *P. australis* from V-H SSF hybrid CWs. However, Holcová et al. (2009) report that *P. australis* ET rates fluctuate between 4.7-12.4 mm/day depending on meteorological conditions. Ondok et al. (1990) estimated similar ET rates ranging from 6.9-11.4 mm/d for *P. australis* from studies carried out in summer from temperate region. On the contrary, Pedescoll et al. (2013) reported a higher mean ET rate of 23.0 mm/d from hydroponic wetlands planted with *P. australis* over a complete growing cycle from a Mediterranean region which is comparable to the mean ET recorded for *P. australis* during the present study. The high ET rates for *P. australis* obtained from the present study have demonstrated implication for nutrient removal which was revealed through a strong positive linear relationship between ET with NH_4^+-N removal. These results agree with Platzer & Netter (1994) who reported high NH_4^+-N removal rates ranging from 40-70 % with increase in ET rates. The stronger positive linear relationship observed between ET rate and NH_4^+-N removal during LP than during either the initiation or HP (Tables 3-5 & 3-7) suggests that NH_4^+-N removal because of water loss via plant uptake is an important nitrogen removal mechanism. Therefore, due to the high ET rates of *P. australis* its selection for use in SSF CWs particularly for meeting nutrient standards especially for nitrogen may be important. In summary, ability of the V-H SSF hybrid CWs to meet the NH_4^+-N effluent discharge standard during LP demonstrate the feasibility to use NH_4^+-N as a design and operational parameter for SSF CWs receiving pretreated wastewater described as weak in terms of NH_4^+-N concentration.

Several studies including Abidi et al. (2009) and Herrera Melián et al. (2010) have reported contradicting results about water quality regarding COD from V-H SSF hybrid CWs. For instance, Herrera Melián et al. (2010) reported low COD effluent concentrations of 72 mg/L during continuous operation of a V-H SSF hybrid CW over a seven-month monitoring period, which is comparable to that obtained during the initiation phase i.e. six-month monitoring period. Although the COD effluent concentration obtained during the initiation phase met the DWS (2013) discharge limit, these results cannot be relied on due to the short monitoring period and given the poor operating conditions of the V-H SSF hybrid CWs. Indeed Gopal (1999) speculated that laboratory and short-term field experimental results have been over generalized to represent a more promising account of the performance of CW systems.

Although Vymazal (2005) hypotheses that using BOD as a design parameter for CWs results in under-estimation of the area; hence this method is appropriate for organic matter and TSS removal but not for nutrients, findings of the present study revealed that changing the design parameter from BOD to NH_4^+-N results into a large area. However, the large area derived for the V-H SSF hybrid CWs designed using NH_4^+-N did not meet the DWS (2013) discharge limit for COD during the optimisation phase (HP & LP). In fact, the COD effluent concentration achieved during the optimization phase i.e. HP and LP after long term monitoring of the system is comparable to 134 mg/L reported by Abidi et al. (2009) in which COD was utilized as a major parameter. This implies that regardless of the design parameter and influent loading rate, *P. australis* based SSF CW don't meet the DWS COD discharge standard on a long-term basis.

The following reasons are proposed to be responsible for the deterioration in COD water quality during the optimization phase. Firstly, the mechanisms of soluble COD removal from CWs are aerobic and anaerobic microbial degradation (Vymazal, 2007; Eslamian, 2016). Thus, surface area of a CW may greatly influence its performance by influencing the microbial populations that carryout the different degradation processes. Surface area and bacterial abundance for instance have been studied and a strong correlation between the two parameters was reported (Deflaun & Mayer, 1983). Therefore, coupled with proper operation, the increase in surface area of CWs because of using NH_4^+-N as a design parameter for CWs may be significant for increasing the surface area for attachment of microbial populations and therefore increased abundance resulting in better removal of pollutants such as NH_4^+-N as observed in the present study. However, it is proposed that the increase in microbial population and abundance due to increase in surface area may be disadvantageous in the overall functioning of the CW systems. Even though properly operated, microbial populations particularly bacteria have a very short life cycle of about a few hours to days (Tyagi & Pande, 2007). Such a short lifespan may imply constant death and degradation of the large numbers of worn-out bacterial cell resulting in release of organic compounds back into the system. This could partly explain the poor effluent concentration from the hybrid CW regarding soluble COD given the limited mechanisms responsible for its removal compared to NH_4^+-N whose removal occurs via a myriad of removal pathways (See Chapter 1). Secondly, the effect of temperature can be proposed as major factor for the poor COD removal recorded in the present study. While the mechanisms of soluble organic matter removal from SSF CWs are aerobic and anaerobic degradation as previously mentioned, aerobic degradation occurs mainly around the root zone due to dispatch of oxygen by the macrophytes (Brix, 1994). However, the amount of oxygen dispatched into the rhizosphere is dependent on temperature since temperature affects photosynthetic activity (Bigambo & Mayo, 2005) implying that it likewise affects COD removal, which was revealed in the present study. The positive linear relationship observed between COD removal rate and water temperature in the present study (Table 3-7) suggests that COD removal is to some extent dependent on temperature. This study is in agreement with several other studies that report that organic matter removal is affected by temperature (Shama et al., 2013; Kern, 2003). Therefore, it is suggested that the decline in temperature during the LP affected photosynthetic activity and therefore oxygen production, thus resulting in poor COD effluent concentration from the V-H SSF hybrid CWs. Thirdly, through effect of their roots, macrophyte are reported to play an important role in performance of CWs regarding COD removal by especially increasing the surface area for microbial growth that carry out the degradation process (Brix, 1994; 1997). However, release of carboneous compounds due to constant death and mineralization of *P. australis* root and rhizomes within the treatment bed is further suggested to be responsible for observed poor water quality regarding COD. This argument is linked to Hunt & Poach (2001) who explained that CWs cannot totally eliminate carbon and solid compounds because of the microbial and vegetative decay which continuously releases organic matter to the system. Furthermore, in their study of the *Phragmites* die-back, Kovacs et al. (1989) initially suggested that one of the causes of the *Phragmites* die-back was due to death and mineralization of the underground parts of the reed that led to the release of higher than normal concentrations of some volatile organic acids mainly acetic, butyric and propionic to mention a few which was later confirmed by Armstrong & Armstrong (1999). Additionally, it is well-known that living organisms contain more carbon than nitrogen implying that during mineralization, the rate at which the bound organic carbon is release back into the water column is higher than that for NH_4^+-N. Besides, soluble COD removal is accomplished exclusively via two processes compared to NH_4^+-N removal from CWs occurs through various removal pathways. Lastly but not least, sludge accumulation within the SSF bed could be another contributing factor to the poor COD removal. SSF CWs are one of a few wastewater treatment systems that are reported to produce minimal sludge (Bostanian et al., 2012). Hence this advantage has been employed by CW researchers to advocate for SSF CWs for wastewater treatment. Although as the system ages, there is a tendency for sludge to build up, the effect of sludge accumulation in SSF CWs has never been addressed in previous reports. Therefore, at this point it is reasonable to mention that the observed poor effluent regarding COD is probably due to decomposition of accumulated sludge within the system.

As expected, the V-H SSF hybrid CWs designed using NH_4^+-N were very efficient in reducing TSS during all phases to effluent concentrations, well within the DWS (2013) effluent discharge standard. The levels of TSS removal achieved in the present study are comparable to removal efficiencies of 98 % and 95 % reported by Abidi et al. (2009) and Herrera Melián et al. (2010) respectively. TSS removal from SSF CWs is mainly due to physical processes including sedimentation, filtration and adsorption provided by gravel and plant roots (Kadlec & Knight, 1996). However, the higher TSS removal rates obtained during the optimisation than the initiation phase suggests that these processes are more effective at eliminating TSS at longer than shorter HRT.

There are limited reports on PO_4^{3-}-P removal from V-H SSF hybrid CWs. Although initially, the treated water showed an increase in PO_4^{3-}-P concentration, the removal efficiency attained in the present study is comparable to 24 % reported by Herrera Melián et al. (2010). Several processes including adsorption, precipitation, plant and microbial uptake are responsible for

PO_4^{3-}-P removal from SSF CWs (Kadlec & Knight, 1996). However, the main processes that account for removal of PO_4^{3-}-P from CWs are adsorption and precipitation with metals e.g. Aluminum, Calcium and Magnesium (Arias et al., 2001; Rousseau et al., 2004; Vymazal, 2007). Unfortunately, adsorption is a reversible process occurring in aerobic environment. However, as conditions become anaerobic, phosphorus may be mobilized from the sediment and released back into the water column (Cramlet &Turyk, 2002). Therefore, the observed increase in PO_4^{3-}-P in treated water during the initiation and HP was probably due to release of adsorbed PO_4^{3-}-P since the greatest part of SSF CWs is considered anaerobic with aerobic conditions occurring mainly around the root zone (Brix, 1994). To overcome the problem associated with phosphorus removal from SSF CWs as observed during the initiation phase, Vymazal (2005) proposes that using a special media with high sorption capacity such as clay would be important. Additionally, discard coal is an industrial solid waste material that is known to be rich in metals. Therefore, it is proposed that the use of discard coal as a treatment media in CWs instead of gravel could be useful since the high metal content would facilitate the precipitation process thus reducing phosphorus from wastewater. However, following an increase in the HRT during the optimisation phase, a significantly higher removal rate was achieved than the literature value, which suggests the role of microbial uptake in PO_4^{3-}-P removal due to increase in contact time between the water and microbial populations. Despite the lower removal of PO_4^{3-}-P from the V-H SSF hybrid CWs during the initiation and HP than the LP, the PO_4^{3-}-P effluent concentration remained within the DWS (2013) discharge limit during all operational phases probably due to low PO_4^{3-}-P concentration recorded in the AFP effluent which was below discharge limit of 10 mg/L.

Escherichia coli was used as an indicator for faecal contamination and to determine the disinfection efficacy of the V-H SSF hybrid CWs. The high FC removal attained by the V-H SSF hybrid CWs during the optimization phase was expected and is in the range reported in literature for V-H SSF hybrid CWs (Keffala & Ghrabi, 2005; Herrera Melián et al., 2010), which further confirms that CWs do not consistently achieve 100 % FC disinfection. Although the mean FC effluent concentration attained during the LP doesn't meet the discharge standards, this value is not significantly different from the standard ($p=0.49$). The high FC removal attained in the present study is due to a myriad of mechanisms responsible for pathogen removal from SSF CWs including sedimentation, filtration, adsorption, die-off, UV degradation and predation (Kadlec & Knight, 1996). Consequently, influent FC loading rate showed a strong positive relationship with FC removal rate demonstrating that sedimentation, filtration and adsorption, which are mainly dependent on influent loading rate (Travaini-Lima & Sipaúba-Tavares, 2012) played a significant role in pathogen removal from the V-H SSF hybrid CWs. The ability of the V-H SSF hybrid CW to achieve 100 % faecal coliform disinfection during the HP could be additionally attributed to improvement in water temperature during spring and summer. Various studies report that in CWs, improvement in air temperature increases photosynthetic activity of plants resulting in translocation of oxygen in the rhizosphere (Brix, 1994; Bigambo & Mayo, 2005; Villalobos et al., 2013). Thus, the dispatch of oxygen in the vicinity of the roots could have encouraged high activity of predators such as protozoans resulting in elimination of faecal coliform hence better removal achieved at higher temperatures during the HP than during low temperatures (< 20 ^{o}C) recorded either during the initiation phase or the LP.

The DWS (2013) requires that WWTPs should discharge treated water into the environment with NH_4^+-N, NO^{3-}-N + NO_2^--N, PO_4^{3-}-P, soluble COD and faecal coliforms concentrations of ≤ 6 mg/L, 15 mg/L, 10 mg/L, 75 mg/L and 1000 CFU/100 mL respectively. The objectives of wastewater treatment are producing water of a quality suitable for: i) discharge into a water resource with the aim of protecting both the aquatic environment and public health; and, ii) reuse especially in agriculture.

Overall, the ability of the V-H SSF hybrid CWs to meet the DWS (2013) effluent discharge standards for nutrients (NH_4^+-N and PO_4^{3-}-P) during LP demonstrates the feasibility to use NH_4^+-N as a design and operational parameter for SSF CWs receiving pretreated wastewater described as weak in terms of NH_4^+-N concentration. This would be crucial especially for protecting water bodies that are susceptible to eutrophication with a requirement of only simple disinfection to eliminate FC particularly where the receiving waterbody is used as a source of potable water or for recreation purposes. The use of NH_4^+-N to design SSF CWs treating municipal sewage with low NH_4^+-N concentration may particularly apply where the immediate receiving water body is a river or stream since lotic ecosystems are capable of self-purification (Spellman, 2015). The self-purification process may be beneficial in reducing COD to levels that are not detrimental to aquatic life.

The present study however, revealed significantly higher NH_4^+-N concentrations from the AFP effluent during the initiation and HP than that reported about a decade ago (Rose et al., 2002). Similarly, it has been reported that NH_4^+-N concentrations in municipal wastewater are high (>

60 mg/L), which could be due to South Africa's water scarcity and consequent low water use per capita resulting in rise in urine per litre water use and therefore increasing the NH_4^+-N concentration (Ekama George, Professor of civil engineering, University of Cape Town, Personal notes, 14th December 2017). This implies that currently, the NH_4^+-N concentration applied to the V-H SSF hybrid CWs during the LP is unlikely to be encountered in South Africa even with the best pre-treatment available, unless separation of urine at the source is practiced-urine contributes about 80 % of the wastewater NH_4^+-N concentration. Furthermore, despite the outdated AFP structure, the high NH_4^+-N concentration recorded in the AFP effluent during the HP is indicative of the typical wastewater quality that is likely to be generated from a well-functioning pretreatment facility. The failure of the V-H SSF hybrid CWs to meet the NH_4^+-N effluent discharge limit during the HP therefore suggests that SSF CWs designed using NH_4^+-N can hardly be applied as secondary wastewater treatment systems to treat wastewater characterized as medium to strong in terms of NH_4^+-N concentration especially where protection of water resources from eutrophication is required.

Nevertheless, considering the drought that hit South Africa in 2015 indicates that it is water scarce country with a requirement to reuse treated wastewater from WWT systems to sustainably manage its water resources. Consequently, the ability of V-H SSF hybrid CWs to consistently meet TSS and FC during the HP phase suggests that this system is a potential secondary treatment technology for application where water reuse especially in agriculture is inevitable as may be the case in South Africa. The high FC concentration in the treated water during the LP (Table 3-6) however suggests that V-H SSF hybrid CWs are not consistent in eliminating FC, which implies that simple disinfection measures such as chlorination is crucial to ensure 100 % removal of FC before water reuse in areas with high variability in temperature such as Grahamstown. The high NH_4^+-N concentration could lower the capital costs incurred in procurement of inorganic fertilizers while the high COD concentration in the effluent may not be harmful to plants but rather could be beneficial for adding organic matter to poorly developed soils (Babu, 2011).

The performance of V-H SSF hybrid CWs designed and operated using NH_4^+-N, planted with *P. australis* and operated under continuous flow was studied for treatment of municipal sewage in Grahamstown from February 2015 to July 2016 under three phases i.e. the initiation phase, HP and LP. In conclusion, under optimal operating conditions of HLR, HRT and influent NH_4^+-N loading rate during the LP, the V-H SSF hybrid CWs designed and operated using NH_4^+-N as a major parameter under Grahamstown climatic conditions produced an effluent that met the DWS (2013) effluent discharge limit into the environment for all water quality parameters except COD and FC. This suggests that NH_4^+-N is an important design and operational parameter for SSF CWs treating municipal sewage characterized as weak in terms NH_4^+-N concentration with "only" a requirement for simple disinfection e.g. chlorination to eliminate FC.

The study described in this thesis (Chapters 2 & 3) employed gravel as treatment media in V-H SSF hybrid CWs. Like in literature (Keffala & Ghrabi, 2005; Abidi et al., 2009; Kouki et al., 2009; Herrera Melián et al., 2010), gravel was used in the present study in preference to other treatment media that have been tested and proved efficient for use in SSF CWs for wastewater treatment including vermiculite, zeolite, lime, lapilli, charcoal and slag to mention a few (Brix et al., 2001; Herrera Melián et al., 2010; Zhu & Ketola, 2011; Sheridan et al., 2013) due to its local availability in Grahamstown. Indeed, the occurrence of gravel in almost every geographical region is the major reason as to why it is traditionally used as a treatment media not only in SSF CWs but also in other passive WWT systems such as trickling filters. However, the problem associated with using gravel as a treatment media in SSF CWs is that, in areas where it is sourced, there may result in negative environmental impacts including air pollution, erosion and sedimentation of particulate material into nearby surface water bodies and landslides. Additionally, it may not be economically feasible to transport large quantities of gravel in an instance where a stone quarry is situated far away from the WWT facility.

Therefore, given the negative environmental and economic impacts associated with excavation of gravel, treatment media that can be used as an alternative to gravel in SSF CWs for wastewater treatment are sought. On the other hand, in most countries that rely on mining as a major economic activity, pollution arising from dumping of industrial solid wastes such as discard coal, gold, platinum etc. cause threats both to the environment and public health as previously mentioned (Chapter 4). Therefore, to mitigate environmental degradation arising from dumping industrial solid wastes, it is proposed that such materials could be used as an alternative treatment media to gravel in SSF CWs especially in mining areas where they are locally available. This would assist in controlling the negative environmental impacts arising from both dumping of industrial solid wastes and excavation of gravel. Therefore, the work described in this thesis (Chapter 4) investigated the feasibility of using industrial solid waste materials with a special focus on discard coal to replace gravel as support media in SSF CWs on laboratory scale using HSSF CWs. To the best of my knowledge until this study was carried out, there has not been any information about investigations into the use of discard coal as an alternative treatment media to gravel in SSF CWs. Unlike in the earlier study (Chapter 3), the work described in Chapter 4 employed HSSF CW; a section of the V-H SSF hybrid CW system due to financial constraints. Furthermore, there are various industrial solid wastes arising from mining activities in South Africa and these were previously mentioned in this chapter. However, discard coal was chosen for the study because presently, South Africa's economy depends largely on coal mining. Hence, it is likely that most environmental problems faced in most mining areas in South Africa arise from dumping of discard coal. As previously mentioned (Chapter 4), the specific objectives of the study were to: i) find out if discard coal supports growth and propagation of the macrophyte *P. australis*; and, ii) assess the effluent quality from HSSF CWs in which coal discard was used as treatment media.

Although dryland bio-remediation is the well-established method for rehabilitation of abandoned coalmines, dryland plants are reported to find difficulty in establishing on coalmine waste due to low pH and lack of readily available macronutrients (Otte & Jacobs, 2009; Limpitlaw et al., 2015). However, the present study revealed that discard coal-containing HSSF CWs fed with either tap or AFP water supported growth of *P. australis* in a similar way to gravel-containing HSSF CWs. These results suggest that *P. australis* is a suitable candidate for use in discard coal based HSSF CWs. Additionally, several studies report that coalmine waste is characterized by low pH (Annandale et al., 2009; Campaner et al., 2014). However, the pH recorded from water after treatment through all HSSF CWs was near neutral indicating that the ability of coal- containing HSSF CWs to support the growth of *P. australis* is due to the potential of SSF CWs systems to buffer highly acidic mine water to close to neutral (Otte & Jacobs, 2009). Besides, *P. australis* is adapted to a wide range of environmental conditions including survival on soils with varying pH (3.7-8.7) and poor nutrient concentration (Chabreck 1972; Rydén et al., 2003; Mal & Narine, 2003) as evidenced in this study. The better plant growth recorded for *P. australis* in discard coal-containing HSSF CW fed with AFP water than that fed with tap water was however, due to higher essential nutrient concentration (N, P & S) recorded in the AFP effluent than in tap water (Tables 4-2). This suggests that while changing the treatment media from gravel to discard coal does not affect the growth of *P. australis*, for its successful colonisation, a nutrient rich water source is required. Despite successful establishment of *P. australis* on discard coal, plant health assessment (i.e. chlorophyll fluorescence (Fv/Fm)) revealed values from all experiments, which were significantly lower than an optimal Fv/Fm value of ~0.83 ($p=0.00$) reported for many plant species (Johnson et al., 1993). According to Johnson et al. (1993), lower values than 0.83 suggest plant stress. Various factors are reported to induce plant stress including biotic and abiotic factors (Haferkamp, 1988; Kennelly et al., 2012). The general decline in temperature from summer through winter could be responsible for the observed plant stress in all the experiments. Additionally, in discard coal- and gravel-containing HSSF CWs fed with tap water, deficiency of essential nutrients in tap water could have affected plant health while presence of heavy metals in discard coal could be responsible for observed plant stress in discard coal- containing HSSF CWs.

Apart from supporting plant growth, the present study revealed that changing the treatment media from gravel to discard coal is important in achieving a better treated water quality particularly regarding nutrients. Thus, the reduction of PO_4^{3-}-P and NH_4^+-N from AFP effluent after treatment by the discard coal-containing HSSF CW was especially surprising since it would be expected that the presence of P and N in discard coal would result in increase of PO_4^{3-}-P and NH_4^+-N concentration in the treated water. As earlier on discussed in this chapter, the main mechanisms of P removal from SSF CWs are adsorption and precipitation (Kadlec & Wallace, 2009) with several studies reporting an increase in PO_4^{3-}-P concentration in treated water, which is attributed to release of adsorbed PO_4^{3-}-P back into the water column due to the anaerobic conditions that prevail in SSF CW (Chapter 3; Hammer, 1989). The general reduction of PO_4^{3-}-P from either gravel- or discard coal-containing HSSF CWs contrary to an increase reported in literature could however, be attributed to the high planting density. The planting density of ~ 33 plants/m^2 adopted in this study was massive compared to a recommendation of 4 or 14 plants/m^2 (UN-Habitat, 1998; US. EPA, 2001). Although nutrient removal is considered insignificant in SSF CW (Vymazal, 2007), the high planting density could have increased removal of PO_4^{3-}-P through: 1) direct plant uptake and 2) increasing the surface area for attachment of microorganisms, which further aided in PO_4^{3-}-P uptake and adsorption of phosphorus. Furthermore, although no significant difference was observed, the lower PO_4^{3-}-P concentration recorded in treated water from discard coal than from gravel-containing HSSF CWs was probably due to effect of the treatment media. The reason is that in addition to adsorption, the presence of more metals recorded from discard coal than from gravel (Table 4-1) could have allowed better precipitation of PO_4^{3-}-P and therefore better removal from discard coal- than from gravel-containing HSSF CWs.

The most interesting result of this study (Chapter 4) is that the NH_4^+-N water quality obtained from discard coal- containing HSSF CWs fed with AFP effluent is much lower than that reported for HSSF CWs that employed gravel as a treatment media (the present study, Chapter 4, Okurut, 2000; Mburu et al., 2013). The better water quality concerning NH_4^+-N concentration from discard coal- than from gravel- containing HSSF CWs could also be attributed to the effect of: i) planting density as previously discussed for PO_4^{3-}-P; and, ii) treatment media. One of the factors that limit nitrification and denitrification the major pathways responsible for nitrogen removal from SSF CWs is deficiency of carbon source in domestic wastewater (Vymazal, 2007; Lee et al., 2009). However, the presence of higher percentage of elemental carbon in discard coal than gravel (Table 4-5) suggests that it is a better treatment media than gravel for providing a sufficient carbon source required for nitrification and denitrification hence a major contributing factor to better treated water quality regarding NH_4^+-N in water after treatment by discard coal- than gravel- containing HSSF CWs.

Even though discard coal-containing HSSF CWs reduced nutrients from the feed water, they however, failed to reduce SO_4^{2-} and metals (Fe and Mg), which was due to leaching of these pollutants from discard coal. Nevertheless, the increase in SO_4^{2-} and metals concentration in the discard coal- containing HSSF CWs was not expected since mechanisms of removal of these pollutants depend largely on water logged and anaerobic conditions that are characteristic of SSF CWs. Thus, it could be thought that these two unique features of SSF CWs could be exploited to control pollution originating from discard coal containing HSSF CWs. Indeed, McLean & Bledsoe (1992) claims that pollutants originating from waste coal, particularly metals are immobile under waterlogged anaerobic conditions while SO_4^{2-} is an electron acceptor for anaerobic respiration. Thus, under anaerobic conditions, SO_4^{2-} is reduced to hydrogen sulphide (S^{2-}) (equation x) by sulphate reducing bacteria (Tuttle et al., 1969). The formation of hydrogen sulphide is important since it precipitates out metals including Fe, Mg and Zn etc. that are more stable and insoluble providing the soil remains (O'Sullivan et al., 1999, Sheridan et al., 2013) as shown in equation xi below. Me represents a metal ion.

$$SO_4^{2-}+2CH_2O+2H^+ \longrightarrow H_2S+2H_2CO_3 \qquad \text{(xiii)}$$

$$Me^{2+}+H_2S \longrightarrow MeS+2H^+ \qquad \text{(xiv)}$$

However, the failure of the discard coal- containing HSSF CWs to reduce SO_4^{2-} and metals could be attributed to the containers used for the experiments that exhibited shallower depth than 0.4 m recommended for HSSF CW (UN-Habitat, 1998). Thus, the shallow containers could probably have decreased the anaerobic zone that is important for sulphate and metal reduction. This reason could be supported by literature; in another study for instance, Mburu et al. (2013) reported a decrease in SO_4^{2-} from 66.7 mg/L in the influent to 29.3 mg/L and 20.1 mg/L in the effluent of HSSF CW1 and 2 respectively that were planted with *C. papyrus* in which the bed treatment depth was 0.6 m. This suggests that to the achieve efficient reduction of SO_4^{2-} and metals from SSF CWs, it is very crucial to consider a treatment depth that is ≥ 0.4 m as recommended by the UN-Habitat (2008). Despite the observed anomalies, the mean SO_4^{2-} and concentration in the treated water from discard coal- containing HSSF CWs is lower than that reported for coalmine water (Sigh, 1988; Annandale et al., 2009). This indicates that besides SO_4^{2-} reduction which is considered as a major S O_4^{2-} and metal removal process, other removal mechanisms such as plant uptake play a major role in removal of SO_4^{2-} and metals from SSF CWs. In fact, various studies report that wetland macrophytes uptake and accumulate metals in their tissue (Weiss et al., 2006; Basile et al., 2012; Matache et al., 2013) hence the potential to maintain the concentrations of these pollutants to levels way below those reported for coalmine water.

It is important to mention that DWS (2013) does not provide effluent discharge standards for SO_4^{2-} and Mg. Yet, presence of these parameters at optimal concentrations in treated water from WWTPs may have negative environmental and socio-economic implications. For instance, although most fish species can survive SO_4^{2-} concentration as high as 3378 mg/L, the effects of SO_4^{2} on human health is related to its transformation. SO_4^{2-} is an electron acceptor for anaerobic respiration. Thus, under anaerobic conditions; SO_4^{2-} is reduced to Sulphide (S^{2-}). The effects of S^{2-} include: i) exposure of humans to low concentrations of hydrogen sulphide may cause irritation to the eyes, nose or throat, ii) causes corrosion to metals such as iron, steel, Copper and Brass; and, iii) its presence in drinking water has been reported to cause nausea, illness and death in extreme cases (Oram, 2012). Furthermore, water decanting from waste coal dumps and discharging into surface waterways is usually “hard”, which is attributed to high concentration of dissolved minerals of which Magnesium (Mg) forms part. Much as hard water due to Mg^{2+} does not pose a health threat, it can be a nuisance due to build-up of minerals in water distribution pipes as well as causing wastage of detergents due poor scum formation (WHO, 2003), which results in increase in costs for water treatment. Therefore, due to the negative effects of presence of SO_4^{2-} and Mg, there is a need for the DWS to revise its water quality guidelines and regulations regarding discharge of treated water from WWTP.

Even though the HSSF CWs did not reduce SO_4^{2-} and metals from the feed water, SSF CWs are proposed as a passive option to dry land bioremediation in managing industrial solid wastes particularly from mining activities such as discard coal. The reason is that, the present methods of phyto-bioremediation of discard coal involve supplying the area under rehabilitation with large quantities of top soil, fertilizers and water to support the seedling, which is not environmentally and economically feasible. Besides, there is no evidence of degradation of the underlying coal discards except where microorganisms such as fungi are introduced to facilitate the degradation process (Sekhohola, 2016). With evidence from the proximate and elemental analysis results (Table 4-5), the present study confirms that the use of discard coal instead of gravel as treatment media in HSSF CWs has a biotechnological application in pollution control of coal discards. A change in ash content and fixed carbon were previously used as a measure of decomposition of materials in several biogeochemical studies (Clymo, 1984; Krüger et al., 2015). Ash content of coal refers to the unburnt residue remaining after complete combustion of a given coal sample while fixed carbon is the organic part of coal that remains after the volatile matter, ash, and moisture have been removed (Falcon and Ham, 1988; McKendry, 2002). The significantly higher percentage ash content values recorded from discard coal obtained from tap and AFP water fed discard coal-containing HSSF CWs at the end of the experiment than before introduction into the HSSF CWs therefore suggest decomposition of discard coal. Furthermore, the higher percentage fixed carbon composition recorded from discard coal before introduction into the HSSF CWs than discard coal from either tap or AFP water fed HSSF CWs at the end of the study period further suggest decomposition of discard coal (Engel et al., 2010). Elemental analysis also revealed lower percentage elemental composition of C, H, N & S from discard coal-containing HSSF CWs at the end of the experiment than before introduction into the HSSF CWs. These findings further suggest that degradation of discard coal occurred within the HSSF CWs. The lower C recorded at the end than at the start of the experiment suggests its use by organisms in many biogeochemical pathways to build-up biomass (Barnhart et al., 2016) while the lower S and N at the end than at the start of the experiment suggests their use as electron acceptors under anaerobic conditions (Tuttle et al., 1969), which is characteristic of subsurface flow CWs.

The use of discard coal to replace gravel as treatment media in HSSF CWs as described in the present study first of its kind. In summary, discard coal-containing HSSF CW fed with AFP water supported better growth of *P. australis* and produced better treated water quality than discard coal-containing CW fed with tap water while at the same time allowing decomposition of discard coal. This suggests that discard coal is a potential alternative treatment media to gravel for use in SSF CWs for municipal wastewater treatment. This application may be beneficial in reducing the capital costs for establishing SSF CWs for individual households or small communities by replacing gravel with discard coal especially in coal mining areas. Alternatively, in abandoned coalmine areas, discard coal dumps can be treated as CWs by planting with *P. australis* and irrigating plants with pre-treated wastewater from a nearby municipal WWTP to support plant growth. This application could allow decomposition of discard coal in the long run thereby resulting into its elimination from the environment.

While SSF CWs have proved efficient in treating municipal wastewater (Chapter 3 & 4), a recent study on evaluation of wastewater treatment systems in South Africa which comprise mainly of WSP system reports that less than 50 % of these systems meet regulatory national and international water quality standards for wastewater treatment (Mthembu et al., 2013). This is attributed to the ever-growing population in many towns due to migration resulting in overloading of these WWTPs, which therefore presents a need to upgrade these wastewater treatment systems to meet local discharge standards. Due to the large land area requirement of WSPs, an alternative wastewater treatment system is required to replace the oxidation pond system. However, the choice of a wastewater treatment system is an important consideration in determining the overall quality of treated water; and according to Tsagarakis et al. (2003), in most developing countries where financial resources are limited, the choice of an appropriate wastewater treatment system should be cost effective, environmentally sound and socially acceptable.

The work presented in this thesis (Chapter 5) therefore, further evaluate the use of V-H SSF hybrid CWs previously described (Chapters 2 & 3) and a series of HRAOPs to determine a system that could be suitable for selection to supplant the oxidation pond component of an overloaded and/or dysfunctional WSP system such as Bedford WWTP to restore functionality and or reduce overloading. The aspects considered included a comparison of the treated water quality from the two systems, land footprint requirement and cost of implementation.

Findings of the present study (Chapter 5) revealed that the quality of treated water from both the V-H SSF hybrid CWs and HRAOPs is comparable although with some discrepancies (Table 5-2). The factors that contributed to the failure of the V-H SSF hybrid CWs to meet the DWS authorization discharge limit for NH_4^+-N and COD were previously discussed in this chapter.

Furthermore, WSP systems are reported to produce unreliable treated water quality especially regarding nitrogen (US. EPA, 2002b), which is mainly due to stratification of the pond system and reduced floc formation therefore, resulting in reduced DO in deeper layer and small surface area for attachment of nitrifiers bacteria population (Zimmo et al., 2003; Babu, 2011). However, these problems are likely to be overcome in HRAOPs thus, the observed high-water quality regarding NH_4^+-N in the present study. In HRAOPs, the vertical mixing of water to full depth of the pond by the paddle wheel continuously exposes algae to light resulting in production of high concentration of algal biomass. The high algal biomass is important in several ways: i) increasing the attachment surfaces for nitrifying bacteria, ii) increasing production of oxygen to full depth of the pond, thereby facilitating pollutant removal processes that depend on DO concentration e.g. organic matter degradation and nitrification (Green et al., 1996). However, the unexpectedly low DO concentration recorded from HRAOPs effluent is probably due to its continuous utilization in organic matter degradation and nitrification processes. In fact, this study revealed significantly higher NO_3^--N + NO_2^--N concentration from HRAOPs than from WSP (Tebitendwa, 2011), which is due to utilisation of DO to convert NH_4^+ to NO_2^- and finally to NO_3^--N (nitrification), iii) high concentration of algal biomass in HRAOPs also enhances removal of nitrogen through algal uptake (Garcia et al., 2000); and, iv) unlike SSF CWs where nitrification and denitrification are limited by the low C/N ratio of wastewater, it is proposed that the high algal biomass produced in HRAOPs is also important for providing a carbon source thereby increasing nitrification.

Furthermore, besides the aerobic conditions that dominate HRAOPs and are necessary for organic matter degradation, the better water quality recorded from HRAOPs than from V-H SSF hybrid CWs regarding soluble COD could be due to continuous harvesting of sludge from the ASPs which limits its accumulation within the system. This contrasts with V-H SSF hybrid CWs where mineralisation of the accumulated sludge encourages build-up of soluble COD within the system and therefore limiting its removal.

The better water quality achieved for TSS from V-H SSF hybrid CWs than HRAOPs was expected. The SSF conditions limit light penetration thereby reducing algal growth in SSF CWs hence accounting for the better water quality observed for TSS from V-H SSF hybrid CWs than HRAOPs. However, the present study revealed that the TSS effluent concentration obtained from HRAOPs coincides with Oswald et al. (1994) who obtained high TSS concentration (163 mg/L) from St. Helena IAPS in California. Unlike in V-H SSF hybrid CWs, the open nature of pond systems allows light penetration that is necessary for algal growth hence accounting for the high TSS concentration reported in the present study (Mara, 2003).

As opposed to HRAOPs, the V-H SSF hybrid CWs reduced FC to acceptable standards for discharge into the environment (Table 5-2), which was due to improvement in mean water temperature (> 20 ^{o}C) during HP (Table 3-3) that facilitated removal of FC via various routes including UV degradation and predation among others that depend on temperature as previously discussed in this Chapter. Nevertheless, the failure of the HRAOPs to reduce FC to acceptable authorization discharge standards was unexpected since the high pH recorded during the present study (Table 5-2) was supposed to facilitate FC removal. According to Oswald (1991), algae in HRAOPs tend to raise the water pH and a pH of 9.2 for 24 hrs will allow almost 100 % disinfection of *E. coli*. Furthermore, the open nature of ponds would be expected to facilitate removal of FC through UV disinfection. However, the poor removal of FC from HRAOPs could be due to the high algae production of the system and the resulting TSS concentration in the effluent. A positive linear relationship was observed between FC and algae/TSS concentration from pond effluent which is due to attachment of FCs on algae in search for dissolved organic carbon (Kreuzinger Nobert, Vienna University of Technology, personal communication, July 2010).

Thus, while the HRAOPs at EBRU were designed and operated without a polishing stage, the poor water quality from this system regarding TSS and FC suggests a requirement for a tertiary treatment system to ensure that its effluent complies with the general authorization discharge standards even in regions experiencing air temperatures higher than 20 ^{o}C as may be the case for tropical regions. This contrasts with V-H SSF hybrid CWs whereby tertiary treatment may only be required to ensure compliance of the effluent with FC discharge standards only in regions with high fluctuation in air temperatures such as Grahamstown and with a treatment objective for water reuse as observed in the present study (Tables 3-4 & 3-6).

The remarkably higher capital costs of implementing a V-H SSF hybrid CW than a series of HRAOPs reported in this study is attributed to the cost of lining the system which accounts for 41.7 % of the total costs. Although cost evaluation reports on V-H SSF hybrid CWs are scarce, in another study, Mburu et al. (2013) reported similar findings; the cost of lining was one of the major capital cost for a SSF CW contributing about 42.2 %. Mburu et al. (2013) propose that the construction costs for SSF CWs may however, be lower in areas where clay soils exist that can be used for compacting instead of using a PVC liner. Furthermore, in coal mining areas where discard coal is considered as waste, this may be used as an alternative treatment media to gravel; thus, reducing capital costs for V-H SSF hybrid CW by 21.6 % (Table 5-4). Even though changing the treatment media from gravel to discard coal results in reduction of capital costs for the V-H SSF hybrid CW, the present study reveals that regardless of the treatment media, a series of HRAOPs require significantly lower capital than a V-H SSF CW for implementation (5-5).

As discussed earlier in this Chapter, the objectives of wastewater treatment are to produce an effluent that is suitable for discharge into the environment and reuse especially in agriculture. Therefore, where the treatment objective is protecting a surface water resource that is susceptible to eutrophication, the lower NH_4^+-N concentration produced from HRAOP effluent coupled with lower capital cost requirement than V-H SSF hybrid CWs suggests preference of HRAOPs over V-H SSF hybrid CWs to upgrade the oxidation pond component of an overloaded and defunct WSP especially where financial resources are limited. However, considering that South Africa is a water scarce country, the major objective of wastewater treatment could be production of sufficient water quality that can be reused for different purposes particularly irrigation of croplands. However, implementation of HRAOPs for irrigation purpose could be limited by the high TSS and FC concentrations recorded in the treated water (Table 5-2). The high TSS levels may result in blockage of the distribution system particularly where drip irrigation is applied while FC may threaten public health. On the other hand, as previously mentioned, evaluation of the performance of V-H SSF hybrid CWs revealed that this system does not meet the South African discharge standards for NH_4^+-N and COD concentration. While the NH_4^+-N and COD concentrations may not be suitable for discharge into a water resource, this water quality may be appropriate for reuse in agriculture due to the low TSS and FC levels. As mentioned earlier on in this chapter, the high NH_4^+-N concentration may be beneficial in reducing costs incurred in procurement of inorganic fertilizers while COD could be useful for adding organic matter to poorly developed soils. However, implementation of V-H SSF CWs as a secondary wastewater treatment system to upgrade a defunct WSP with an intention for water reuse may be limited by the high capital costs (Table 5-4 & 5-5) especially in developing countries where financial resources are limited. Thus, given the high polishing capacity of SSF CWs as evidenced from this study (Table 5-2) it is proposed that SSF CWs could be best utilized as tertiary rather than secondary wastewater treatment systems. This could particularly be applicable if SSF CWs are incorporated into the IAPS to polish HRAOP effluent to ensure compliance of its effluent with the general discharge standards especially regarding TSS and FC. SSF CWs could be a better choice to maturation ponds that are traditionally used as tertiary treatment units in the pond system that are faced with a problem of meeting TSS discharge limit (Okurut, 2000; Mburu et al., 2013).

In conclusion, the V-H SSF hybrid CW and HRAOPs produced comparable water quality with each system presenting its own constraints whereby the V-H SSF hybrid CWs failed to meet NH_4^+-N and COD while HRAOPs failed to comply with TSS and FC effluent discharge standards. Since the land footprint requirement of the two systems was comparable, the choice of the treatment system depends on availability of capital for implementation of the treatment system and treatment objective. Therefore, the lower NH_4^+-N concentration produced from HRAOPs coupled with lower capital cost requirement suggests preference of HRAOPs over V-H SSF CWs where financial resources are limited for upgrading a dysfunctional WSP system. However, a SSF CW is recommended as an appropriate tertiary treatment system to polish HRAOP effluent to ensure compliance discharge standards especially regarding TSS and FC.

As mentioned earlier on in this thesis, SSF CWs are a low-cost on-site wastewater treatment technology that have been successfully used to treat various types of wastewater types including municipal sewage. Nevertheless, the major limitation of these systems is their inability to meet the general authorization limit mainly for nitrogen, which is attributed to the design and operation of these systems mainly based on BOD as a design parameter. Therefore, the purpose of the work described in this PhD thesis set out to evaluate the performance of SSF CWs designed and operated using NH_4^+-N as a major parameter. Furthermore, the feasibility of changing the treatment media in HSSF CWs from gravel to discard coal was evaluated. Finally, the result of these studies was used to determine the most suitable technology for application in mitigating a defunct and/or overloaded traditional WSP sewage treatment plant, which was achieved by comparing the treated water quality, land footprint requirement and capital costs of a V-H SSF hybrid CW with a HRAOP series. Thus, following the discussion of results in this chapter, the following general conclusions are derived:

- Evaluation of the performance of the V-H SSF hybrid CWs design and operated using NH_4^+-N revealed that under optimal operating conditions of HLR, HRT and influent NH_4^+-N loading rate during the LP, the treated water achieved the South African General

Authorization standards for discharge into a water resource for all the monitored water quality parameters except COD and FC. Thus, the ability of the V-H SSF hybrid CWs to meet discharge standards particularly for nutrients (NH_4^+-N and PO_4^{3-}-P) suggest that NH_4^+-N is an important design and operational parameter for low loaded SSF CWs for discharge of treated water into the environment with a need of only simple disinfection methods such as chlorination to eliminate FC.

- An assessment of use of discard coal as an alternative support medium to gravel in HSSF CWs revealed that discard coal- containing HSSF CWs supported growth of *P. australis* in a similar way to gravel- containing HSSF CWs. Proximate analysis of discard coal obtained from HSSF CWs at the end of the experiment revealed an increase and a reduction in percentage ash content and fixed carbon respectively while elemental analysis showed an overall reduction in percentage elemental composition of discard coal but without compromising the water quality. These results demonstrate the suitability of discard coal for use as an alternative media to gravel in HSSF CWs to produce the acceptable water quality while at the same time allowing its degradation thus, suggesting the potential use of SSF CWs to control environmental pollution arising from both municipal wastewater and discard coal.
- Evaluation of the performance of V-H SSF hybrid CWs and a series of HRAOPs with a purpose to supplant the oxidation pond component of a dysfunctional and overloaded WSP system revealed that the treated water quality from both systems regarding most water quality parameters met the South African General Authorization standards for discharge into a surface water resource. However, V-H SSF hybrid CWs and HRAOP series showed difficulty in reducing COD and TSS to acceptable effluent discharge limits respectively. Nonetheless, each technology system revealed its own set of limitations including; failure to satisfactorily remove COD and TSS from V-H SSF hybrid CWs and HRAOPs respectively. Although the land footprint requirement PE of the two treatment systems was comparable, HRAOPs required significantly less capital than V-H SSF hybrid CWs for implementation. The latter proposes that HRAOPs could be preferred over V-H SSF hybrid CWs as a technology of choice to increase capacity of defunct and / or overloaded WSP sewage treatment plants particularly in regions where financial resources are limited such as in sub-Saharan Africa.
- Overall, the V-H SSF CWs designed and operated using NH_4^+-N achieved the acceptable water quality for discharge to the environment particularly for nutrient (N & P) while the

replacement of gravel as treatment media with discard coal in HSSF CWs permitted its degradation without compromising the treated water quality. Thus, these results indicate that where financial resources are readily available, NH_4^+-N is a potential design and operational parameter for SSF CWs to achieve the acceptable treated water quality and bioremediation of discard coal.

However, the following recommendations are proposed for future study:

- The study described in chapter 3 was carried out to determine the quality of treated water from a V-H SSF hybrid CWs designed using NH_4^+-N as a target parameter to ascertain if it suitable for discharge into the environment. The aquatic environment is however, a complex ecosystem with the interaction of various biotic and abiotic components including phyto- and zoo-plankton, fish of different species, benthic micro- and macro-invertebrates, bacteria, nutrients, air, sediment to mention a few. Although Svobodová et al. (1993) suggest that Cyprinids can tolerate an optimum COD in the range of 20-30 mg/L in ponds or rivers, which implies that the COD concentration of treated water obtained from V-H SSF hybrid CWs in the present study (Chapter 3) could be lethal to such fish. However, the eco-toxicological effect of COD on other aquatic microorganisms is poorly understood suggesting that such a study is highly recommended to determine the optimum COD concentration that may be tolerated not only to a wide variety of fish species but also other aquatic organisms such as algae, insects, crustaceans, molluscs etc. Such a study would help to establish the optimum COD discharge limit that can be allowed to WWTPs for discharge into a water resource with minimal damage to aquatic biota. Furthermore, the study described in chapter 3 employed *P. australis* due to its local availability in Grahamstown. However, there are various macrophytes available for use in CWs including (Chapter 1, Table 1-6). Therefore, future studies should focus on evaluating the influence of different wetland macrophytes on the performance of V-H SSF hybrid CWs designed using NH_4^+-N. This would help to select the best choice of macrophyte that could be employed in SSF CWs for wastewater treatment to meet the required effluent discharge standards. Although the design of CWs reported in literature (Chapter 1, Figure 1-8) does not incorporate a tertiary treatment unit, this study has confirmed that SSF CWs do not consistently achieve 100 % disinfection of FC. Therefore, future design for SSF CWs should include a tertiary treatment unit especially in regions with high variations in temperature to ensure complete elimination of FC.

- The feasibility of using discard coal as support media to replace gravel in HSSF CWs planted with *P. australis* was investigated in chapter 4. The following recommendations are proposed for future study: i) although *P. australis* was used in the present study due to its local availability in Grahamstown, as previously mentioned, there are various macrophytes available for use in CWs (Table 1-6). Therefore, further studies using pilot scale HSSF CWs are encouraged to evaluate the performance of discard coal- based HSSF CWs using various locally available wetland macrophytes depending on the climatic region. This would allow for selection of a better macrophyte that can be used in discard coal based CWs for wastewater treatment. ii) Wastewater and discard coal contain various parameters that are detrimental to the environment. However, this study only focused on the use of HSSF CWs to remove selected parameters from discard coal and water. Therefore, future studies should investigate the potential discard coal-containing HSSF CWs to remove pollutants from wastewater and discard coal through a full- spectrum analysis of the treated water from HSSF. iii) Microorganisms facilitate the decomposition process in natural treatment system such as HSSF CWs. However, there is no information regarding microorganisms associated with discard coal- based HSSF CWs. Therefore, future studies should focus on describing the microbial composition of discard coal-based HSSF CWs and influence on pollutant removal. iv) Future work should focus on determining the decay rate of discard coal within the SSF CW. This would help to determine the life span of HSSF CWs in which discard coal is used as treatment media. v) there are various industrial solid wastes resulting from mining activities as previously mentioned. While the presented study evaluated the use of discard coal as an alternative treatment media to gravel in HSSF CWs, the potential use of other industrial solid wastes such as gold, platinum etc. should be investigated.

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Appendices

1. Effluent discharge standards into surface water resources for selected countries in sub-Saharan Africa.

Parameter	General standards				
	South Africa[1]	**Uganda[2]**	**Zimbabwe[3]**	**Tanzania[4]**	**Kenya[5]**
pH	5.5-9.5	6.0-6.6	6.0-9.0	6.5-8.5	5.0-9.0
EC (mS/m)	70- 150	Ni	100	ni	ni
Temperature (°C)	ni	20-35	35	20-35	40
Turbidity (NTU)	ni	300	5	300	ni
Cl as free chlorine (mg/L)	0.25	1	ni	ni	ni
Cl^- (mg/L)	ni	500	ni	200	1500
Filtered COD mg/L)	75	100	60	60	120
BOD_5 (mg/L)	ni	50	30	30	40
NH_4^+-N (mg/L)	6	10	0.5	10	1
NO_3^--N+NO_2^--N (mg/L)	15	22	3	21	1
TN (mg/L)	ni	10	10	ni	10
TP (mg/L)	ni	10	ni	6	ni
PO_4^{3-}-P (mg/L)	10	5	0.5	6	1
TSS (mg/L)	25	100	25	100	35
Sulphates (mg/L)	ni	500	ni	500	1500
Alluminum (mg/L)	ni	0.5	ni	2.0	5
Lead (mg/L)	ni	0.1	0.05	0. 1	ni
Dissolved iron (mg/L)	0.3	10	1	5	2
FC (CFU/100 mL)	1000	1000	1000	1000	<200[a]

ni=not indicated, a=MPN per 100 mL, [1]= DWS (2013), [2]= NEMA-Uganda (1999), [3]= Nhapi & Gijzen (2002), [4]=NEMA-Tanzania (2007), [5]= NEMA (2003).

2. Parameters employed in calculating porosity of gravels.

Parameters	**Values**
π	3.14
r (cm)	4.75
H (cm)	10.5
W_1 (g)	460
W_2 (g)	780

3. Calculation of the gravel requirement of the hybrid constructed wetlands.

- Initially, a rectangular container was obtained and its volume (v) was calculated using the formula: V=lwh; where: V = volume of the rectangular container (m^3), l, w and h are the length, width and height of the container (m) respectively.
- The container was placed on a weighing scale, zeroing the scale, different gravel sizes were filled in the rectangular container and their weight recorded as summarized in the Table below.

Volume of container =0.335×0.24×0.09=0.0072 m^3

Gravel size (mm)	**up to 8 mm**	**14 mm**	**22-54 mm**
Mass (kg)	6.7	7.9	8.3

(a) The volume of the CWs to be filled by the different gravel sizes was calculated.

(a) HSSF CW gravel size requirement: 22-54 mm

The volume of gravel was calculated based on the following dimensions. The outlet and inlet zones were 0.2 ×1×0.4 m each. Therefore

- Inlet and outlet gravel for the HSSF CW 2(1*0.2*0.4=0.8 m^3) =0.16 m^3
- Total volume for the 2 HSSF CWs= 0.32 m^3

By extrapolation method, the weight of gravel that would be required to fill the HSSF CWs was calculated as shown below.

- If 0.0072 m^3 of the container is occupied by 8.3 kg of gravel, then 0.32 m^3 of the CW will be occupied by 368.8kg

Considering that 1400kg of gravel= 1m^3; therefore 368.8kg= ~0.26 m^3 of 22-54 mm gravel

- **VSSF Gravel requirement: 22-54 mm**

The volume of gravel was calculated based on the following dimensions. The bottom and surface layers occupied a depth of 0.15 and 0.05 m respectively (i.e. total depth=0.2 m); hence covering a volume of $1 \times 1 \times 0.2\ m^3$.

- Total volume for the 2 VSSF CW=0.4 m^3.

If 0.0072 m^3 of the container is occupied by 8.3 kg of gravel, then 0.4 m^3 of the CW will be occupied 461.1 kg.

Considering that 1400kg of gravel= 1m^3; therefore 461.1kg= ~0.33 m^3 of 22-54 mm gravel

Total 22-54 mm gravel= ~ 1 m^3

- **HSSF CW gravel size requirement: 14 mm**

The volume of gravel was calculated based on the following dimensions. The treatment zone was $1.6 \times 1 \times 0.4\ m^3$.

- Therefore, the total volume for the 2 VSSF CW=1.28 m^3.

▪ If 0.0072 m^3 of the container is occupied by 7.9 kg of gravel, then 1.28 m^3 of the CW will be occupied 1404.4 kg.

▪ Considering that 1400kg of gravel= 1m^3; therefore 1404.4 kg= ~1.00 m^3 of 14 mm gravel.

- **VSSF CW gravel size requirement: 14 mm**

The volume of gravel was calculated based on the following dimensions. The support zone was $0.05 \times 1 \times 1\ m^3$.

- Therefore, the total volume for the 2 VSSF CW=0.1 m^3.

If 0.0072 m^3 of the container is occupied by 7.9 kg of gravel, then 0.1 m^3 of the CW will be occupied 109.7 kg.

- Considering that 1400kg of gravel= 1m^3; therefore 109.7 kg= ~0.1 m^3 of 14 mm gravel

Total gravel of 14 mm requirement= 1.1 m^3. However, since it was impossible to order 1.1 m^3, 1.5 m^3 were ordered.

- **VSSF CW gravel size requirement: up to 8 mm**

The volume of gravel was calculated based on the following dimensions. The support zone was 0.45 ×1×1 m^3.

- Therefore, the total volume for the 2 VSSF CW=0.9 m^3.

If 0.0072 m^3 of the container is occupied by 6.7 kg of gravel, then 0.9 m^3 of the CW will be occupied 837.5 kg.

Considering that 1400kg of gravel= 1 m^3; therefore 837.5 kg= ~ 1 m^3 of 14 mm gravel.

N.B Since 33 % of the gravel is fine sand, **1.5 m^3** were ordered so that 1 m^3 of gravel is obtained after sieving.

4. Estimation of the V-H hybrid SSF CW area using NH_4^+-N as a design parameter.

Reeds et al. (1995) model was used to estimate the area for the SFF CW (Chapter 1, equation ii) using the DWS (2013) instead of 1998 effluent discharge limit.

A summary of parameters employed in the calculation is given below.

- A conservative design inflow rate (Q) of 0.1 m^3/d was considered.
- The average temperature of the coldest month (July), T_w was taken as 17 ^oC (Rhodes University, online metrological data, (2014).
- Influent NH_4^+-N concentration (C_{in}) of 12.1 mg/L i.e. average NH_4^+-N concentration from February 2015 to March 2016.
- Effluent NH_4^+-N concentration (C_{out}) of 6 mg/L was employed.
- A treatment depth y, of 0.4 was considered.

- The depth of the bed occupied by the root zone (rz) of 95 % (0.95) was assumed since roots of the macrophyte used (i.e. *P. australis*) can penetrate to a depth of about 0.4 m (Reeds et al., 1995).
- Porosity n, of treatment media of 0.43 was borrowed from Chapter 2.

Thus, by employing Reeds et al. (1995) model, a surface area of ~ **1 m²** was estimated.

5. Estimation of the V-H hybrid SSF CW area using NH_4^+-N as a design parameter.

Reeds et al. (1995) model was used to estimate the area for the SFF CW (Chapter 1, equation ii) using the DWS (2013) instead of 1998 effluent discharge limit.

A summary of parameters employed in the calculation is given below.

- A conservative design inflow rate (Q) of 0.17 m^3/d was considered.
- The average temperature of the coldest month (July), T_w was taken as 17 °C (Rhodes University, online metrological data, (2014).
- Influent NH_4^+-N concentration (C_{in}) of 21.7 mg/L i.e. average NH_4^+-N concentration during the initiation phase.
- Effluent NH_4^+-N concentration (C_{out}) of 6 mg/L was employed.
- A treatment depth y, of 0.4 was considered.
- The depth of the bed occupied by the root zone (rz) of 95 % (0.95) was assumed since roots of the macrophyte used (i.e. *P. australis*) can penetrate to a depth of about 0.4 m (Reeds et al., 1995).
- Porosity n, of treatment media of 0.43 was borrowed from Chapter 2.

Thus, by employing Reeds et al. (1995) model, a surface area of ~ **4 m²** was estimated

6. Estimation of the V-H hybrid SSF CW area requirement during the HP using NH_4^+-N as a design parameter considering the DWS (2013) NH_4^+-N discharge limit using Reeds et al. (1995) model.

A summary of parameters employed in the calculation is given below.

- Inflow rate (Q) of 0.1 m^3/d was considered i.e. mean inflow rate during HP
- The average temperature of the coldest month (July), T_w was taken as 17 °C (Rhodes University, online metrological data, (2014).
- Influent NH_4^+-N concentration (C_{in}) of 30.5 mg/L (adopted from the high NH_4^+-N LP).
- Effluent NH_4^+-N concentration (C_{out}) of 6 mg/L was employed.
- A treatment depth y, of 0.4 was considered.
- Porosity n, of treatment media of 0.43 was borrowed from Chapter 2.
- 95 % was assumed for rz (the depth occupied by the roots in the CW since roots of the macrophyte (i.e. *P. australis*) can penetrate to a depth of about 0.4 m (Reeds et al., 1995).

Thus, by employing Reeds et al. (1995) model, a surface area of ~ **3 m^2** was estimated.

7. Monthly meteorological data of Grahamstown recorded during the study period (February 2015-July 2016).

Sampling date	Air Temperature	Rainfall	Humidity	Wind speed	Estimated ET
Initiation phase					
Feb. 2015	19	1.8	98	11	48.5
Mar. 2015	19	1.7	78	14	23.4
Apr. 2015	15	2.5	77	13	5.8
May. 2015	15	0.1	68	11	23.4
Jun. 2015	12	2.3	65	15	15.6
Jul. 2015	11	0	75	0	10
Optimization phase					
High NH_4^+-N LP					
Oct. 2015	17	0.4	74	14	17.07
Nov. 2015	10	2.4	72	14	19.07
Dec. 2015	20	0	71	14	10
Jan. 2016	21	0.4	74	14	
Feb. 2016	15	1.4	73	13	38.06
Mar. 2016	19	1.8	74	13	25.13

Sampling date	Air Temperature	Rainfall	Humidity	Wind speed	Estimated ET
low NH_4^+-N LP					
Apr. 2016	18	0.8	67	14	29.14
May. 2016	16	2.1	66	14	28.77
Jun. 2016	14	0.2	58	15	25.2
Jul. 2016	14	1.6	52	16	16.6

The units for the parameters are air temperature (°C), rainfall and ET (mm), humidity (%) and wind speed (Km/hr).

8. Influent-effluent Physico-chemical water quality recorded from the hybrid CWs during the study period (February 2015-July 2016).

	Influent water quality				Effluent water quality			
Sampling date	Temp	DO	pH	EC	Temp	DO	pH	EC
Initiation phase								
10.Feb.2015	25.8	6	7.7	111	21.4	2.9	7.4	110
04.Mar.2015	23.8	4.2	6.8	128	21.5	2.3	6.6	125
01.Apr.2015	20		6.9	130	21.6		6.8	126
12.May.2015	18.5		7.4	125	18.1		7.2	112
24.Jun.2015	12.6		7.2	124	11.5		6.6	102
22.Jul.2015	11.9		6.8	126	10.6		7.4	104
Optimization phase								
High NH_4^+-N LP								
05.Oct.2015	19.5	2.5	7.4	120.0	18.1	1.6	7.2	126.0
06. Nov.2015	20.7	2.1	7.4	113.0	19.0	2.0	6.9	112.0
07.Dec.2015	21.3	2.2	7.0	123	24.7	2.9	6.5	119
06. Feb.2016	23.5	3.9	7.2	113	23.1	2.1	7.2	112
02. Mar.2016	23	3.3	7.2	110	22.8	2.6	7.3	142
low NH_4^+-N LP								
05. Apr.2016	17.7	4.3	7.4	113	17.4	2.5	7.2	152
21.Apr.2016	22.1	2	7.8	109	20.2	2.6	7.3	118
04. May.2016	19.7	4.7	7.8	120	19.2	3.5	7.5	141
24.May.2016	20.5	3.8	7.2	113	19.5	3	7.1	140
01.Jun.2016	19.9	3.6	7.5	114	19.2	2.7	7.3	120
21.Jun.2016	16.7		7.6	85	13.6		7.2	103
07.Jul.2016	12.6	3.4	7.2	82	12.7	3.5	7.3	104
21.Jul.2016	13.3	3	7.3	87	12.9	2.8	7.2	101

The units for the parameters are: water temperature (°C), DO (mg/L), pH (No unit) and EC (mS/m)

9. Month data for the influent-effluent water quality and performance of the hybrid CWs during the study period February 2015-July 2016.

Parameter	Sampling period	Cin	Qi	Lin	Co	Qo	Lo	R R	% R
TSS	**Initiation phase**								
	10. Feb.2015	143	0.1	8.10	13.00	0.0	0.1	7.97	98.4
	04. Mar. 2015	229	0.1	12.98	25.00	0.0	0.7	12.23	94.2
	01. Apr. 2015	95	0.1	5.70	6.00	0.1	0.3	5.36	94.0
	12. May. 2015	93	0.1	5.58	13.00	0.1	0.4	5.10	91.5
	24. Jun. 2015	86	0.1	5.45	10.00	0.1	0.5	4.95	90.8
	22. Jul. 2015	75	0.1	2.75	16.70	0.0	0.4	2.30	83.8
	Optimization phase								
	High NH_4^+-N LP								
	05. Oct. 2015	150	0.1	5.00	8.30	0.0	0.1	4.83	96.7
	06. Nov. 2015	50	0.1	1.67	5.00	0.0	0.1	1.57	94.0
	07. Dec. 2015	135	0.1	4.50	7.50	0.0	0.2	4.30	95.6
	Jan.2016								
	06. Feb. 2016	150	0.1	6.50	10.00	0.0	0.0	6.43	99.0
	02. Mar. 2016	60	0.1	2.40	5.00	0.0	0.0	2.32	96.5
	low NH_4^+-N LP								
	05. Apr. 2016	47	0.1	1.72	7	0.0	0.0	1.68	97.3
	21. Apr. 2016	60	0.1	2.20	7	0.0	0.0	2.13	96.8
	04. May. 2016	70	0.1	2.57	5	0.0	0.0	2.52	98.1
	24. May. 2016	50	0.1	1.83	7	0.0	0.0	1.76	96.2
	01. Jun. 2016	40	0.1	1.47	7	0.0	0.0	1.44	98.4
	21. Jun. 2016	120	0.1	4.40	7	0.0	0.1	4.26	96.8
	07. Jul. 2016	80	0.1	3.47	8	0.0	0.1	3.28	94.6
	21. Jul. 2016	97	0.1	3.23	0	0.0	0	3.23	100.0
Soluble COD	**Initiation phase**								

Parameter	Sampling period	Cin	Qi	Lin	Co	Qo	Lo	R R	% R
	10. Feb. 2015	64	0.1	3.63	25.00	0.0	0.2	3.38	93.1
	04. Mar. 2015	64	0.1	3.63	33.00	0.0	0.9	2.64	72.7
	01. Apr.2015	124	0.1	7.44	49.00	0.1	2.7	4.66	62.7
	12. May. 2015	63	0.1	3.78	56.00	0.1	2.0	1.73	45.7
	24. Jun. 2015	117	0.1	7.41	67.00	0.1	3.3	4.06	54.8
	22. Jul. 2015	118	0.1	4.33	99.00	0.0	2.6	1.69	39.0
	Optimization phase								
	High NH_4^+-N LP								
	05. Oct. 2015	149	0.1	4.97	108.0	0.0	2.1	2.81	56.5
	06. Nov. 2015	122	0.1	4.07	102.0	0.0	2.0	2.03	49.8
	07. Dec. 2015	208	0.1	6.93	88.60	0.0	2.3	4.57	65.9
	Jan.2016								
	06. Feb.2016	184	0.1	7.97	95.00	0.0	0.6	7.34	92.1
	02. Mar.2016	109.5	0.1	4.38	132.1	0.0	2.2	2.18	49.7
	low NH_4^+-N LP								
Soluble COD	05. Apr. 2016	154	0.1	5.65	140	0.0	0.9	4.71	83.5
	21. Apr. 2016	317	0.1	11.62	260	0.0	2.6	9.02	77.6
	04. May. 2016	142	0.1	5.21	128	0.0	1.2	3.93	75.4
	24. May. 2016	139	0.1	5.10	94	0.0	0.9	4.16	81.6
	01. Jun.2016	138	0.1	5.06	92	0.0	0.3	4.75	93.9
	21. Jun. 2016	83	0.1	3.04	129	0.0	2.5	0.46	15.2
	07. Jul. 2016	76	0.1	3.29	76.4	0.0	1.7	1.51	45.9
	21. Jul. 2016	111	0.1	3.70	83.00	0.0	1.9	1.76	47.7
PO_4^{3-}-P	**Initiation phase**								
	10. Feb. 2015	4.9	0.1	0.28	3.9	0.0	0.0	0.24	86.0
	04. Mar. 2015	3.8	0.1	0.22	3.9	0.0	0.1	0.10	45.7

Parameter	Sampling period	Cin	Qi	Lin	Co	Qo	Lo	R R	% R
	01. Apr. 2015	3.6	0.1	0.22	7	0.1	0.4	-0.18	-83.6
	12. May. 2015	7.7	0.1	0.46	7.7	0.1	0.2	0.18	38.9
	24. Jun. 2015	5.4	0.1	0.34	8.7	0.1	0.4	-0.09	-27.2
	22. Jul. 2015	5	0.1	0.18	11.2	0.0	0.3	-0.12	-62.9
	Optimization phase								
	High NH_4^+-N LP								
	05. Oct. 2015	24.6	0.1	0.90	7.6	0.0	0.1	0.75	83.1
	06. Nov. 2015	7.8	0.1	0.29	12.7	0.0	0.2	0.03	11.2
	07. Dec. 2015	5.8	0.1	0.21	7.7	0.0	0.2	0.01	3.4
	Jan.2016								
	06. Feb. 2016	6.4	0.1	0.28	4.7	0.0	0.0	0.25	88.7
	02. Mar. 2016	7.1	0.1	0.28	13.2	0.0	0.2	0.06	22.5
	low NH_4^+-N LP								
	05. Apr. 2016	1	0.1	0.04	10.5	0.0	0.0	-0.03	-90.9
	21. Apr. 2016	4.1	0.1	0.15	3.4	0.0	0.0	0.12	77.4
	04. May. 2016	3.2	0.1	0.12	3.6	0.0	0.0	0.08	69.3
	24. May. 2016	2.8	0.1	0.10	3.4	0.0	0.0	0.07	66.9
	01. Jun.2016	3	0.1	0.11	3.4	0.0	0.0	0.10	89.7
	21. Jun. 2016	3.1	0.1	0.11	1.1	0.0	0.0	0.09	80.6
	07. Jul. 2016	1.5	0.1	0.07	1.5	0.0	0.0	0.03	46.2
	21. Jul. 2016	4.3	0.1	0.14	3.3	0.0	0.0	0.07	46.3
NH_4^+-N	**Initiation phase**								
	23. Feb. 2015	10.5	0.1	0.60	2.50	0.0	0.0	0.57	95.8
	24. Mar. 2015	11.7	0.1	0.66	8.70	0.0	0.2	0.40	60.6
	22. Apr. 2015	13.2	0.1	0.79	9.00	0.1	0.5	0.28	35.6
	21. May. 2015	15.0	0.1	0.90	13.50	0.1	0.5	0.41	45.0

Parameter	Sampling period	Cin	Qi	Lin	Co	Qo	Lo	R R	% R
	24. Jun. 2015	43.5	0.1	2.76	16.00	0.1	0.8	1.96	71.0
	22. Jul. 2015	36	0.1	1.32	21.60	0.0	0.5	0.74	56.4
	Optimization phase								
	High NH_4^+-N LP								
	05. Oct. 2015	36.0	0.1	1.20	22.40	0.0	0.4	0.75	62.7
	06. Nov. 2015	40.7	0.1	1.36	30.00	0.0	0.6	0.76	55.8
	07. Dec. 2015	29.4	0.1	0.98	8.50	0.0	0.2	0.75	76.9
	Jan.2016								
	06. Feb. 2016	24.6	0.1	1.07	7.50	0.0	0.0	1.02	95.3
	02. Mar. 2016	21.6	0.1	0.86	11.20	0.0	0.1	0.68	78.4
	low NH_4^+-N LP								
	05. Apr. 2016	12.8	0.1	0.47	7.4	0.0	0.0	0.42	89.5
	21. Apr. 2016	15.5	0.1	0.57	1.7	0.0	0.0	0.55	97.0
	04. May. 2016	11.4	0.1	0.42	3.0	0.0	0.0	0.39	92.8
	24. May. 2016	18.0	0.1	0.66	2.8	0.0	0.0	0.63	95.8
	01. Jun. 2016	16	0.1	0.59	2.1	0.0	0.0	0.58	98.8
	21. Jun. 2016	13.3	0.1	0.49	2.0	0.0	0.0	0.45	91.8
	07. Jul. 2016	12.5	0.1	0.54	1.9	0.0	0.0	0.50	91.8
	21. Jul. 2016	19	0.1	0.63	3	0.0	0.0	0.56	88.9
NO_2^--N+NO_3^--N	**Initiation phase**								
	10. Feb. 2015	0.4	0.1	0.02	0.7	0.0	0.0	0.02	69.1
	04. Mar. 2015	0.3	0.1	0.02	0.3	0.0	0.0	0.01	47.1
	01. Apr. 2015	0.4	0.1	0.02	0.1	0.1	0.0	0.02	76.4
	12. May. 2015	0.2	0.1	0.01	0.2	0.1	0.0	0.00	38.9
	24. Jun. 2015	0.3	0.1	0.02	0.1	0.1	0.0	0.01	73.7

Parameter	Sampling period	Cin	Qi	Lin	Co	Qo	Lo	R R	% R
	22. Jul. 2015	0.1	0.1	0.00	0.2	0.0	0.0	0.00	-45.5
	Optimization phase								
	High NH_4^+-N LP								
	05. Oct. 2015	0.2	0.1	0.01	0.2	0.0	0.0	0.00	45.5
	06. Nov. 2015	0.4	0.1	0.01	0.2	0.0	0.0	0.01	72.7
	07. Dec. 2015	0.3	0.1	0.01	0.2	0.0	0.0	0.01	51.5
	Jan.2016								
	06. Feb. 2016	0.3	0.1	0.01	0.3	0.0	0.0	0.01	84.6
	02. Mar. 2016								
	low NH_4^+-N LP								
	05. Apr. 2016	0.1	0.1	0.00	0.1	0.0	0.0	0.00	81.8
	21. Apr. 2016	0.1	0.1	0.00	0.2	0.0	0.0	0.00	45.5
	04. May. 2016	0.3	0.1	0.01	0.1	0.0	0.0	0.01	90.9
	24. May. 2016	2.2	0.1	0.08	0.1	0.0	0.0	0.08	98.8
	01. Jun. 2016	0.3	0.1	0.01	2.7	0.0	0.0	0.00	18.2
	21. Jun. 2016	0.1	0.1	0.00	0.1	0.0	0.0	0.00	45.5
	07. Jul. 2016	0.2	0.1	0.01	0.2	0.0	0.0	0.00	46.2
	21. Jul. 2016	0.1	0.1	0.00	0.3	0.0	0.0	0.00	-110.0
E. coli	**Initiation phase**								
	10. Feb. 2015	4E+0	0.1	2266	0	0.0	0.0	22667	100
	04. Mar. 2015	3E+0	0.1	1700	0	0.0	0.0	17000	100
	01. Apr.2015	4E+0	0.1	2400	1500	0.1	85.	23915	99.6
	12. May. 2015	9E+0	0.1	5400	2250	0.1	83.	53918	99.8
	24. Jun. 2015	6E+0	0.1	3800	1500	0.1	75.	37925	99.8
	22. Jul. 2015	6E+0	0.1	2200	3500	0.0	93.	21907	99.6
	Optimization phase								
	High NH_4^+-N LP								

Parameter	Sampling period	Cin	Qi	Lin	Co	Qo	Lo	R R	% R
	05. Oct. 2015	3E+0	0.1	1000	500	0.0	10	9990	99.9
	06. Nov. 2015	1E+0	0.1	3333.	0	0.0	0.0	3333.3	100
	07. Dec. 2015	7E+0	0.1	2333	0	0.0	0.0	23333	100
	Jan. 2016								
	06. Feb. 2016	5E+0	0.1	1950	0	0.0	0.0	19500	100
	02. Mar. 2016	6E+0	0.1	2400	0	0.0	0.0	24000	100
	low NH_4^+-N LP								
E. coli	21. Apr. 2016	3E+0	0.1	1100	0	0.0	0	11000	100
	24. May. 2016	4E+0	0.1	1466	1000	0.0	10	14657	99.932
	21. June. 2016	5E+0	0.1	1833	2000	0.0	27.	18307	99.855
	21. Jul. 2016	7E+0	0.1	2800	3000	0.0	70	27930	99.75

Cin= influent concentration (mg/L), Qin= inflow rate (m^3/d), Lin= influent loading rate (g/m^2/d), Co=Effluent concentration (mg/L), Qo= outflow rate (m^3/d), Lo= Effluent loading rate (g/m^2/d).

10. Summary of raw data for the NO_3^--N + NO_2^--N concentration (mg/L) from the different stages of the hybrid constructed wetlands.

Sampling date	**VSSF influent**	**VSSF effluent**	**HSSF effluent**
Initiation phase			
10.Feb.2015	0.3	1.2	0.7
04.Mar.2015	0.4	0.7	0.3
01.Apr.2015	0.2	2.5	0.1
12.May.2015	0.3	2.4	0.2
24.June.2015	0.1	9.5	0.1
22.July.2015	0.2	3.2	0.2
Phase 2			
Optimization phase			
High NH_4^+-N LP			
05.Oct.2015	0.2	2.6	0.2
06.Nov.2015	0.4	12.4	0.2
07.Dec.2015	0.3	9.3	0.2
Jan.2016			
06.Feb.2016	0.3	9.1	0.3
02.Mar. 2016			
Low NH_4^+-N LP			
05.Apr.2016	0.1	7.8	0.1
21.Apr.2016	0.1	1.4	0.2
04.May.2016	0.3	10.3	0.1
24.May.2016	2.2	0.1	0.1
01.Jun.2016	0.3	0.2	2.7
21.Jun.2016	0.1	0.9	0
07.Jul.2016	0.2	0.2	0.2
21.Jul.2016	0.1	0.2	0.3

11. Cost estimates implementation of a 1750 m^3/d treatment capacity AIPS at Bedford wastewater treatment plant.

Item	cost (US $)
Direct costs:	
1. P & G	138813.57
2. inlet works	3571.68
3. Anearobic Ponds (2 No.)	75716.4
4. Primary facultative pond (1 No.)	54228
5. High rate algal ponds incl. mechanical/electrical installations (5 No.)	631455.48
7. Algal settling ponds (2 No.)	45471.84
8. Effluent irrigation works	107780.4
9. General	7200
10. Contingencies (10 %)	106423.74
Sub-Total direct costs	**1170661.1**
VAT	163892.55
Total for direct costs	1334553.7
Indirect costs:	
10. Professional fees	117066.11
11. Disbursements	87799.583
12. Sampling and testing	2400
13. Flood-line determination	4200
14. Survey	3000
15. Geotechnical investigation	6600
16. EIA	24000
17. Licensing	5400
18. Health and safety	14400
Sub-total indirect costs	**264865.69**
VAT	37081.198
Total for indirect costs	301946.89
Total for direct and indirect costs	**1636500.6**

Additional notes:

The highlighted fields in the table above were extrapolated to estimate the cost of HRAOPs and ASPs treating wastewater of 500 PE (75 m^3/d) IAPS at EBRU as follows.

- The 5 HRAOPs at BWWTP were designed with a total treatment volume of 10296 m^3 and these would cost US $. 631255.48. Therefore, the 3 HRAOPs at EBRU designed with a total treatment volume of 450 m^3 would cost US $. 27598.4.
- The 2 ASPs at BWWTP were designed with a total treatment volume of 400 m^3 and these would cost US $. 45471.84. Therefore, the 3 ASPs at EBRU with a total treatment volume of 57 m^3 would cost US $. 6479.7

12. Summary of the capital costs for V-H SSF hybrid CWs described in Chapter 2.

Description	**Unit**	**Qty**	**Rate (US $)**	**Cost (US $)**
Foundation excavation	m^3	0.075	1.5	0.1
Foundation concrete (20 Mpa)	m^3	0.075	65.4	4.9
Cement stabilised base (75 mm)	m^2	1.58	1.4	2.2
Concrete blocks	pc	104	0.9	93.6
Cost of gravel (incl. transport to the site)	m^3	2	46.5	93
Plastic lining	m^2	20.2	8.6	173.7
Macrophyte collection & planting	hr	1.5	1.5	2.3
Total cost				**369.8**

13. Calculation of the constructed wetland hydraulics

- **Hydraulic residence time (HRT)**

The theoretical hydraulic retention time of the SSF CW was calculated based on the following equation (Tousignant et al., 1999):

t (days) = LWny /Q

Where: L = Length of system -parallel to flow direction (m), W = Width of system (m), n = porosity of the bed, d = depth of submergence (m) and Q= Average flow through the system (m^3/d).

- **Hydraulic loading rate (HLR)**

HLR (mm/d) = Q/A×1000

Where Q= inflow rate (m^3/d) and A= surface area (m^2).

14. Determination of the elemental composition of gravel and discard coal.

Triplicate samples of gravel and discard coal of the same size (1 g) were placed on a stamp, stacked with a double-sided carbon tape, placed under a scanning electron microscope (SEM) (VEGA TESCAN Oxyford instrument) installed with Vega Tc and INCAPenta FETx3 softwares. The softwares generated peaks (i.e. yellow peaks shown in the figure below); and these were translated into the elemental composition of each sample.

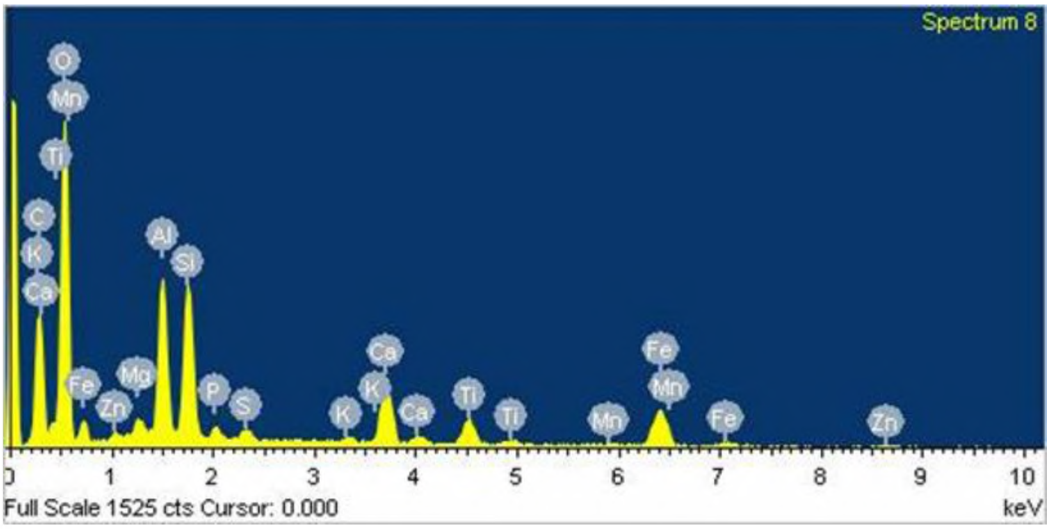

Figure I. Elemental composition of discard coal generated using a SEM.