A PILOT STUDY ON MUNICIPAL WASTEWATER TREATMENT USING A CONSTRUCTED WETLAND IN UGANDA

A Pilot Study on Municipal Wastewater Treatment Using a Constructed Wetland in Uganda

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Contents

Acknowledgements IX

Abstract XI

Chapter 1 General Introduction..1

Chapter 2 The Kirinya Constructed Wetlands: Structure and Hydrology.........17

Chapter 3 Organic Matter and Suspended Solids Removal..........................45

Chapter 4 Removal of Nutrients...71

Chapter 5 Distribution and Removal of Faecal Coliforms.........................107

Chapter 6 Performance Evaluation of a Single Household Constructed Wetland..125

Chapter 7 Economic Evaluation of Constructed Wetland Potential in Uganda..143

Chapter 8 Summary and Conclusions..162

Curriculum Vitae...171

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Abstract

The potential of using constructed wetlands as a cheaper and yet effective alternative method for treating domestic wastewater in tropical environments was investigated in this study from May 1996 - April 1999. The major aim was to determine their technical viability with respect to treatment performance under different operating conditions and the economic competitiveness of the technology in Uganda and within the region. A pilot constructed wetland design, based on horizontal flow criteria and receiving pre-settled sewage from the Jinja Kirinya Sewage anaerobic lagoons was used in the study. The wetland had a total surface area of 320 m^2, which was sub-divided into eight individual units. Four of the units were planted with *Cyperus papyrus* floating without a substratum base and two with *Phragmites mauritianus* anchored on a substratum base. Two units were used as controls, one of them had a substratum base of similar volume as the planted ones. The wetlands were operated over three consecutive phases. In the initial phase, all planted units remained intact but in the 2nd phase, plant biomass was removed from a quarter of the area of the two *Cyperus papyrus* and one *Phragmites mauritianus* wetland units. In the last phase, two wetland units of each plant type and one control (open) were joined in series; two other papyrus wetland units with smaller areas of alternating planted and unplanted sections were joined in series. The hydraulic loading rates applied to the different units ranged between 1.3 - 12 cm day^{-1} over the whole experimental period.

The shoot density and size of open areas controlled the physical–chemical conditions in the wetland units. When the shoot density was greater than 50 shoots/m^2, low oxygen concentrations < 2 mg/l, pH values of 7 – 7.5 and water temperatures of 22 ± 0.5 °C prevailed. In the open ponds with relatively larger open areas, peak oxygen concentration of 27 mg/l, pH of 10.4 and water temperature 29 °C were obtained. In the reduced open surface areas applied in the 2nd and 3rd phases, peak oxygen concentration of 12 mg/l, pH values of 7.5–8 and temperature of 23 °C were registered. These conditions influenced the extent of reduction of the pollutants achieved in each case. Removal efficiencies in both *C. papyrus* and *P. mauritianus* wetland units' of just over 70% of the input settled COD (maximum 350 kg ha^{-1}) and settled BOD (maximum 100 kg ha^{-1}) were obtained when the shoot cover was intense. It increased to over 80% in the third phase. TSS reduction above 80% of the input (maximum 250 kg ha^{-1}) was obtained in all the vegetated wetland units and in all the phases. Similar findings were observed in the household wetland. A significant faecal coliform removal of 4 log units was obtained in the control ponds as compared to 3 log units derived from the vegetated wetland units or 1.13 log units from the household wetland. The faecal coliform reduction in the planted units was correlated with TSS and particulate organic matter removal. The lethal impact of direct sunlight and its secondary effects such as high pH were considered more significant in ponds with open areas. All the results show that *Cyperus*

papyrus planted units performed better than *Phragmites mauritianus* units. The low residual background BOD concentration of 12 mg/l in papyrus units as compared to 17 mg/l in *Phragmites* units further confirmed the observed trends. In addition, the rate of BOD decay in papyrus units was higher as indicated by the areal first order rate constant (0.084 m d^{-1}) than in *Phragmites* (0.039 m d^{-1}). Throughout, the effluent concentrations of these parameters from the vegetated wetland units were consistently below the Uganda wastewater discharge standards.

Nutrient (N & P) removal from the wastewater via plant uptake showed extreme variability at different growth phases; uptake was correlated with the biomass yields exhibited in the different phases. Uptake rates of 7.1 kg N ha^{-1} day^{-1} and 0.24 kg P ha^{-1} day^{-1} in papyrus and 10.4 kg N ha^{-1} day^{-1} and 0.26 kg P ha^{-1} day^{-1} in *P. mauritianus* were derived in the exponential growth. Mass balance considerations over exponential phase showed plant uptake contribution of 15% N and 10 % P of the total input to papyrus wetland and 58% N and 37% P of the total input to *P. mauritianus* units. This contribution declined to less than 4% for both plant types at the slower growth phase. It is concluded that nitrogen and phosphorus removal via plant uptake is only significant at the exponential growth phase and more so in *P. mauritianus* which had nearly 90% of its total biomass as above ground. Nitrogen removal by other routes was influenced by the environmental conditions. Low oxygen concentrations which, were prevalent when the shoot density was high, minimised ammonium removal via the sequential nitrification-denitrification pathway. The pH (>10) and temperature (max 29 ^{0} C) characterising the open units favoured ammonium loss by volatilisation of ammonia gas. In the papyrus wetland series with small alternating vegetated and non-vegetated zones applied in the third phase, up to 90% of total ammonium N input (26 kg N ha^{-1}) was removed, 77% of this loss was attributed to nitrification- denitrification process. Effluent with ammonium concentration below the Uganda regulatory discharge limit of 10 mg/l was only obtained in the last phase. The wetland serial configuration applied in this phase is concluded to be the most suitable for nitrogen removal from wastewater in the tropical environments.

The economic viability of using constructed wetlands in Uganda was deduced from the total annual costs of the wetland and the waste stabilisation ponds designed for a population equivalent of 4000. The total annual cost for waste stabilisation ponds was 21 % more than that of the constructed wetlands (US $ 11,400). The recurrent cost for both systems were similar but nearly eight times lower than that estimated for conventional treatment systems.

Based on the overall results of the treatment performance and economic costs, it is concluded that application of constructed wetlands in Uganda and in the sub-region can be considered both technically and economically as a viable option for municipal wastewater treatment.

Chapter 1

General Introduction

1.1 Background 3

1.2 Wetland Wastewater Treatment Technology 4
1.2.1 Natural wetlands 4
1.2.2 Constructed wetlands 5

1.3 Aquatic Plants 7
1.3.1 *Cyperus papyrus* L 8
1.3.2 *Phragmites mauritianus* (Kunth) 9

1.4 Hydrological Factors 10

1.5 Kirinya wetland systems 10

1.6 Research scope and relevance of this thesis 11

1.7 Objectives of this thesis 11

1.8 Outline of the thesis 12

References 13

1.1 Background

Sewage collection and subsequent treatment has been practised to varying extents for over one hundred years in Europe. Initial emphasis was on draining the cities of contaminated and foul smelling sewage but shifted to reduction of organic matter. However, from the early 1960's, the negative environmental impacts that were linked to wastewater discharges led to change in the attitude and levels of treating wastewater. Expectations of wastewater treatment operations were expanded to include the removal of nutrients (nitrogen and phosphorus containing compounds). Wastewater with high concentrations of these nutrients, when discharged into water bodies such as lakes, rivers and canals causes oxygen depletion and alterations in their trophic status above the natural state. These changes are known to have the capacity to trigger off processes that are detrimental to the water thereby limiting the uses for which the water is suitable (Golterman, 1975; Chapman, 1996). Regulating authorities in Uganda and in many other countries have set up effluent discharge limits for these nutrients just like for other pollutants.

Many developing countries are presently experiencing rapid population and economic growth especially in the urban centres. The provision of services, including wastewater collection, treatment and disposal has however not kept pace with these developments. The principle constraint is lack of financial, technical and institutional resources. Besides, lack of interest in alternative wastewater treatment and disposal methods other than waterborne sewage, has also been identified as one of the limiting factors in the service delivery (Kalbermatten *et al.*, 1982). The appraisal given by the World Bank (1992), showed that in the majority of the urban centres in the developing countries, waterborne sewerage services were not available. In a few of the centres where this centralised collection and treatment systems existed, the prohibitive operating and maintenance costs limited their satisfactory utilisation.

Investment in sanitation services as compared to water supply is also not given the priority it deserves by the national governments in the developing countries. Briscoe (1993) showed that even in the World Bank financed water and sewerage projects, wastewater collection and treatment components accounted for only a fifth of the budgets. In contrast, in the developed countries, because of their strong economies and technical expertise, off-site wastewater disposal has been given priority attention at similar levels with water supply.

During the United Nations drinking water and sanitation decade (1980 - 1990), significant improvements in the infrastructure for delivery of safe water and facilities for sanitary disposal and treatment of wastewater together with planning methods were registered in the developing countries (Christmas and de Rooy, 1991) (Fig. 1.1). However, despite these achievements, sanitation services still remain insufficient. Sanitary conditions, especially in the slums and peripheral urban centres which lack planned infrastructure, are appalling. Outbreaks of waterborne diseases such as diarrhea, cholera and dysentery still occur in these locations which often are also epicentres of population growths. Provision of waterborne sewerage services for these locations cannot be realised given the present economic circumstances in these countries. Lower cost solutions for both the collection and treatment of wastewater which answer the needs of the majority are required.

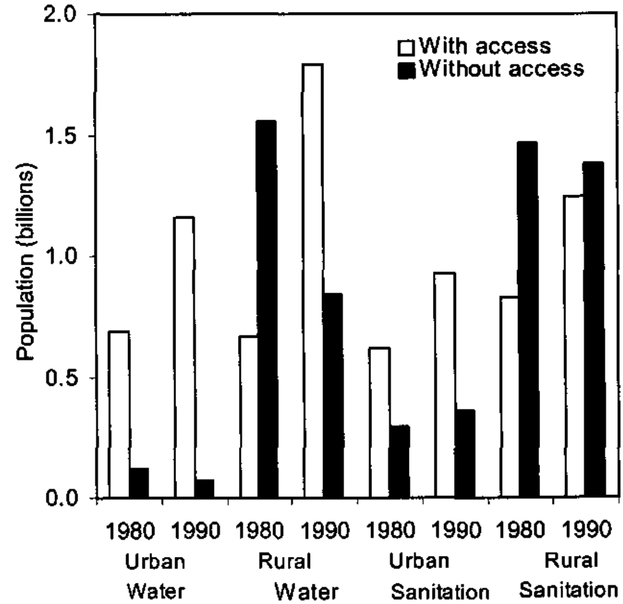

Fig. 1.1 Access to safe water and adequate sanitation in developing countries in 1980 and 1990 (from Christmas and de Rooy, 1991).

1.2 Wetland Wastewater Treatment Technology

The need for alternative wastewater treatment systems which are low cost in terms of investment, operation and maintenance especially in the developing countries, is long overdue. The systems required should be the ones that can easily be decentralised and scaled down to small sizes. The systems should also be adapted to the climate and should make use of simple technology and available skills for construction and operation with a possible re-use of the end product.

The use of treatment wetlands is one such option that meets these criteria. It has attracted special attention and interest from decision-makers, engineers and scientists around the world (Brix and Schierup, 1989; Denny, 1997). Wetlands introduce another beneficial aspect of nutrient cycling and ecosystem production in the overall process, through a symbiotic relationship between the plants and the associated microorganisms.

1.2.1 Natural wetlands

Natural wetlands are usually found at the interface between the terrestrial and aquatic ecosystems (Denny, 1985; Mitsch and Gosselink, 1993). They have been used world wide as dumping or disposal sites (the treatment potential mostly not identified) for different types of wastes. In the United States of America, some natural wetlands are documented to have been receiving domestic waste water for over 80 years (Kadlec and Knight, 1996). In Uganda, the Nakivubo and the Luzira natural wetlands which are dominantly colonised by *Cyperus papyrus* L. and *Miscanthidium violaceum* Robyns plants, have been receiving urban run off mainly from Kampala City centre and final effluent from the Bugolobi Sewage Treatment Works (BSTW) for over 50 years (Denny, 1997; Kansiime and Nalubega, 1999).

Several studies so far undertaken have demonstrated the functioning of natural wetlands in water quality improvement. However, their wide application for wastewater treatment is not yet well established and supported for several reasons. Brett (1989) and Verhoeven (1990) report the unpredictability of treatment performance that is realised when using natural wetlands in the temperate climates. Kansiime and Nalubega (1999) also found in the Nakivubo swamp near Kampala in Uganda, that although some form of treatment takes place, there is strong variability in the whole treatment process which is imposed by external factors. These findings indicate an inherent difficulty in managing and optimizing the functionality of natural wetlands with respect to the influent wastewater, the hydraulic flow pattern in the system and other process variables.

In addition to the unpredictability, natural wetlands are not suitable for direct application for wastewater treatment because of their other competing natural functional values. These include: biodiversity preservation, habitat and breeding sites for wild life, hydrological and hydraulic functions (Maltby, 1991; Denny, 1995). These wetland values could be interfered with and compromised by direct loading of wastewater into the natural wetland. The use of constructed wetlands specifically designed for the purpose of water quality improvement, is therefore considered a viable alternative that is not subject to the competing demands experienced in natural wetlands.

1.2.2 Constructed wetlands

The utilization of constructed wetlands (CW) in water pollution control provides an alternative perspective that is based on the water quality functions and values of natural wetlands but which is not limited by legal and conservation regulations. The historical background on the use of CW in water pollution control originates from the research pioneered by Seidel and Kickuth in Germany from 1952 (Bastian and Hammer, 1993; Kadlec and Knight, 1996). Utilisation in different countries started and developed at different times and rates. For example, in The Netherlands, the use of constructed wetlands started in 1967 with experimental work using *Scirpus lacustris* in a camping site in Flevoland (de Jong, 1976). Application in the USA commenced in 1967 (Kadlec and Knight, 1996) while in the United Kingdom, the use of the technology started in 1985 (Cooper and Green, 1998). At present, extensive research work on CW technology is being undertaken but mainly, in the temperate regions. Several key technical conferences dedicated to the use of wetlands in water quality improvement have been held dating back to 1976 from the chronology outlined by Bastian and Hammer (1993) and Kadlec and Knight (1996). The recent conferences were in Austria 1996 (Haberl *et al*., 1997) and Brazil in 1998 (Cooper *et al.*, 1998).

Review of the literature shows evidence of only limited research and use of the CW technology in the tropical regions. Many of the low income countries without the wastewater treatment infrastructure, are located within this belt and could therefore benefit more from the application of this technology. In sub-Saharan Africa for instance, some constructed wetlands have been operational in South Africa (Wood, 1990) and in Kenya (Nyakang'o, 1997). In Asia, investigations on the wetland technology are on going at the Asian Institute of Technology (Koottatep and Polprasert, 1997; Koottatep, 1999).The Kirinya constructed wetland described in this study, is the first to treat municipal wastewater using indigenous plants and, where detailed investigations into their functioning under tropical conditions have been done. It is envisaged that this study will form the basis for further research on the subject in the region. The study is also expected to be a catalyst for the countries in the region to exploit and harness the potential of wetlands in the provision of a cheap, effective, reliable and sustainable way of treating wastewater.

1.2.2.1 Types of constructed wetlands

Constructed wetlands may be classified on the basis of the dominant macrophyte such as: (i) submerged macrophyte (ii) free-floating macrophyte and (iii) emergent macrophyte (Brix and Schierup, 1989). These wetlands systems can be used alone, in combination or as final effluent polishers when used together with conventional treatment works. The last two types are widely used and will be discussed in the next paragraphs.

(a) *Free-floating wetlands*

The use of free floating plant systems in wastewater treatment exploits the rapid growth nature of these plants, which enables them to assimilate large quantities of pollutants into their biomass, often in excess luxury uptake (Denny, 1985; Reddy and DeBusk, 1987; Abbasi, 1987; Vymazal *et al.*, 1998). A schematic representation of these systems is illustrated in Fig. 1.2. The common plant species used include: water lettuce (*Pistia stratiotes*), Salvinia (*Salvinia* sp), duckweed (*Lemna* sp), mosquito fern (*Azolla* sp) and water hyacinths (*Eichhornia crassipes.*

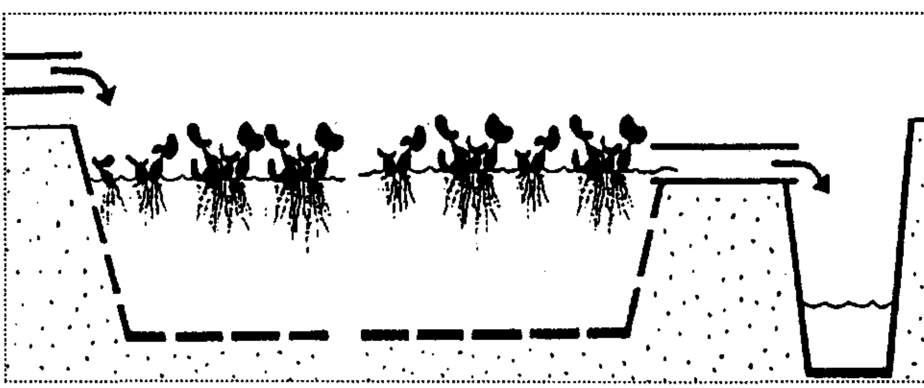

Fig. 1.2 Schematic representation of a free floating macrophyte based wastewater treatment system (from Brix, 1993).

In *Eichhornia crassipes* systems for example, high removal rates of suspended solids, organic matter (measured as biochemical oxygen demand, BOD) and nutrients are reported. Reed *et al.*, (1988) and Reddy *et al.*, (1989), associate this high efficiency to the ability of the plants to translocate oxygen from the shoots to the root zone. This is enhanced by the extensive root zone that also provides a large surface area for entrapment of solids and attachment of bacteria which are responsible for rapid degradation of BOD and nitrification. Uptake of nutrients by plants with subsequent plant harvesting is essential for export of nutrients (N & P) from this type of systems. Large macrophytes, such *Cyperus papyrus* can also be used in free floating systems as was the case in this study. It has similar structural characteristics of rooting medium in form of the thick root - rhizome mats with a large surface area.

(b) *Emergent wetlands*

Emergent macrophyte systems may be subdivided into three categories based on the flow pattern used. The Horizontal surface flow systems (SF or FWS) are characterised by wastewater flow above and through the rooting medium in shallow basins (Fig. 1.3). The reduced flow velocities provide the ideal conditions for the removal of suspended solids and particulate organic matter, while the biofilm (bacterial growth) on the plant stems is responsible for organic and nitrogen degradation. A lot of information on wetland performance is based on the data generated from these type of wetlands since they were the first generation of constructed wetlands built. Review of their performance by Reed *et al* (1988), Watson *et al* (1989), Cooper *et al* (1996) shows a large variability, especially with respect to nutrients removal .

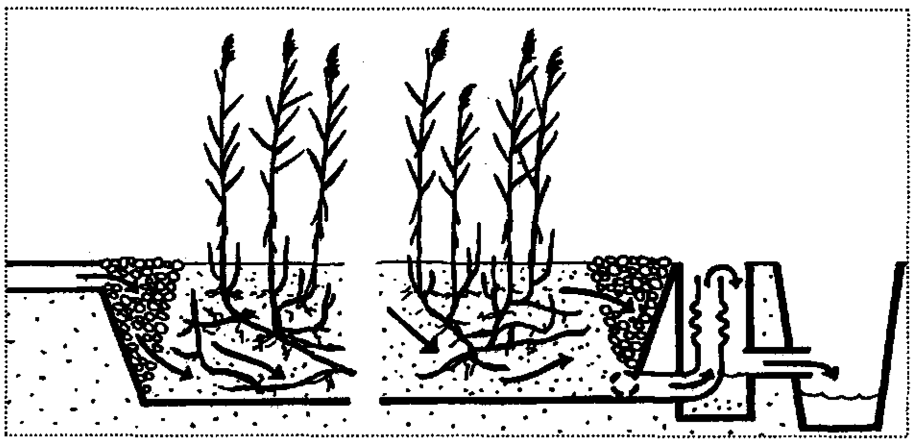

Fig. 1.3 Schematic representation of an emergent sub-surface flow macrophyte wastewater treatment system (from Brix, 1993).

The second category of emergent wetlands, are the subsurface flow systems (SSF). In these systems, wastewater is infiltrated into the porous medium with little or no water exposure on the surface. The infiltration can be at the inlet and wastewater flows horizontally under the bed and is collected in the outlet at the end of the bed (Brix and Schierup, 1989). The infiltration can also be introduced vertically and the wastewater percolates down through different layers of the porous medium and effluent is collected at the bottom (Cooper, 1993). In both cases it is during the passage of wastewater through the rhizosphere that it gets cleaned by the microbiological degradation and physical/chemical processes. The treatment efficiency in respect to nutrients is relatively higher than in surface flow types. A major disadvantage identifiable with the vertical flow infiltration is the clogging of the substratum medium.

1.3 Aquatic Plants

Several types of plants have been applied in treatment wetlands located in temperate and sub-tropical conditions. They are mostly found in the native environments of the wetlands. Readily available nutrients, light and water are responsible for making ecosystems colonised by wetland plants worldwide, to be the most productive (Thompson, 1985; Wetzel, 1993). Greenway (1997), lists the several types of wetland plants used for wastewater treatment in Australia.

Vymazal *et al* (1998), gave a detailed description of the types and classification of wetland plants applied in European and North American constructed wetlands.

The common plants in their listing include: (i) the emergents: *Typha* spp. (Cattails), *Scirpus* spp (Bulrushes), *Glyceria* spp. (Mannagrasses) and *Phragmites australis* (common reed); (ii) floating: *Eichhornia crassipes* (Water hyacinths), *Pistia stratiotes* (Water lettuce) and *Lemna* spp.(Duckweed) and (iii) *the submerged: Elodea* spp. among others. The use of the common tropical African wetland plants namely; *Cyperus papyrus* and *Phragmites mauritianus* in constructed wetlands, is initially referred to only in the greenhouse studies carried out in the Netherlands (Bruggen *et al.*, 1992; Okia, 1993). It was therefore the aim of this study to initiate the use of these native plants in a constructed treatment wetland specifically located in the tropics.

1.3.1 *Cyperus papyrus* L

(i) General characteristics

Cyperus papyrus L. is historically associated with early civilisation in Egypt and the Roman Empire, where it was used for making writing materials (Jones, 1983). It is an emergent aquatic sedge that is found mainly in the East and Central African wetlands fringing the lake shores and swamp valleys (Thompson, 1985; Gaudet, 1977; Chale, 1985; Denny, 1985; Bugenyi, 1993; Balirwa, 1998). It grows in virtual monoculture stands that can form a dense vegetation canopy (Carter, 1955; Beadle, 1974). The main structural features as described by Gaudet (1977), include: (a) the umbel, which is constituted of finely dissected bracteoles, it bears flowers and it is the main photosynthetic organ of the plant; (b) the culm (stem) which has a large proportion of a spongy aerenchyma on its inside and, to a small extent, it is capable of photosynthesis; (c) the rhizome and the roots which together form a mat like structure that is the base for swamp development. In natural swamps the rooting mat was estimated to contribute up to 30 - 52% of the total biomass (Beadle, 1974 and Thompson, 1985).

Cyperus papyrus grows both as a rooted and a floating macrophyte only in fresh water environments with stable hydrological regimes (permanently flooded). It cannot cope with rapid water level changes (Thompson, 1985). The floating wetland mat also known as the sudd (barrier) can float in water even as deep as 3 - 4 m (Balirwa, 1998; Kansiime and Nalubega, 1999).These mats are usually found along the fringes of lakes or river systems, while the rooted wetlands are found at the interface with terrestrial environments.

(ii) Environmental conditions

Papyrus-dominated wetlands like all other natural wetlands, are characterised by low dissolved oxygen concentrations. The main reason for this state is that surface aeration and photosynthetic oxygen transfer mechanisms are reduced or non existent due to the dense plant canopy. Any oxygen transferred from the shoots to the roots is utilised for root respiration and decomposition of the abundant organic matter by heterotrophic bacteria. For instance, the oxygen concentration measured by Carter (1955) and Gaudet (1979) in some natural wetlands in Uganda which were more or less undisturbed ranged from 0.9 - 4.6 mg /l at the surface and 0 - 2.4 mg /l at the bottom. Kansiime and Nalubega (1999) on the other hand obtained values ranging from 0 - 3 mg / l in the Nakivubo wetland, which has had many human interventions. The pH of the swamp water is characteristically acidic as compared to any adjacent open waters. The cause of this low pH is linked to the humic acids produced by anaerobic degradation processes that predominate these wetlands (Visser, 1962).

(iii) Productivity

The productivity of natural papyrus wetlands is found to be variable and controlled by different factors such as climate, nutrient availability and the prevailing general hydrological conditions. However, unlike other standing aquatic plants, its high standing biomass and productivity rates makes these plants have a high potential for nutrient removal more so in wetlands receiving water with a high nutrient load. In Kampala Uganda, a papyrus swamp both floating and rooted at the shores of Lake Victoria is used to purify secondary effluent from the city's sewage works and most of the urban surface run off. A study by Kansiime and Nalubega (1999) details the functioning of this wetland in this regard.

Table 1.1 shows values for the productivity and nutrient uptake of *Cyperus papyrus* and other aquatic plants under different growth conditions. There is similarity in the different natural papyrus swamps but were significantly different from the potted experiments which may be considered as a prototype constructed wetland. The difference may be attributed to the readily available nutrients in the latter system as compared to the natural ones.

Table 1.1 Standing biomass production and plant nutrient uptake rates under varying growth environment

Plant Type	Growth Environment	Productivity - standing biomass (kg DW ha^{-1} day^{-1})	P uptake (kg ha^{-1} day^{-1})	N uptake (kg ha^{-1} day^{-1})
Cyperus papyrus	Natural swamp water, Lake Naivaisha [1]	124 - 155	0.19	1.35
	Natural swamp water Uganda [2]	131 -392	0.06	1.18
	Sewage fed Nakivubo wetland [3]	130	0.21	1.30
	Potted (anaerobic pond effluent) [4]	1450	0.77	5.92
Phragmites australis	Infiltration wetland [5]	191	0.22	2.14
	Diverse [6]		0.05 - 0.08	0.5 - 0.6
Eichhornia crassipes	Diverse [6]		0.2 - 2	1.6 -6.6

Source: [1] Muthuri et al., 1989; [2] Thomson et al., 1979; [3] Kansiime and Nalubega, 1999; [4] Lizhiboa, 1995; [5] Meuleman, 1999 and [6] Reddy and DeBusk, 1987.

1.3.2 Phragmites mauritianus (Kunth)

Phragmites mauritianus (Kunth) belongs to a group of monocots, family of Poaceae (Gramineae).

Its distribution is exclusively tropical and it is one of the abundant wetland plants in East and

Central Africa. It is intolerant to flooding and is usually found growing in areas of shallow hydrologic gradient. Therefore, the plants show preference for the outskirts dominating the transition between terrestrial and aquatic systems, with water-logged back up streams (Thompson and Hamilton, 1983). It is one of the native plants that colonise the adjacent natural wetland at Kirinya.

The choice of this plant in the study was based on the extensive and the success application of its generic sister, *Phragmites australis* in many treatment wetlands in temperate climates (Cooper, 1996., Urbanc-Bercic and Gaberscik, 1997; Vymazal et al., 1998). *Phragmites australis* has high productivity rates which makes it suitable for use in constructed wetlands. Primary production from an infiltration wetland of up to 70 ton ha^{-1} yr^{-1} was obtained by Meuleman (1999).These attributes associated with *Phragmites australis* were assumed to be interchangeable with *Phragmites mauritianus*. This was one of the reasons for using the latter in this study. *Phragmites mauritianus* has a much larger culm than P. australis and this makes it have extensive use as a fencing and roofing material. Utilization of these plants in constructed wetlands for wastewater purification, would enhance its value as a final tangible product with much more beneficial values.

1.4 Hydrological Factors

Wetlands are by definition created and maintained by water. Several factors such as the water source, water depth, flow rates, residence time etc., influence the wetland hydrodynamics and the physico-chemical properties of the wetland substrata sediments. The types of flora and fauna that develop in a wetland from a particular region together with the nutrient dynamics and biological transformations taking place are largely influenced by the hydrodynamics in the wetland (Mitsch and Gosselink, 1993; Kadlec and Knight, 1996). The more time water spends in the wetland, the higher the chances for interactions between waterborne substances and the wetland ecosystem. This aspect is exploited in constructed wetlands.

Water movement in wetlands is influenced by vegetation and strong interaction with the atmosphere via precipitation and evapotranspiration. In treatment wetlands, these factors may influence the treatment process. Precipitation dilutes the concentrations and speeds the flow while evapotranspiration increases concentrations and reduces flows (Kadlec, 1987). In this study, during one operation phase (1997/98), there was excessive rainfall which was associated with the El Nino phenomenon. It created a negative impact in the treatment performance with respect to all parameters.

1.5 Kirinya wetland systems

The Kirinya pilot constructed wetlands are located at the National Water and Sewerage Corporation Sewage Treatment Works, in Kirinya Jinja Municipality. Three factors led to the establishment of this pilot plant at this site in Jinja. The first was the feasibility of using *Cyperus papyrus* which was demonstrated in the studies carried out at the greenhouse of the Technical University of Delft (Bruggen et. al, 1992; Okia, 1993). Subsequent investigations carried out by Lizhibowa (1995), Kiwanuka (1996) and Sekiranda (1996) further showed *Cyperus papyrus* and *Phragmites mauritianus* as good candidate plants for use in constructed wetlands.

The second aspect was the willingness by the National Water and Sewerage Corporation

(NWSC), a statutory organisation responsible for the supply of drinking water as well as collection and treatment of sewage in the big urban centres in Uganda, to implement the research in one of its facilities based in Jinja. At the time, in all areas of its operation, final effluent from treatment works was channelled to natural wetlands. The new national regulatory requirements on the quality of effluent discharged from the treatment works, are putting pressure on the organisation to seek for improved and cost effective means for polishing its effluents prior to discharge. Participation in this research was given priority by the organisation.

The last and probably the most crucial aspect, was the financing of all research components by Rijksinstituut voor Integraal Zoetwaterbeheer en Afvalwater behandeling, RIZA (The Institute for Inland Water Management and Wastewater Treatment), which is an advisory institute for the Netherlands Ministry of Transport, Public works and Water Management.

The constructed wetlands were based on the use of floating *Cyperus papyrus* and rooted *Phragmites mauritianus* plants. Both species are native. The wetlands were categorised as horizontal surface flow (SF) even in the case of *Cyperus papyrus*.

1.6 Research scope and relevance of this thesis

This study was set to provide solutions that could be applied to larger constructed wetland systems within the tropical environments. A specific task was to find out the optimal loading and operating conditions for the reduction of bulk pollutants and nutrients; the suitability and functioning of the wetland plants in the overall treatment process and the impact of the hydrological regimes on the effectiveness of the units in reducing pollutants. An additional study task was to evaluate the applicability of the design and operation of the smaller systems in wastewater treatment at the household level.

Specific relevance of this study was based on the need to evaluate the potential of constructed wetlands planted with native macrophytes as a low cost and yet effective technology in water pollution control. Since the use of the technology is new in the region, using the demonstration pilot wetland was considered crucial in providing observable results by the citizens that can be applied locally. The existing warm and favourable climate all the year round in the tropics was expected to stimulate high plant productivity thereby creating conducive conditions required by the biological communities to thrive and degrade pollutant

In addition to building capacity in many of the stakeholder institutions in the country, the study was expected to form a basis for integration of the wastewater treatment with other production systems such as, irrigation and fish farming. This would make the technology have more diverse benefits.

1.7 Objectives of this thesis

The general aim of this study was to improve our knowledge base of wastewater purification processes using constructed wetlands in tropical environments with a view to developing optimal design and operation criteria that may be applied in wetlands sited in similar environments.Specific objectives were:

- To determine processes and performance attainable in constructed wetlands with *Cyperus papyrus* and *Phragmites mauritianus* plant species under different loading rates and operating conditions with respect to chemical oxygen demand (COD), biochemical oxygen demand (BOD_5), suspended solids (TSS), nutrients (nitrogen, phosphorous) and pathogens.
- To determine the functional role of macrophytes used in uptake of nutrients and their storage capacities in the standing and rooting biomass.
- To evaluate the design and performance of a household constructed wetland.
- To propose guidelines for design, construction, use and management of constructed wetlands based on information collected on processes and costs involved.

The research work was carried out using a pilot constructed wetland systems as described in the next Chapter.

1.8 Outline of the thesis

This thesis contains two main sections, each dealing with a different study area and different size of the systems. The first section describes the studies in Kirinya pilot constructed wetlands in Jinja that received settled municipal sewage (Chapter 2 to 5). The second section deals with studies carried out using a single household constructed wetland receiving effluent from a septic tank (Chapter 6).

In Chapter 2, a description of the Kirinya pilot wetlands is given detailing, the system design and lay out, the start up of the systems including the plant equilibration procedures. The pond configurations adopted and the operational modes applied in the three phases are described. The wetland hydrological conditions and water balance in each of the ponds is outlined.

Chapter 3 deals with the performance of the wetland systems with respect to organic matter and suspended solids. The impact of environmental parameters namely oxygen, temperature and pH on the removal of these pollutants is emphasised.

In Chapter 4, the impact of pH and dissolved oxygen derived from the three different wetland configurations on the removal of the nutrients especially, ammonium is outlined. Plant growth characteristics, biomass yields and nutrient distribution in the two plant types is detailed. The contribution of plant uptake to the removal of nutrients from the systems is clearly illustrated by the mass balance computation.

Chapter 5 deals with the removal of faecal coliform bacteria by the different wetland units. The effect of hydraulic loading and exposed open surface areas on the systems performance is demonstrated.

In Chapter 6, the household demonstration wetland treatment performance is evaluated. Factors that will influence the functioning and acceptability of the technology, together with the public perceptions about the wetland are documented.

In Chapter 7, an economic appraisal using the total annual cost of a constructed wetland in comparison with a waste stabilisation ponds is outlined.

The summary and overall conclusions derived from the study are outlined in Chapter 8. The general guidelines for design, operation and management of the constructed wetlands in Uganda and within the East African region are given based on the findings of this study.

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Chapter 2

The Kirinya Pilot Constructed Wetlands: Structure and Hydrology

Abstract 19

2.1 Introduction 19
- 2.1.1 Research Strategy 20
- 2.1.2 Location of Pilot constructed wetlands 21
- 2.1.3 Design Criteria 21
- 2.1.4 Construction Aspects 22
- 2.1.5 Start up of Systems 22

2.2 System Operational Modes 24
- 2.2.1 Pond calibration 24
- 2.2.2 Phase 1 26
- 2.2.3 Phase 2 27
- 2.2.4 Phase 3 28

2.3 Wastewater Sampling 30

2.4 Wetland Hydrology 31
- 2.4.1 Introduction 31
- 2.4.2 Water Balance terms 31

2.5 Materials and Methods 32

2.6 Results 32

2.7 Discussion 36

2.8 Conclusions 37

References 37

Appendix 41

Abstract

The Kirinya pilot constructed wetlands comprised of eight units which were constructed in such a way that infiltration of water into or out of them was prevented. *Cyperus papyrus* plants were planted in four units and *Phragmites mauritianus* in two units. Two units had no plants and were used as controls. The pond configurations of the planted wetland units were varied in three distinct consecutive phases. In the first phase, all planted wetland units were undisturbed but in the second phase, standing and rooting biomass was removed from an area of 10 m^2 near the inlet in three vegetated wetland units. In the last phase, the pond lengths were extended by two modes; combining in series, two vegetated ponds with a non-vegetated one and combining two papyrus units having alternating open and planted zones, each 10 m^2.

Applied hydraulic loading rates were varied in each pond and each series but overall ranged from 1.2 cm/day to 10 cm/day over the whole experimental period. The pond water temperatures in the plant covered ponds ranged from 22 to 25 ^{0} C; a maximum temperature of 30 ° C was measured in the control (open) ponds. Mean rainfall measured at the pilot wetland site varied over the experimental period from 3.3 to 4.3 mm/day. Mean evapotranspiration rates in *Cyperus papyrus* and *Phragmites mauritianus* systems and in the control ponds were 6.1 mm/day, 5.6 mm/day and 3.8 mm/day respectively.

2.1 Introduction

Kirinya Sewage Works (the site of the pilot constructed wetlands) is one of the two sewage treatment facilities run and managed by the National Water and Sewerage Corporation (NWSC) in Jinja municipality. It treats both domestic and industrial wastewater by use of a series of five stabilisation ponds: two anaerobic, two facultative and one maturation. The final effluent is discharged to an adjacent natural wetland fringing Lake Victoria. The general characteristics of effluent from each of these units is given in Table 3.1 (Chapter 3).

The climate of the study area is a typical equatorial type modified by its location at the fringes of lake Victoria, and by the altitude. It is characterised by a double rainfall maximum in March-May and September-November, with an annual mean rainfall of 1300 mm. Air temperatures range between 15 - 30°C and the relative humidity was in the range of 45% to 85%. (Source: Uganda, Meteorological Department, Kampala).

The adjacent fringing natural wetland is dominated by two native plants: *Cyperus papyrus* and *Phragmites mauritianus*. *Cyperus papyrus* is predominant in the permanently flooded zones near the open waters of the lake while *Phragmites mauritianus* is mainly found in the non-flooded outer zones of the swamp. The fringing natural wetland is also inhabited by many species of wetland-related animals, like frogs, rodents, monkeys, birds, snakes (cobras) and monitor lizards. Many of these animal species became part of the constructed wetland ecosystem as well within a few months of establishment.

2.1.1 Research Strategy

Background

Natural and constructed wetlands systems have a very complex hydraulic and microbiological nature which is influenced by several factors. The detailed processes occurring inside these systems is still not well understood. It is now the focus of many investigations by various researchers. Hitherto, most of the constructed wetlands which are in use in America and Europe have been designed without this detail but relied more on influent and effluent data and, mainly of the 5 day biochemical oxygen demand (BOD) and suspended solids (EPA, 1988; Hammer, 1989; Cooper and Findlater, 1990; WPCF, 1990 and Moshiri, 1993). Many of these systems are operated as continuous surface or subsurface flow units as described in section 1.2.2 (Chapter 1). Plug flow conditions have been assumed to operate in the systems. However, results to date indicate that the flow is not uniform (Tchobanoglous, 1993; Kadlec and Knight, 1996 King *et al.*, 1997). Therefore plug flow consideration are only used to provide an approximation of the conditions. Lakshman (1981) and, Kadlec and Knight (1996) extended these assumptions to fit and explain results of BOD and total nitrogen reduction in three surface flow wetlands operated in a discontinuous (batch) flow mode. The same rationale was applied in choosing the operational modes and computations in this study.

Approach

In this study, the design strategy adopted was therefore to have a system that would be responsive to both continuous and intermittent (batch) loading of wastewater. These two feed scenarios both apply to areas where there is a potential use of the constructed wetland technology in the country. The following approach was adopted in the design and study:

(i) To use a pilot wetland built in such a way that only controlled and measurable quantities of wastewater and rain are the only input into the system.

(ii) To build a pilot wetland unit in such a way that it is operated in a flexible and variable format to allow for optimization studies on wastewater treatment process for the various pollutants outlined.

(iii) To design the pilot wetland using the criteria previously applied for continuous surface flow (SF) systems, but it was to be adaptable to other feed regimes and configurational modifications as well.

(iv) To use two native wetland plants – *Cyperus papyrus* and *Phragmites mauritianus* in the units but in conditions that are suitable for their optimum growth in the native environments. In the case of *Cyperus papyrus* this means floating, without any substratum and for *Phragmites mauritianus* this means rooted on a substratum base or soil. Gravel was used as a substratum base on assumption that it would improve hydraulic flow and minimize clogging, which frequently occurs with a soil substratum base.

(v) To have a blank unit for each plant type and operated under similar conditions. At the design stage, the strategy was to have this unit covered completely to prevent algal growth but this was not practical at the site. The units were left exposed for the rest of the experimental period. The results obtained from these units were used for comparison with the vegetated units but good comparison would be with efficiently run lagoons.

2.1.2 Location of pilot constructed wetlands

The Kirinya pilot constructed wetland is within the Kirinya Sewage Works premises which is situated at latitude 00° 27' N, longitude 33° 11' E and at altitude of 1175 m above mean sea level. The sewage works are located 2 km East of Jinja Municipality (which is 80 km east of Kampala, the capital city of Uganda) on Kirinya Prisons road (Fig.2.1).

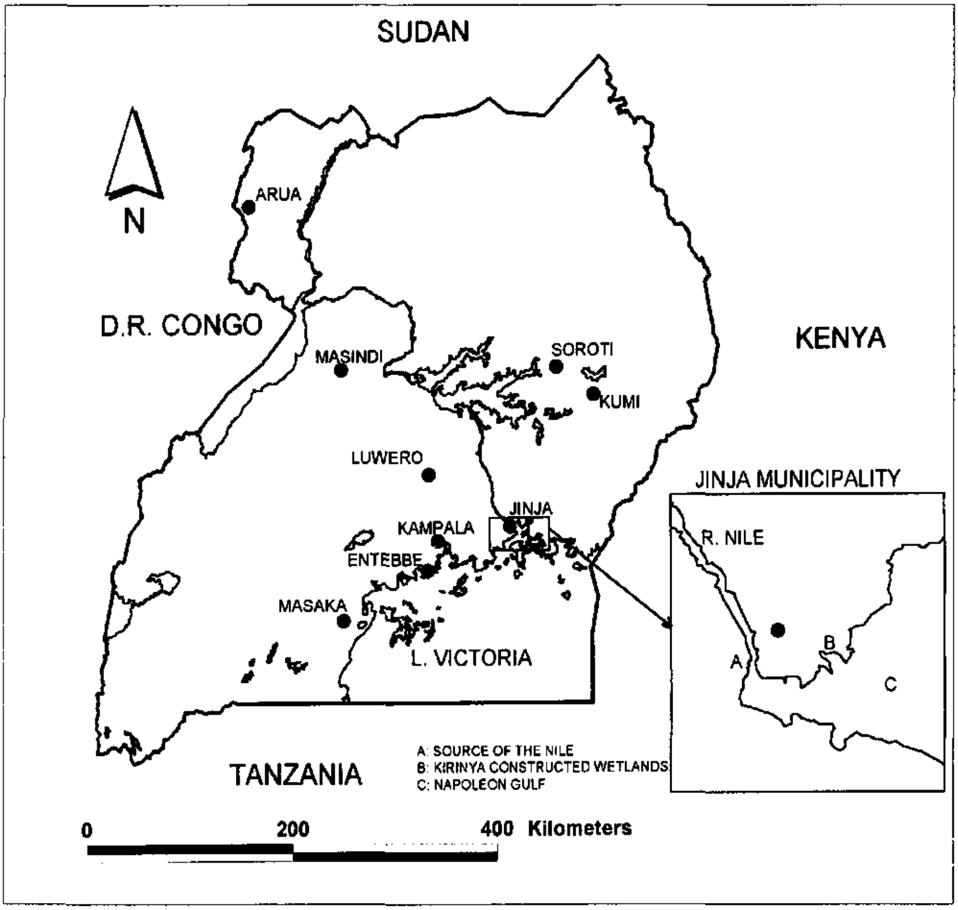

Fig. 2.1 Map of Uganda showing the location of Kirinya constructed wetlands, in Jinja municipality.

2.1.3 Design Criteria

The design of the wetland units was based on the Water Research Council (WRC) (1990) and the Water Pollution Control Federation (WPCF) (1990) guidelines for surface (horizontal) flow wetland systems. As mentioned earlier, BOD was the critical parameter utilised in the computation. Some modifications were however made with respect to the big floating *Cyperus papyrus* and the warmer temperatures that prevail in the study area.

The summarised design criteria used were:

Area per nominal population equivalent (PE) = 1.2 m^2/PE;
Influent BOD concentration: maximum 150 mg/l;
Effluent BOD: 30 mg/l.

The resulting unit design data were as follows: total wetland surface area, 320 m^2; mean depth 1 m; length 20 m and width of 16 m. For better hydraulic control at the inlet, the wetland was divided into eight parts to give individual units, each 2 m wide and resulting into a length-to-width ratio of 10: 1.

2.1.4 Construction Aspects

The constructed wetland occupied a total area of about 500 m^2 and it comprised of eight wetland units (ponds). Each unit was built with a bottom slope of 2 %. The pond bottom was made of a 125 mm concrete slab, laid over an impermeable plastic lining overlying a 50 mm concrete blinding and 150 mm hard core. Gabions (2 m x 1 m x 1 m) packed with 60-100 mm diameter pebbles were fixed at the inlet and outlet positions of each wetland unit. A 2000 mm x 100 mm diameter collector cast iron pipe with 5 mm perforations was fitted at the bottom of the outlet gabions. An adjustable flexible hosepipe was fitted to the outlet pipe in the drain chamber for regulating the water level of each pond. The detailed drawing for a typical unit is illustrated in the appendix.

A pump house was built in a position of a sump that was created by cutting a section of outflow pipe from the anaerobic lagoons. A submersible pump was fitted in this house for delivery of wastewater to an elevated feeder header tank with a capacity of 60 m^3. A bulk water meter was installed at the outlet of the feed pipe from the header tank for volumetric measurements of wastewater loaded into the wetland systems. A schematic lay out of the constructed wetland units is given in Fig. 2.2. Plates 1 and 2 in the appendix show the wetland at different construction stages.

The wetland units (ponds) numbered from 1 to 8, had the following arrangement:

Ponds 1, 2, 5 and 6 had *Cyperus papyrus* plants without any substratum.
Ponds 7 and 8 had *Phragmites mauritianus* plants on a gravel substratum
Pond 3 was a control (with no plants) for *Cyperus papyrus* units
Pond 4 was a control (with no plants) for *Phragmites mauritianus* units and had a gravel substratum.

2.1.5 Start up of Systems

(i) *Plant Nursery*

A *Cyperus papyrus* nursery was established to generate a sufficient number of young plants required for planting in the wetland units.
Plant propagules with an average height of 10 cm were collected from the cultivated sections of

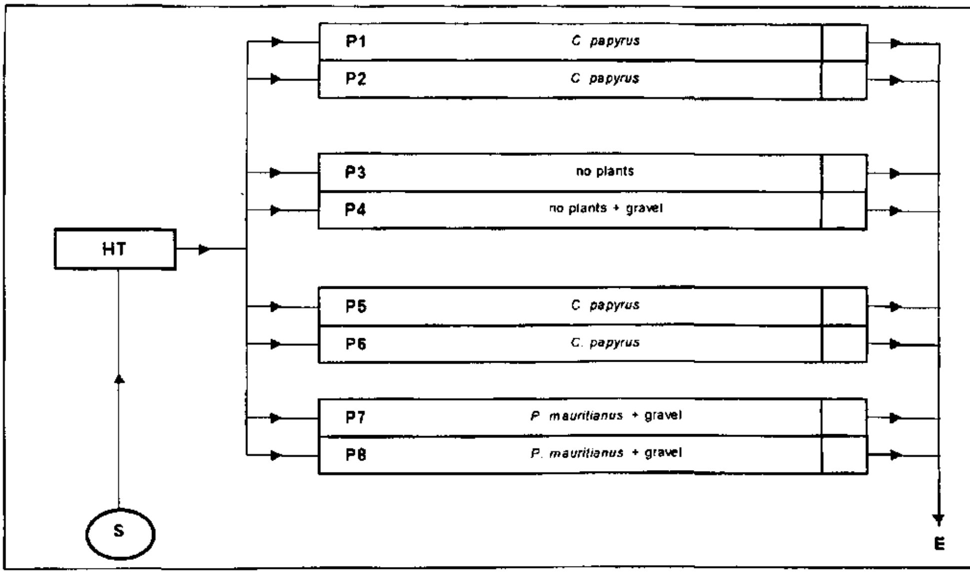

Figure 2.2. Lay out of constructed wetlands ponds at the Jinja Kirinya Sewage works.
Key: HT = Header tank, S = pump house, E = Drain outlet.
Transmission main, → Feed and drain lines.
P 1- P 8 = the eight wetland units (ponds).

the adjacent natural wetland. In the nursery, each propagule was planted in special polyethylene bags (1 litre) filled with soil mixed with dried sludge and saturated with wastewater from the facultative pond. The plants were allowed to grow up to an approximate height of 40 cm before transplanting into the wetland units.

(ii) *Substratum filling*

Gravel composed of a mixture of granitic and laterite types of variable size range (10 -52 mm diameter), purchased from a local quarry near Kirinya. The gravel was added into three ponds: P 4, P 7 and P 8 up to a uniform depth of 10 cm from the inlet position and up to a depth of 30 cm at the drain end of the ponds. It constituted a substrate solid matrix for anchoring *Phragmites mauritianus* plants in ponds 7 and 8.

(iii) *Transplanting of propagules*

All wetland units were filled with wastewater up to depths of 15 cm and 25 cm in systems designated for *P. mauritianus* and in *C. papyrus* respectively, prior to the introduction of plants. Propagules of *C. papyrus* taken from the nursery, were transplanted into wetland units No. 1, 2, 5 and 6. The propagules were initially planted in three rows spaced 0.3 m with each row having 3 plants at equidistant positions and were anchored in position using strings and wooden sticks suspended across the ponds. However, wind disturbed the young plants and consequently, in the subsequent planting, propagules were introduced differently to minimise this effects.

Culms of the propagules were cut off leaving only the rooting biomass tufts which were then floated in between wooden rectangular rafts each with approximate dimensions of 1.8 m x 1.5 m.

Phragmites mauritianus propagules were introduced to wetland units No. 7 and 8, twice. In the first instance, the propagules were planted directly into the saturated gravel solid matrix at a similar spacing as in *C. papyrus*. However, these plants were easily uprooted by the monkeys within the first week and had to re-planted but this time, by burying nodular stems in a saturated gravel matrix. Well anchored shoots sprouted within a week.

All planted systems were left to equilibrate for 1 month. This enabled plants to stabilise and allowed for new shoots and roots to develop.

2.2 System Operational Modes

As stated in section 2.1.1, the operation of the wetland units was to be responsive to both batch and continuous wastewater feeding regimes. Throughout the experimental period, wastewater loading regimes were therefore manipulated with the guiding principle of achieving optimum treatment efficiency with respect to bulk pollutants and nutrients, without an external energy or chemical input. This was achieved by running the wetland units in three distinct ways, hereafter referred to as phases. The sketch side view typical for the different system configurations in each phase is illustrated in Fig. 2.3 A- E.

In all phases, wastewater was loaded into the different ponds using a flexible tube connected to a distribution junction from the header tank. On each feed day, the header tank was flushed prior to filling with influent wastewater. The volume of wastewater loaded to the ponds was measured using the bulk meter.

2.2.1 Pond calibration

In each experimental phase, water volumes in the wetland were calibrated against water levels. This was essential for the determination of effluent volumes and the increasing volume of the rooting biomass. Each unit was completely drained and loaded with 2 m^3 of wastewater at intervals while recording the corresponding water levels at mid position of each of them. A time lag of 10 minutes was maintained between wastewater loading and water level measurement to allow for equilibration.Water level - volume calibration curves and equations were generated for each of the planted wetland units during every measurement cycle.

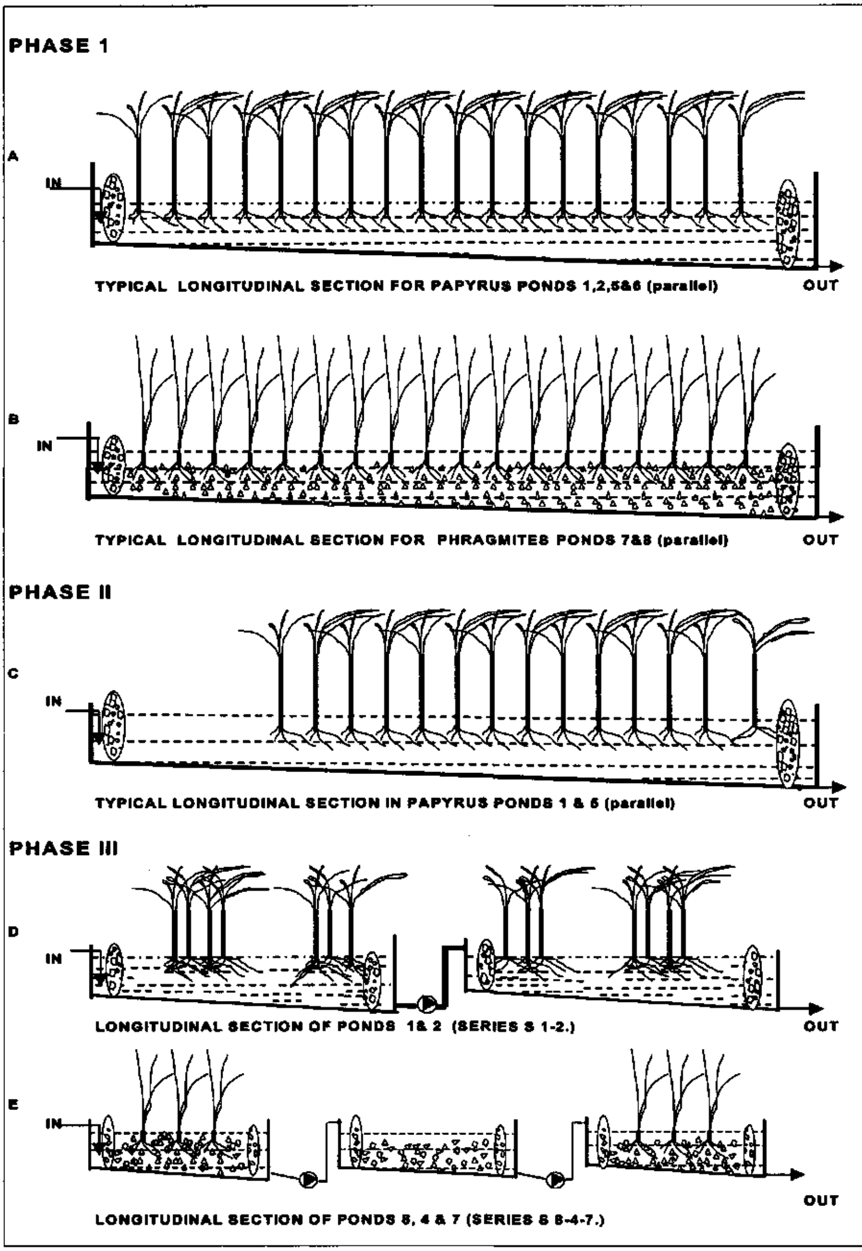

Fig 2.3 Sketch of the longitudinal sections of the ponds at different operational phases. In and Out show positions of wastewater inflow and outflow, respectively.

2.2.2 Phase 1

(i) *Introduction*

This was the exploratory phase that was operational from April 1996 - March 1997 and had the configurational setup illustrated in Figs 2.3 A & B. Experimental measurements commenced in May 1996 in *Cyperus papyrus* units and in September 1996 for the other units. The plant density at the commencement of experiments was 20 shoots /m^2 with shoot heights ranging between 1 m to 1.5 m. Theoretical hydraulic retention times were fixed for each of the wetland units but were spread within the ranges reported for constructed wetlands. To achieve the variable ranges in retention times, wetland units were loaded with different volumes of wastewater, and also drained to different water levels.

Water levels were set for each unit and wastewater was loaded up to that level each time. The corresponding volume of water read from the bulk meter at that level, here referred to as effective volume, was recorded. Effective (pond) volumes and loaded wastewater volumes were adjusted periodically in response to reduced overall wetland volume caused by increasing plant rooting biomass.

(ii) *Loading format*

Wastewater was loaded and drained batch-wise on a weekly basis in all ponds with the exception of pond 5 which was loaded twice a week. In each case including pond 5, effluent was always drained first prior to loading. The sequence followed on a sampling day for each pond was as follows:

(1) Reading the water level at the middle position of each unit.
(2) Draining off the water completely or to a fixed level established for unit and
(3) Loading wastewater up to the fixed level for each unit.

The underlying assumptions for the weekly batch-loading format were:

(a) the long contact time of the wastewater and the plants created as a result, would increase the extent of removal of pollutants.
(b) the intermittent load - drain format applied during each loading cycle was expected to stimulate increased transfer of oxygen from the air into the exposed roots and mat complex. This in turn would enhance degradation of organic carbon and nitrogen degradation which are oxygen demanding processes. The loading format would therefore increase the level of treatment attained.

A summary of the average operational data applied is given in Table 2.1.

Table 2.1 Volumetric loads and derived operational data applied; in papyrus ponds 1, 2, 5 and 6, the data range covers the period September 96 - March 97 while for the other ponds it applies from October 96 - March 97.

Pond	Feed cycle (days)	Volume added (m^3)	Pond Vol. (m^3)	Hydraulic loading rate (cm/day)	Nominal retention time (days)
1	7	16	16	5.71	7
2	7	13.5	13.5	4.82	7
3	7	7.1	12.5	2.54	12
4	7	3.6	6.2	1.29	12
5	4	13.5	15	8.44	4
6	7	9	15	3.21	12
7	7	6.6	6.6	2.36	7
8	7	3.6	6	1.29	12

Key: Pond volume refers to the volume calculated based on the water level height measured after loading and assuming a porosity of 1 (Kadlec and Knight, 1996).

2.2.3 Phase 2

(i) *Introduction*

This phase covered activities undertaken from September 1997- May 1998. In the previous phase, persistent low dissolved oxygen concentrations in the water (< 2 mg/l) prevailed in the planted units, especially as the plant density increased whereas high values (> 12 mg/l) were obtained in the open ponds. Reduction of ammonium concentrations was low in all the planted units but high in the open ponds. The focus in this second phase was to improve oxygen supply into the planted wetland systems.

A new strategy adopted was to modify the configuration of the wetland units by creating open zones which would allow for increased surface aeration and limited algal growth. As suggested by Green *et al* (1997), these alterations were expected to result into elevated oxygen concentrations in the open zones. This would in turn enhance the nitrification process, and as water flowed through the adjacent plant covered parts which were anoxic, denitrification would take place causing eventual reduction of nitrogen loaded into the systems.

(ii) *Pond configuration modifications*

Plant biomass (standing and rooting) was completely removed from an area of 10 m^2 at the front end of three ponds: 1, 5 & 7; the remaining ones served as controls (Fig. 2.3 C). In order to maximise on the expected increased dissolved oxygen, the systems were to be operated at reduced hydraulic loading rates.

(iii) *Wastewater loading format*

Batch weekly loading was applied from September 1997 - December 1997.
This was expected to provide data for comparison with the operation of the previous format (phase 1). It was based on the assumption of increased oxygen transfer into the wetland units as a result of the created open surface areas.

Daily batch loading (semi-continuous) was implemented from Januari 1998 - April 1998. This mode was a first attempt to obtain the scenarios in the real systems that run on continuous daily loading. Secondly, it was in response to the effect of unexpected heavy rainfall experienced over that time. Mean monthly rainfall of 250 mm was recorded in the months of October 1997 to December 1997 compared to the mean of 75 mm for the same period from previous years' data (data from the Kimaka Meteorological Station). This had a direct effect in the functioning of the systems under the weekly batch feed format especially, as the outflow levels were fixed. In some periods complete wash-out and flooding of the wetland units took place before the next loading. These events caused alterations in the concentrations of pollutants within the wetland.

A summary of the operational data is given in Table 2.2.

Table 2.2 a. Waste water load volumes and derived operational data in the 1st half of Phase 2; September to December 97.

Pond	Load Vol. (m^3 /day)	Pond Vol. (m^3)	Hydraulic loading rate (cm/day)	Nominal retention time (days)
1	0.72	10	1.8	14
2	0.72	10	1.8	14
3	0.72	10	1.8	14
4	1.42	10	3.6	7
5	1.42	10	3.6	7
6	1.42	10	3.6	7
7	1.42	10	3.6	7
8	1.42	10	3.6	7

Table 2.2 b. Waste water load volumes and derived operational data in the 2nd half of Phase 2; January to April 98.

Pond	Load Vol. (m^3 /day)	Pond Vol. (m^3)	Hydraulic loading rate (cm/day)	Nominal detention time (days)
1	1.10	10	2.7	9
2	1.10	10	2.7	9
3	1.10	10	2.7	9
4	1.72	10	4.3	6
5	1.74	10	4.4	6
6	1.75	10	4.4	6
7	1.86	10	4.7	5
8	1.74	10	4.3	6

Note: *The nominal detention time was computed considering only the influent flow rate (Eq.* 3.6 Chapter 3).

2.2.4 Phase 3

(i) *Introduction*

Preliminary investigations in this phase were done in May 1998 but detailed studies were carried out between September 1998 to March 1999. Results from phase 2 had shown an increase in dissolved oxygen concentration in the wetland units with partial plant clearance. However, the final concentration of ammonium from the modified as well as the unmodified planted ponds was still high. This indicated that oxygen demands for both organic matter and nitrogen compounds had not been met. An additional modification was made to increase oxygen.

(ii) *Pond length modifications - 60 m*

This modification involved loading wastewater to a set of ponds connected in series. The first series comprised of two *C. papyrus* ponds (P 5 and P 6) and the control pond 3. They were loaded (joined) in a such a way that influent from the header tank was fed into pond 6, effluent from pond 6 was fed as influent to pond 3, effluent from pond 3 was loaded as influent to pond 5 and the final effluent discharged from pond 5.This series is here referred to as S 6-3-5. The second series comprised of *Phragmites* ponds (P 8 and P 7) and the control pond 4. Wastewater was loaded in the same format as described for *C. papyrus*. This series is named as S 8-4-7. The total serial pond length in each case was 60 m. The drain flexible weir in each pond was fixed in a such way that after loading, the pond water volume was 10 m^3. A typical illustration is given in Fig. 2.3 E.

The basic assumptions for creating this serial configurations were:

(a) Removal of most carbonaceous material from the influent would take place in the first pond. This would improve the nitrifying bacteria competitiveness for oxygen in the second pond. (Results from previous phases showed high organic removal in the 1^{st} pond).
(b) The open water pond would enhance oxygen transfer via surface aeration and algal photosynthesis, which would increase the nitrification rate.
(c)The last pond would provide an environment for the denitrification process to take place as well as sedimentation / filtration and decomposition of secondary organic matter and suspended solids from algae, plant debris etc.

(iii) *Pond length modifications - 40 m*

The last series comprised of *C. papyrus* ponds 1 and 2. Waste water from the header tank was loaded into pond 1 and effluent from pond 1 was loaded as influent for pond 2. This series referred to as S 1-2, had an overall serial pond length of 40 m. Wastewater was loaded to each of the ponds until levels that maintained a water volume of 10 m^3. Prior to this loading format, plant biomass (rooting and standing) was removed from the two ponds in an alternating manner: 10 m^2 open area followed by 10 m^2 planted area thereby creating equal total areas of planted and unplanted zones (40 m^2, each). The configuration is illustrated in Fig. 2.3 D.

The assumptions stated above for the other series were still valid for this series (S 1-2) as well, but an additional hypothesis was considered in this case. That pH and dissolved oxygen concentrations would increase in the open zones to levels necessary for nitrification.

A summary of the data derived from measurements taken for all the three phases is given in Table 2.3

Table 2.3 Wastewater volumetric loadings and operational data in Phase 3, (daily averages), September 98 - March 99.

Series	Influent Vol. to 1st Pond (m^3/day)	Effluent Vol. from last pond (m^3 /day)	Hydraulic Loading rate (cm/day)	Retention time (days)
S 1-2	4.1	3.8	5.1	2.5
S 6-3-5	5.3	3.2	4.4	3.5
S 8-4-7	6.6	5.1	5.4	2.6

Note: The hydraulic loading rates and retention times were computed for the whole series in each case

2.3 Wastewater Sampling

The approach used during sampling in the course of this research was based on the modest but adequate equipment available and accessible to the researcher. Procedures for sampling and sample handling were followed as given in standard methods and procedures for water and wastewater analysis (APHA, 1992).

(i) *Sampling points*

In phases 1 and 2, sampling points were identified and fixed in each unit at pond lengths of 4, 10, 18 & 20 m from the influent position. In phase 3, additional sampling points were located at distances of 22, 27, 30 and 40 m in papyrus series S 1-2 and at length of 30, 40, 50 and 60 m along *C. papyrus* series S 6-3-5 and *Phragmites* series S 8-4-7.

(ii) *Sampling frequency*

In phase 1, sampling was carried out weekly in all the ponds except pond 5 which was sampled twice a week. Weekly sampling was repeated in phase 2 in all ponds during the first half but in the second half of phase 2 and in phase 3, bi-weekly sampling routine was applied. In all phases, composite influent samples were taken during loading operation.

(iii) *Sampling formats*

Sampling was normally done between 1000 - 1200 hrs. A fixed sampling procedure was followed and essentially involved:

a) taking duplicate water samples (≈ 300 - 400 ml) using 500 ml plastic containers cleaned and rinsed before draining and or loading the wetland units .
b) taking similar volumes of final effluent (composite) samples commencing at least 5 minutes after the start of wetland draining, (this was to allow for flushing through the collector pipes).
c) taking composite influent samples as loading of wastewater to the units proceeded.

All collected samples were stored in an ice cooled box for transportation within 6 hrs to the National Water Sewerage Corporation Central Laboratory, Bugolobi .

2.4 Wetland Hydrology

2.4.1 Introduction

The influence of hydrology on wetland hydrodynamics, structure and functioning is extensively reviewed in the literature (Gosselink and Turner 1978; Novitzki 1978; Hammer and Kadlec, 1986; Brown and Stark 1989; Mitsch and Gosselink, 1993; Lewis 1995; Kadlec and Knight 1996). Hydrological regimes are notably considered as the most important factors that control the ecological, physical and chemical characteristics of wetlands.

Water movement in wetlands is affected by the strong interaction with the atmosphere via rainfall, wind, evapotranspiration and vegetation. For treatment wetlands, these interactions may influence the purification processes in various ways. Kadlec (1987) indicated that precipitation dilutes the concentration of the wastewater present in the wetland and increases the through-flow. This in effect reduces the time of interaction between waterborne substances and the wetland ecosystem. On the other hand, evapotranspiration tends to concentrate the constituents of wastewater and reduces outflow (increase in retention time) which allows for more interaction time with the wetland ecosystem.

The impact of rainfall and evapotranspiration on the treatment process is critical. High faecal coliform removal, for example, is realised in wetlands when run at long retention times (Ottova *et al*, 1997). This trend was reflected in the results of this study (Chapter 5). Water flow rates in treatment wetlands also affects the nutrient cycling within the systems and the storage extent - short or long term. This was shown by the report of Platzer and Netter (1994), where increased ammonium removals (40-70%) were derived when the evapotranspiration rates were rising.

2.4.2 Water Balance terms

Hydrological characteristics and the water balance of a wetland are essential in determining the levels of performance that can be realised in a specific design. At the Kirinya pilot wetlands, infiltration into or out of the units was negligible due to the way of construction as described in section 2.1.3. The water balance of a wetland system may be expressed as equation 2.1

$$\frac{dV}{dt} = Q_i - Q_o + P - E \tag{2.1}$$

Where:

dV/dt	=	Change in water storage by wetland (rate of change of volume, (m^3/day)
V	=	Volume of water in the wetland (m^3)
t	=	the time (day)
Q_i	=	system influent flow volume (m^3/day)
Q_o	=	system effluent flow volume (m^3/day)
P	=	rainfall (m^3/day)
E	=	evapotranspiration (m^3/day)

2.5 Materials and Methods

(i) *Pond water temperatures*

Measurement of the prevailing water temperature in the ponds was effected using the multiple probes for pH, dissolved oxygen concentration or electrical conductivity determination. Measurements were carried out at different depths but at selected sites in each pond.

(ii) *Inflows and outflows*

Wastewater inflow was measured using a bulk meter while outflow volumes were deduced from water levels measured and read off from the corresponding calibration graphs as described in section 2.2.1 and 2.2.2.

(iii) *Rainfall*

A simple rain gauge (garden type, 35 mm capacity) was used in the measurement. The gauge was suspended on a stick rod at a height of 1 m in an open section of the wetland. After each rain event, the readings were taken. However, during heavy storms, the gauge volume capacity would be surpassed and comparative rainfall data had to be obtained from Kimaka airstrip meteorological station.

(iv) *Water storage changes*

The wetland water volumetric changes were monitored by measurement of the pond water level height changes on each feed cycle during each of the operational phases. In the first three months of commencement of experimental measurements, water levels were measured twice a day at 8.00 am and 5.00 pm but was thereafter changed to daily monitoring between 900 - 1200 hrs to coincide with sampling time. It involved measuring the water levels before and after loading of wastewater using a calibrated steel rod at the mid- pond length positions. The corresponding volumes for the measured water levels in each pond were derived from the water level - volume calibration graphs (section 2.2.1).The change in wetland storage volume was obtained by subtracting volumes deduced after loading and before the next loading.

2. 6 Results

(i) *Temperatures*

The temperature of water in the planted ponds was nearly constant throughout. In *Cyperus papyrus* and *Phragmites* wetland units, water temperatures ranged from 22 ° C to 25 ° C but in the open sections and the large open ponds, a wider temperature range (24 ° C to 29 ° C) prevailed. Temperature recorded from the different ponds of *Phragmites* and *C. papyrus* systems did not show any significant difference ($p > 0.05$). However, all showed significant difference with the temperature obtained in the large open ponds. Over the experimental period, air temperatures (data from Kimaka station) ranged from 16 ° C to 30.5 ° C and relative humidity ranged from 53 % to 82 %.

(ii) *Wetland Inputs and Outputs*

Wastewater inflows into the wetland were rather constant but out - flows were very variable on each cycle depending on the losses and rainfall. Summary data for each ponds during operational phases 1 & 2 are given in Tables 2.4 and 2.5.

The annual rainfall data from Kimaka Meteorological station for 1996, 1997 and 1998 were 1570 mm, 1410 mm and 1207 mm respectively. The monthly patterns measured at Kirinya for the periods May 96 to March 97 and from November 97 to March 1998 are represented in Figs. 2.4 and 2.5 for phases 1 and 2. (Note is made of the exceptional high rains that prevailed between October - December 1997 as compared to the other years).

(iii) *Wetland storage and evapotranspiration*

The changes in wetland storage showed a very strong variability on daily or weekly basis but reduced when data were considered over a month's period. Summary data for each of the systems is given in Tables 2.4 and 2.5. Evapotranspiration rates in each pond were computed using the water balance equation (2.1) on each respective day or week of measurement. Data used for computation in each case was based on the day or week values. Daily or weekly evapotranspiration (ET) rates showed very high fluctuations but tended to normalise when data were considered over a long time of measurement.

The discrepancy in the water balance depicted in Tables 2.4 and 2.5 is a result of the residual water volumes remaining stored in the wetland that could not be measured using the method used especially in the vegetated wetland units. This variation could also be linked to leakage from the ponds, which was very noticeable in pond 7 in the second phase. The monthly variation in different systems over the measured periods in 1996 and 1997 are illustrated in Figs. 2.4 & 2.5.

Table 2.4 Summary data of Inputs and outputs to each of the wetland units during phase 1 operation (May 96 to March 97). Ponds 1, 2, 5 & 6 had *Cyperus papyrus*, ponds 7 & 8 had *Phragmites mauritianus*, ponds 3 & 4 were controls without plants.

Pond	Influent Volume (m^3/day)	Effluent Volume (m^3 /day)	Rainfall (m^3/day)	Evapotranspiration	
				(m^3/day)	mm/day
1	1.86	1.69	0.13	0.28	6.85
2	1.64	1.60	0.13	0.20	4.98
3	0.96	0.93	0.13	0.14	3.50
4	0.51	0.48	0.13	0.14	3.63
5	1.45	1.28	0.13	0.27	6.60
6	1.05	0.96	0.13	0.24	6.01
7	0.94	0.80	0.13	0.25	6.33
8	0.52	0.41	0.13	0.18	4.54

Table 2.5 Summary data of Inputs and outputs to each of the wetland units during phase 2 operation (October 97 to March 98).Ponds 1, 2, 5 & 6 had *Cyperus papyrus*, ponds 7 & 8 had *Phragmites mauritianus*, ponds 3 & 4 were controls without plants.

Pond	Influent Volume (m^3/day)	Effluent Volume (m^3 /day)	Rainfall (m^3/day)	Evapotranspiration	
				(m^3/day)	(mm/day)
1	0.90	0.90	0.28	0.24	5.97
2	0.90	0.92	0.28	0.25	6.02
3	0.89	1.04	0.28	0.12	3.02
4	1.43	1.57	0.28	0.14	3.54
5	1.42	1.44	0.28	0.24	5.79
6	1.50	1.43	0.28	0.24	5.92
7	1.50	1.49	0.28	0.20	5.11
8	1.49	1.48	0.28	0.25	6.36

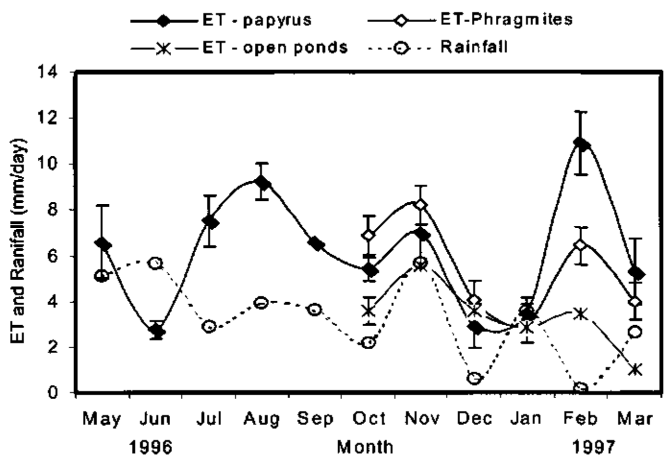

Fig. 2.4 Rainfall and evapotranspiration (ET) rates in papyrus, *Phragmites* and open ponds systems in phase1.

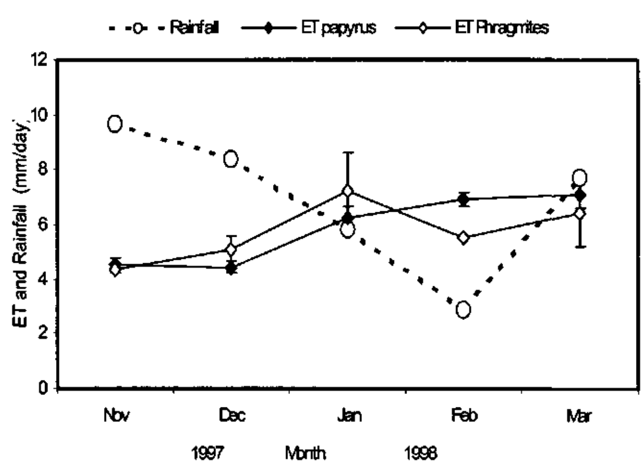

Fig. 2.5 Rainfall and evapotranspiration (ET) rates in papyrus, *Phragmites* and open ponds systems in phase 2.

The mean evapotranspiration rates (ET) that were determined over a total period of 15 months (papyrus systems) and 11 months (*Phragmites* and control ponds) were as follows: papyrus, 6.1 mm/day; *Phragmites*, 5.6 mm/day and the open ponds, 3.8 mm/day. The evapotranspiration rates of individual planted ponds were not significantly different from each other ($p = 0.789$). However, the evapotranspiration rates in the planted systems showed significant difference with ET rates determined for the control open ponds ($p = 0.03$).

2.7 Discussion

The water temperature in the planted wetlands was nearly equal to the average air temperature (22.5 ^{0}C). The strong correlation between the two was expected since the influencing variables, namely wind speed, relative humidity and solar radiation apply in both cases. Empirical predictions from data of 15 wetlands (Kadlec and Knight, 1996) showed similar correlations. The effect of temperature on reaction rates is a well explained phenomenon that is expressed by Arrhenius and Van't Hoffs equations (Brezonik, 1994). Within the physiological limits, an increase in temperature increases rates of biological processes such as the ones taking place in treatment wetlands. The readily available solar energy in the area where this research was done maintained relatively higher temperatures that in turn greatly influenced the rates of breakdown of pollutants in the wetlands without the need for extra external input of energy (chapters 3 to 5). Since the water temperatures at Kirinya were not subject to seasonal variability like in the case for temperate wetlands, the rates of biological transformations are expected to have been nearly steady all year around.

There are no data in literature on evapotranspiration from *Phragmites mauritianus* plants. However, evapotranspiration rates ranging between 1.4 - 6.9 mm/day, were obtained from investigations carried out over the whole growing season in temperate environments using *Phragmites australis* (Smid, 1975). Higher evapotranspiration rates ranging from 6.9 to 11.4 mm/day were derived from separate studies conducted in summer (Ondok *et al,* 1990). The evapotranspiration rates derived from *Phragmites mauritianus* in this study (Table 2.4 & 2.5) could be comparable to the summer ET rates for *P. australis* in temperate environment. The evapotranspiration rates obtained from papyrus plants during this study (average 6.1 mm/day) are similar to the findings reported by Rijks (1969) in the Namulonge natural papyrus swamp, in Uganda. The evapotranspiration rates (ET) were 1.2 times greater than the values obtained by Kansiime and Nalubega (1999) from their recent studies in the Nakivubo wetland dominated by *Cyperus papyrus* and *Miscanthidium violaceum* plants. The ET rates are half of what was reported by Jones and Muthuri (1984) (12.5 mm/day) from their experiments in a fringing papyrus swamp on Lake Naivasha. The results of the latter authors were based on a one day experimental measurements. But given the prevailing seasonal cycles in the region, the validity of their results as representative values for these type of plants is questionable. The variance between the measured evapotranspiration rates with the potential open water evaporation (4. 1 mm/day from Kansiime and Nalubega, 1999), may be a result of the so called 'clothesline effect' (Kadlec and Knight, 1996) that is associated with very small wetlands. However, because of the proximity of the Kirinya pilot wetlands to a large natural wetland and Lake Victoria, observed increases of evapotranspiration rates as a result of this effect, were moderated.

The evapotranspiration rates derived from *C. papyrus* and *Phragmites* systems were not significantly different ($p = 0.34$) despite the difference in their leaf structures and surface areas. This suggests that type of vegetation in the wetland, may not be an important factor in determining the evapotranspiration rates. Similar conclusions were echoed by Bernatowicz *et al* (1976) and Koerselman and Beltman (1988) from their investigations on evapotranspiration rates in different reed species.

Evapotranspiration rates were on average higher than rainfall by a factor of approximately 1.2 and overall accounted for from 15 to 20 % water loss in all the wetland systems. Meuleman (1999), in a study of a *Phragmites australis* wetland in the Netherlands, reports evapotranspiration losses of up to 13 % in the summer months. Kansiime and Nalubega (1999) obtained water losses through evapotranspiration in the Nakivubo wetland of only 5 %. This low value in comparison to the Kirinya one, was probably due to the water contribution or influence of the lake seiches. The relatively higher evaporative losses obtained from the Jinja-Kirinya wetlands may be attributed to the small size of the wetland and influence of advective fluxes from the walls of the ponds.

2.8 Conclusions

Each of the operational modes adopted in the Kirinya pilot wetland generated sufficient data for assessing it suitability for large scale application. The impacts of each of the configurations on the removal of different pollutants is assessed in chapters 3 to 5 and the ultimate choice recommended for the routine application is defined in each of these chapters.

The general conclusions drawn from the hydrological measurements include the following:

(a) The direct inflow - outflow measurement method applied in the evapotranspiration rates determination was adequate since the values obtained were in agreement with those derived from computational methods commonly used. More credence is given to the rates determined in this study compared to previous reports because of the many wetland units used in the investigation and the results obtained were reproducible over the entire period of experiment of more than two years.

(b) The evapotranspiration rates were not significantly influenced by the type of the macrophyte but probably the local climatic conditions seem to have been the controlling factor.

(c) The proximity of the pilot wetland to a natural wetland and L. Victoria minimised the relatively high evaporative losses associated with wetlands of its size.

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APPENDIX

Plate 1 The initial stages of the pilot wetland construction

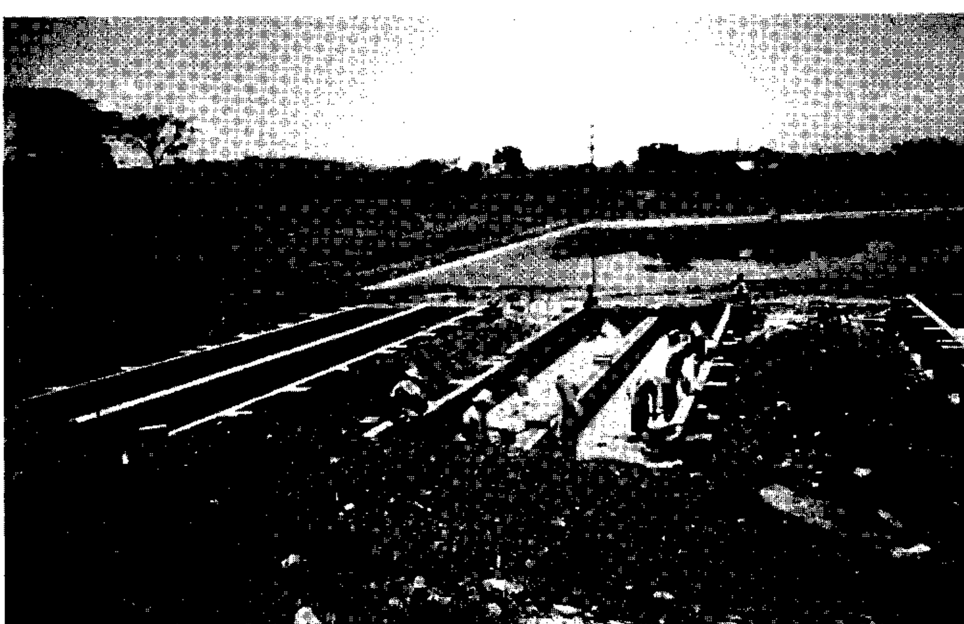
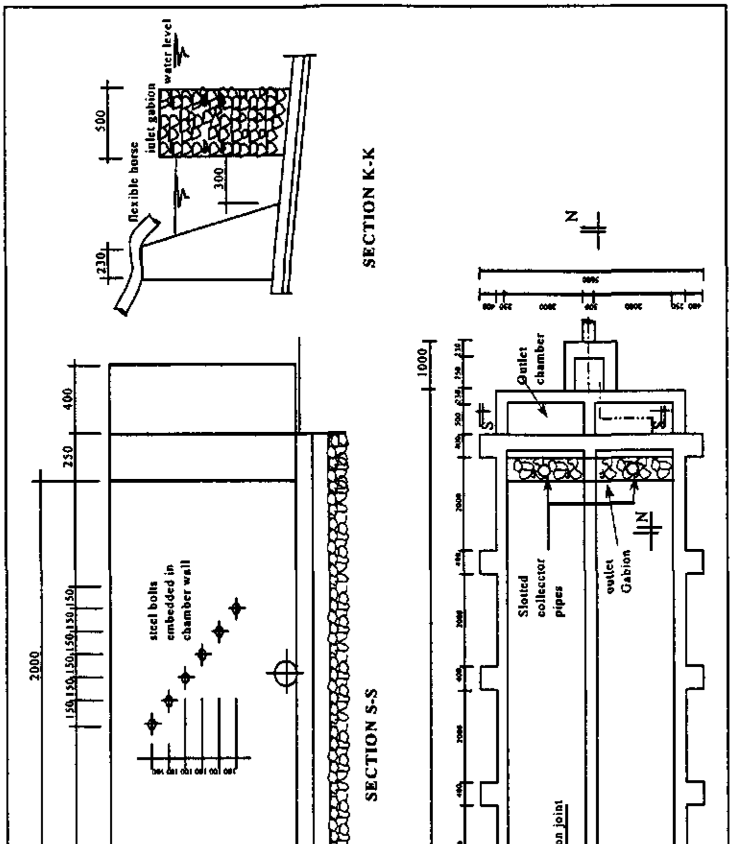


Plate 2 The final stages of the pilot wetland construction



A . Typical details of a single constructed wetland units at te Kirinya pilot wetlands, in Jinja.

Chapter 3

Organic Matter and Suspended Solids Removal

Abstract 47

3.1 Introduction 47
3.1.1 Processes 49
3.1.2 Scope of the study 50

3.2 Materials and Methods 51
3.2.1 Sewage characteristics 51
3.2.2 Environmental parameters 51
3.2.3 Algae 51
3.2.4 Organic matter and suspended solids 51
3.2.5 Data analysis 52

3.3 Results 52
3.3.1 General Characteristics 52
3.3.2 Environmental parameters 53
3.3.3 Removal of organic matter and suspended solids 57
3.3.4 Residual BOD 63

3.4 Discussion 64

3.5 Conclusions 66

References 67

Abstract

The capacity of the Kirinya constructed wetland to remove organic matter (BOD and COD) and suspended solids (TSS) was determined in the three operational phases of this study. The impact of the prevailing environmental parameters in each phase on the treatment performance were evaluated. In all the fully vegetated *Cyperus papyrus* and *Phragmites mauritianus* wetland units, dissolved oxygen was consistently less than 2 mg/l but with a steady pH of 7.5. Dissolved oxygen concentration of up to 27 mg/l and pH of 10.4 were recorded in the non-vegetated ponds. Removal efficiencies > 70% were attained for BOD and COD, and > 80 % for TSS in the individual vegetated wetland units. An increase in removal efficiency of BOD and COD > 80 % was obtained when open and vegetated wetland units were combined in a serial arrangement during the third phase. Reduced removal efficiencies ranging between 25 to 45 % for BOD and COD, and a net increase of TSS instead, were realised in the non-vegetated ponds. A linear relationship between organic loading and removal rates was realised even at maximum applied load rates of 350 kg ha^{-1} day^{-1} for COD, 100 kg ha^{-1} day^{-1} for BOD and 150 kg ha^{-1} day^{-1} for TSS in the papyrus wetlands units. In all the three phases, the effluent BOD, TSS and COD concentrations from the vegetated wetland units were less than the limits set in the Uganda standards for discharge of effluent into water and land. Residual non-zero background BOD concentrations of 12 mg/l and 17 mg/l, and BOD areal rate constants of 0.084 m/day and 0.039 m/day were derived from *Cyperus papyrus* and *Phragmites mauritianus* wetland units, respectively.

3.1 Introduction

Organic matter and suspended solids are the base pollutants present in municipal and industrial wastewater with a potential of causing deleterious effects to public health and the environment. Several methods have been applied in conventional wastewater treatment technologies to remove these pollutants (Metcalf and Eddy, 1991). In the treatment wetlands, the processes and factors that govern the reduction of these pollutants have also been a subject of intense research in the past 15 years although mainly, in temperate climate regions. This is demonstrated by several articles from specialised conferences in treatment wetland technology (Reddy and Smith, 1987; Hammer, 1989; Cooper and Findlater, 1990; Moshiri, 1993 and Vymazal *et al.*, 1998). Results from most of these studies identify oxygen and pH as critical environmental parameters that influence the treatment processes in wetlands.

Previous studies show that organic matter degradation mediated by heterotrophic bacteria is a faster metabolic reaction but is an oxygen demanding process. Reduced levels of oxygen tend to inhibit the functioning of these bacteria and may consequently affect the extent to which organic matter (biochemical oxygen demand, BOD or chemical oxygen demand, COD) is removed by the wetlands systems. Reports by Armstrong and Armstrong (1988), Brix (1990), Sorrel and Boon (1992) reveal that aquatic plants translocate oxygen from the shoots via their well-developed aerenchyma to the rhizosphere. Kemp and Murray (1986) and Caffrey and Kemp (1991) determined the oxygen release rates of *P. australis* ranging from 0.5 - 5.2 g m^{-2} day^{-1}. Kansiime and Nalubega (1999) estimated oxygen release rates of 0.017 g m^{-2} day^{-1} by *C. papyrus* plants. However, Wetzel (1993) and Kadlec and Knight (1996), point out that due to the respiratory demands of microbiota within the rhizosphere, most, if not all the oxygen that leaks out from the roots is consumed immediately by these communities. Therefore, it cannot meet the external demands imposed by loaded wastewater. This in effect, limits the oxygen supply to the wetland systems via this route. Other natural alternative routes by which oxygen may be supplied to the wetlands includes mass transfer across the water-air interface which is described by Fick's law of diffusion and Henry's law of equilibrium distribution between a liquid and gaseous phase (Brezonik, 1994). Another route is photosynthesis by periphyton and phytoplankton when water is not shaded. Direct aeration has been used in some wetland systems as well. The alternative routes were exploited in the small and large open zones created in the wetlands during this study.

Several mechanisms that are responsible for the removal of organic matter and suspended solids in wetlands are reported in literature. BOD and COD reduction is influenced by both, the complex microbial communities present in the wetland as well as the physical structure of the rooting biomass. Wood (1990), suggested that particulate organic matter (POM) removal in the wetlands was controlled by processes related to the structure of the rooting media; these may include: settling, impaction, filtration and predation. Cooper *et al* (1996) and Wanner (1997) have further shown that the dissolved organic matter (DOM) component of wastewater is degraded in the wetlands by both aerobic heterotrophic bacteria and anaerobic autotrophic bacteria. These bacteria are usually found attached to the plant and other solid matrix structures in the wetland.

The dominating degradation pathway is defined mainly by the availability of oxygen. The aerobic pathway is significant when oxygen is available. In the absence of oxygen or under reduced oxygen concentrations, anaerobic decomposition reactions become the major BOD decay routes. The main anaerobic reactions involved as outlined by Mistch and Gosselink (1993) include: fermentation resulting into production of acids and ethanol; methanogenesis yielding methane; sulfate and iron reduction producing hydrogen sulphide and iron(II) and nitrate reduction through the denitrification process. The low residual oxygen concentrations that prevailed in the vegetated wetlands during this study indicated that BOD reduction via the anaerobic pathway was significant.

The other factors that may influence the BOD degradation pathway is suggested to be the physical environmental condition in the wetland such as pH, temperature and salt concentration (Portier and Palmer, 1989). Poindexter (1971) further indicates that the rates of metabolism in both aerobic and anaerobic environments will be governed by the availability of nutrients.

Sedimentation is the principle mechanism that is identified with removal of suspended solids (Stowel *et al.,* 1981; Watson *et al.,*1989; Wood, 1990). Other subsidiary removal routes include microbial metabolism of colloidal solids, electrostatic interaction with electric charges that are associated with plant roots (Wolverton, 1989) and predation by zooplankton such as *Daphnia* (Gearheart and Higley, 1993). All these processes are said to be augmented by the quiescent conditions and long retention times of wastewater in the wetland. Furthermore, the wetland vegetation causes water to flow in and around obstacles, voids etc which increases retention time and provides sites where organic and colloidal matter, and suspended solids settle out through chemical and auto flocculation. In this respect, the type of plant vegetation used and their growth characteristic, can influence the contribution of this pathway.

Retention of suspended solids by the wetland also provides a collateral advantage in that most pollutants, such as particulate BOD, heavy metals and faecal coliforms which partition more in the suspended solid matrix get removed as well. In this study, the papyrus plants in particular, developed a dense and interwoven root-mat structure that augmented the total suspended solids (TSS) removal by the latter process

3.1.1 Processes

The chemistry of BOD reduction in treatment wetlands for continuous surface flow (SF) and subsurface flow (SSF) is described by many authors as a first order kinetic process (Tchobanoglous, 1987; Reed *et al.*, 1988; EPA, 1988; Cooper *et al.*, 1996). Consequently many constructed wetland systems are designed on the basis of the mathematical model based on a first order equation (3.1) while assuming plug flow conditions. As earlier stated in section 2.1.1 (Chapter 2), the equation has been extended to describe organic decay processes in batch fed wetlands as well given the complex nature of these systems. These equations are therefore used to explain the organic processes in all the three modes of operation applied in this study too.

$$\frac{C_0}{C_i} = Exp\ (-\ k_v.\ t) \qquad (3.1)$$

The equation can be modified by substituting (ε. h. A/Q) for t giving the equations:

$$\frac{C_o}{C_i} = Exp\left(\frac{-\ k_V\ .\varepsilon\ .h\ .A}{Q}\right) \qquad (3.2)$$

Or

$$\frac{C_o}{C_i} = Exp\ \left(\frac{-\ k_T\ .A}{Q}\right) \qquad (3.3)$$

Where:

C_o	=	mean effluent BOD (mg/l)
C_i	=	mean influent BOD (mg/l)
k_v	=	volumetric first order rate constant (day^{-1})
t	=	hydraulic residence time (days)
ε	=	wetland void fraction
h	=	water depth (m)
A	=	surface area of wetland (m^2)
Q	=	average daily flow rate (m^3/d)
k_T	=	areal first order rate constant (m/d)

The void fraction and the water depth in the wetland vary with time; equation 3.3 is therefore used in all the relevant computations in this text.

The above equations indicate an exponential decrease in the influent BOD concentration eventually to zero. However, based on the field data collected from very many constructed wetlands in North America, it has been demonstrated that BOD does not exponentially decrease to zero but rather, it approaches to a non- zero background concentration (Kadlec and Knight, 1996).This concentration represents the amount of soluble organic matter generated by the wetland systems themselves through their many internal carbon production mechanisms. This in effect tends to reduce the extent of BOD removal that can be obtained in treatment wetlands with respect to the loaded organic matter (Wetzel, 1993). The non-zero BOD value could be of profound significance to environmental regulators.

Modification of equation 3.3 to include the contribution of the non- zero background BOD is represented in equation 3.4

$$\ln\frac{(C_o - C^*)}{(C_i - C^*)} = -\frac{k_T}{q} \qquad (3.4)$$

Where: C^* = the non - zero background BOD (mg/l)
k_T = 1st order areal rate constant (m/d)
q = hydraulic loading rate (m/d)

The product of the rate constant and the non zero BOD concentration is taken as the rate of BOD generation by a wetland, r (g m^{-2} d^{-1})

$$r = k_T . C^* \qquad (3.5)$$

3.1.2 Scope of the study

Many articles cited in literature show that constructed wetland systems are effective in BOD, COD and suspended solids removal from municipal sewage as well as industrial and agricultural wastewater, land fill leachate, acid mine water and urban storm runoff. Data generated from constructed wetlands mainly using plants such as bulrush (*Scirpus sp*), cattails (*Typha latifolia*) and common reed (*Phragmites australis)* in Europe (Vymazal *et al.*, 1998) and in the USA (Knight *et al.*,1993) generally indicate BOD and suspended solids treatment efficiency ranging between 60 - 95%. Okia (1993) and Bruggen *et al* (1994) report BOD reduction of 80% from experiments carried out using a laboratory scale constructed wetland planted with *Cyperus papyrus* in a greenhouse in Delft, The Netherlands.

However, application of constructed wetlands technology in the tropical or equatorial environments is very limited and data on performance assessments for organic matter and suspended solids is scanty (Denny, 1997). Moreover, the prevailing temperatures in these environments are steady throughout the year and are not subject to fluctuations induced by climatic changes such as in the treatment wetlands located in the temperate regions. These conditions would be expected to accelerate microbial mediated processes such as organic matter breakdown in the wetlands. The overall indication is that the tropical wetlands may have the potential for use in removing the bulk pollutants from wastewater. This study was therefore designed with the following specific objectives:

(a) to determine the influence of the prevailing environmental factors under the different operating conditions on the treatments levels for organic and suspended matter;
(b) to determine the effect of the various modifications made in systems configurations on removal of BOD, COD and TSS and
(c) to determine the influence of different operating parameters (hydraulic loading rates and hydraulic retention time) on the extent of removal of these pollutants.

To achieve the stated objectives, the following activities were carried out:

i) on site measurements of the environmental parameters at the defined positions and depths during all the three operational phases.

(ii) measurement of BOD, COD and TSS in samples taken from the different sites and ponds during the three phases

3.2 Materials and Methods

Throughout the period of this study, APHA (1992) procedures for sampling and analysis as described in Chapter 2, were followed. The National Water and Sewerage Corporation water quality procedure manuals (NWSC, 1997) were followed for the routine work.

3.2.1 Sewage characteristics

The composition of the raw sewage received at the Jinja Kirinya Sewage Treatment Works was obtained from the database of NWSC for Jinja Area spanning a period of 4 years. In addition, data were obtained from reports from previous studies by Gauff and Mott MacDonald consultants during the rehabilitation study (Gauff, 1988; Mott MacDonald, 1996). Laboratory measurements were carried out for samples taken during major storm events since measurements for such events were not in the database, but were essential for this study. The general characteristics of the influent wastewater loaded into the ponds was obtained from measurements as described in later sections below.

3.2.2 Environmental parameters

Three key environmental parameters, namely dissolved oxygen (DO), pH and temperature were determined both on site as well as in the laboratory. Measurements were carried out during each operational phase at the locations specified in Chapter 2. Multi parametric portable pH meter (3071 Jenway) and Oximeter (WTW 323) fitted with temperature sensors were used. They were always re-calibrated at the wetland site prior to use. Occasionally, oxygen was fixed during sampling as per the modified alkaline azide method and later titrated in the laboratory to determine DO concentration. This method was also used for calibration of oxygen meters.

Two overnight field campaigns were undertaken to obtain one hour interval diel measurements at the specific sites along the pond (transect) since continuous automatic monitoring was not feasible. Vertical gradients in the pond water columns were investigated only in phase 1 and by the time when the plants had become well established. Probes were attached to a calibrated stick and immersed to 2.5 cm from the bottom and 2 cm from the surface. Samples for other measurements were drawn from the same depths by a suction pump.

3.2.3 Algae

The amounts of algae present in different wetland systems was determined by measurement of chlorophyll-a concentrations. Water samples for the experiments were taken from mid-pond lengths of the control (open) and in the fully plant covered ponds. Additional samples were taken at the mid-point of the open zone in the partially vegetated ponds during the second phase (Chapter 2). A detailed procedure as outlined in APHA (1992) was followed.

3.2.4 Organic matter and suspended solids

Analyses of organic matter and suspended solids in samples taken from the different sampling sites (Chapter 2), followed procedures given in APHA (1992) and the HACH instrument manual. BOD_5 was determined after incubation at 20 °C for 5 days; a YSI model 5750 DO meter was used in oxygen measurements. COD was determined by a reflux method using a HACH (model 45600) block digester and a CECIL CE 1000 series spectrophotometer at 420 nm wavelength. Total suspended solids were determined by the gravimetric method.

3.2.5 Data analysis

All data generated from this study were analysed using EXCEL spreadsheet and MINITAB Release 10 for Windows software. Comparison of variables was performed using analysis of variance technique (ANOVA) and using Tukeys multiple comparisons to test differences in the means. A GLOBE calibration model was used for calibration and validation of in- and out-BOD data.

3.3 Results

3.3.1 General Characteristics

The typical characteristic ranges of raw sewage received at the Jinja Kirinya Sewage Works and effluent from the different units are given in Table 3.1. (Source: National Water and Sewerage Corporation Water Quality quarterly reports, Jinja Area 1995 - 1999). The raw sewage values are identifiable with the very strong sewage category. The strength is however, reduced by almost ten-fold in the anaerobic lagoons to the extent that the influent loaded to the wetland units was of moderate strength.

Table 3.1 Concentration ranges of wastewater taken from different lagoon units of the Jinja Kirinya Sewage Works (from 3 yr data base).

Parameter	unit	Raw wastewater	Anaerobic pond effluent	Facultative pond effluent	Maturation No.1 pond effluent	Final discharge effluent
COD	mgl^{-1}	900 - 1800	81 - 216	70 - 120	80 - 115	100 - 130
BOD	mgl^{-1}	200 - 800	30 - 136	50 - 66	33 - 55	40 - 78
TSS	mgl^{-1}	280 - 600	40 - 150	53 - 80	60 - 85	80 - 120
NH_4 N	mgl^{-1}	13 - 33	23 - 75	12 - 21	12.2 - 16.1	6.4 - 9.4
NO_3 N	mgl^{-1}	0.0 - 0.03	0.0 - 0.6	0.04 - 0.5	0.9 - 1.6	1.2 - 2.6
PO_4	mgl^{-1}	0.9 - 4.5	0.15 - 3.4	3.1 -3.9	2.8 - 3.6	1.3 - 3.7
EC	µS	1500 -2000	740 - 1300	730 - 1050	600- 700	510 - 608
pH		6.5 - 7.8	6.9 - 7.6	7.2 - 7.8	7.5 - 7.9	7.8- 8.5

3.3.2 Environmental parameters

Phase 1

This was the exploratory phase run between May 96 - May 97. Wetland units were loaded batch-wise on a weekly basis and no plant standing or rooted biomass was yet removed from either papyrus nor *Phragmites* ponds (Chapter 2).

The dissolved oxygen concentrations, pH and temperatures ranges for this phase are given in Table 3.2. Vertical profiles of these parameters measured in all the ponds did not show significant variance at different depths (maximum depth 45 cm in papyrus ponds) ($p > 0.05$), suggesting well mixed water columns. The lower limits given in the data ranges were derived when the plant density cover was nearly 100 %, while the upper limits represented low shoot density situation. In the control (open) ponds, the lower limits of pH and DO represent day measurements at commencement of the operation, while the upper limits were derived at later stages when algal blooms prevailed in both ponds. Dissolved oxygen in the vegetated wetlands showed a strong correlation with shoot density cover ($R^2 = - 0.92$).

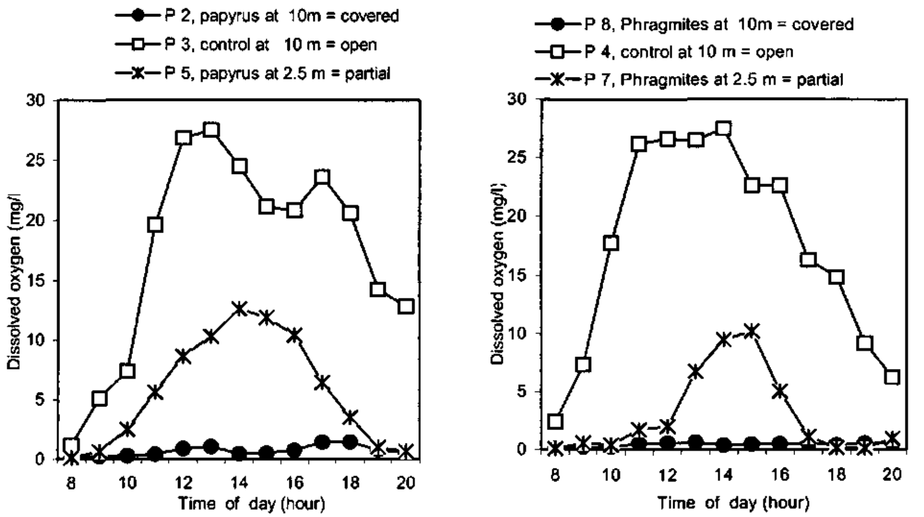
Table 3.2. Data ranges of the environmental parameters measured from May 96 - Mar 97 from the different pond systems.(day measurements).

System	Dissolved oxygen (mg/l)	pH	Water temperature (°C)
Cyperus papyrus	0.2 - 5.4	6.5 - 7.6	22 - 28.5
Phragmites mauritianus	0.5 - 6.2	7.4 - 8.7	25 - 29.2
Control (open)	4.4 - 18	8.6 - 10.3	25 - 30.1

Phase 2

In this phase (September 97 - March 99), three ponds (P 2, P 6 & P 8) were completely covered with vegetation but three ponds (P 1, P 5 & P 7) were partially vegetated after the removal of plant biomass from the first five metres (10 m^2) from influent. (Chapter 2). The control ponds remained as in Phase 1. Weekly and daily batch wastewater load formats were applied.

Results from the diel measurements carried out once showed very high dissolved oxygen concentrations and pH in the control ponds. (Continuous automatic measurements of these physical parameters was not feasible in the circumstances that prevailed). Supersaturated DO of up to 27 mg/l and pH values of 10.5 were recorded (measured at mid-pond position) between 13.00 - 15 .00 hrs (Figs. 3.1 and 3.2). Dissolved oxygen and pH measured in the open zones (2.5 m from the gabions) of the partially vegetated ponds were comparatively lower; maximum DO and pH values were 12 mg/l and 8, respectively.

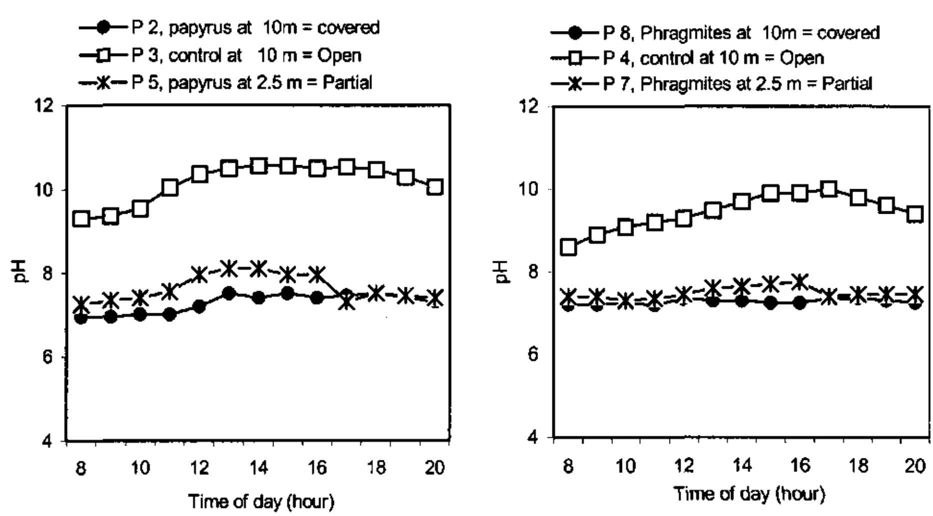


(a) C.papyrus systems

(b) *P. mauritianus* systems

Fig.3.1 a and b Dissolved oxygen concentration diel trends (January 1998) measured at the mid-positions of the control and the fully planted wetland units, and at the middle of the open zone in the partially planted wetland units.

In the completely vegetated ponds, DO and pH remained virtually constant as in the previous phase (Figs. 3.1 & 3.2) but with temperature showing strong positive correlation with pH.



(a) *C. papyrus* systems

(b) *P. mauritianus* systems

Fig.3.2 a and b pH diel trends (January 1998) measured at mid-positions in the control and fully planted wetland units, and at the middle of the open zone in the partially planted wetland units.

The amounts of chlorophyll-a in the different pond systems is illustrated in Fig. 3.3. Significantly higher chlorophyll-a concentrations were obtained in samples taken from the control (open) ponds than in either the open zones of the partially vegetated or in the fully plant covered ponds.

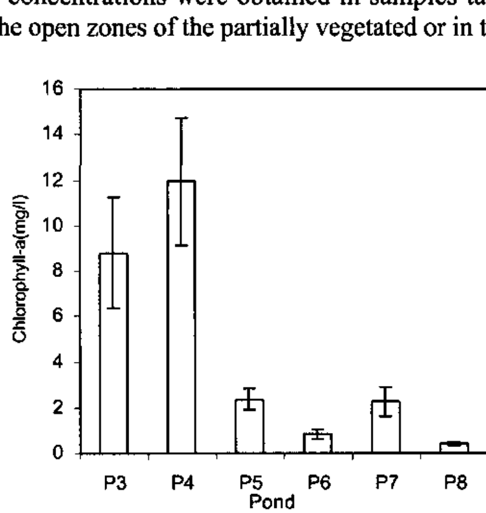


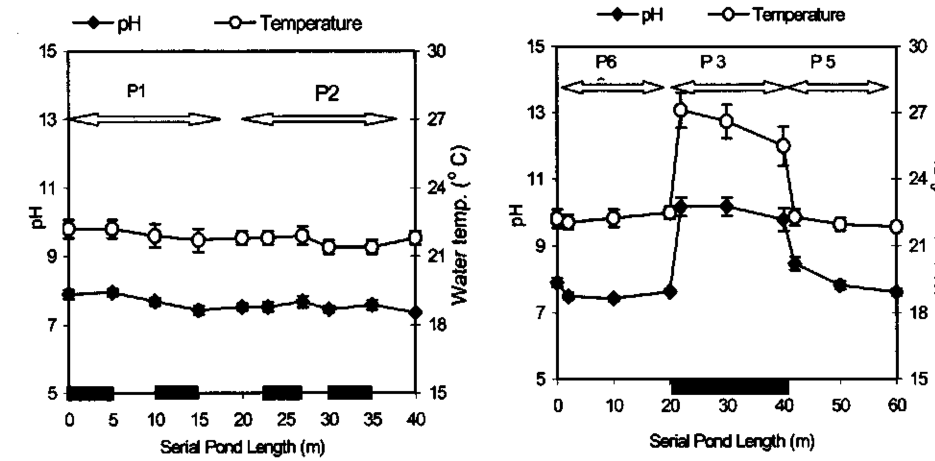
Fig. 3.3 Chlorophyll-a concentrations at the mid pond length of open ponds 3 and 4, fully planted ponds 6 and 8 and in the middle of the open zones in the partially planted ponds 5 and 7. Vertical bars are standard error of the mean of N =5.

Phase 3

During this operational phase (October 98 - March 99), *C. papyrus* ponds 1 and 2 were connected in series S 1-2 and had a configuration with alternating small open areas and planted areas. *Phragmites ponds* 8 and 7 were joined with open pond 4 to form series S 8-4-7 and papyrus ponds 5 and 6 were similarly joined with open pond 3 to form series S 6-3-5 (Chapter 2).

The trends of the environmental parameters observed in this series reflected the characteristics of the individual ponds of the series as was observed in phase 2. Data presented are for measurements carried out between 11.00 hrs to 13.00 hours. The temperature in the series S 1-2 remained relatively steady and uniform along the longitudinal profile (mean 22 °C), with no significant variations ($p > 0.05$) observed between the temperatures recorded in the open and the plant covered sections (Fig. 3.4 a). In the other series S 6-3-5 and S 8-4-7, significant increase in temperatures (by 5 °C) were observed in the large open sections (Fig. 3.4 b).

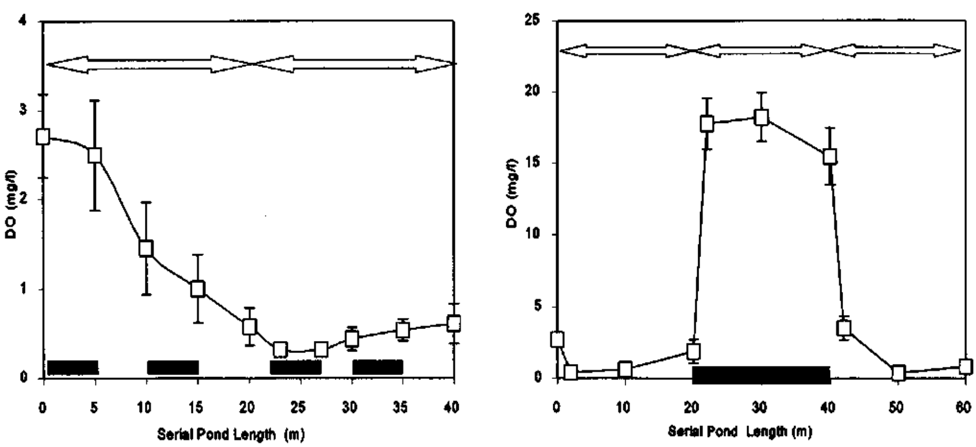
The pH value in the pond series S 1-2 was uniform (7.6 ±0.06) throughout the serial pond length (Fig. 3.4 a). However, in series S 6-3-5, circum-neutral pH was only observed in the plant covered sections but pH increased by nearly 2. 5 pH units in the middle of the open ponds (Fig. 3.4 b). This characteristic trend was also observed in series S 8-4-7.



(a) S 1-2 (b) S 6-3-5

Figs. 3.4 a and b pH and temperatures profiles along the pond length of papyrus series S 1-2 and S 6-3-5. Black sections indicate open zones without plants.(N = 20, Vertical bars = standard error of the mean).

The oxygen profile in series S 1-2 shows a decrease of the oxygen concentration from an average of 2.71 mg/l to the lowest value of 0.3 mg/l at 25 m series length before gradually increasing to 0.61 mg/l by the end of the series (Fig. 3.5 a). In series S 6-3-5, the oxygen concentration profiles were generally characterised by a sharp decline from 2.7 mg/l to 1.87 mg/l by the end of the first pond in the series. This was followed by an increase to a maximum of 18.2 mg/l within the open ponds before declining by the same amount in the last pond of the series (P 5) Fig. 3.5 b.



(a) S 1-2 (b)S 6-3-5

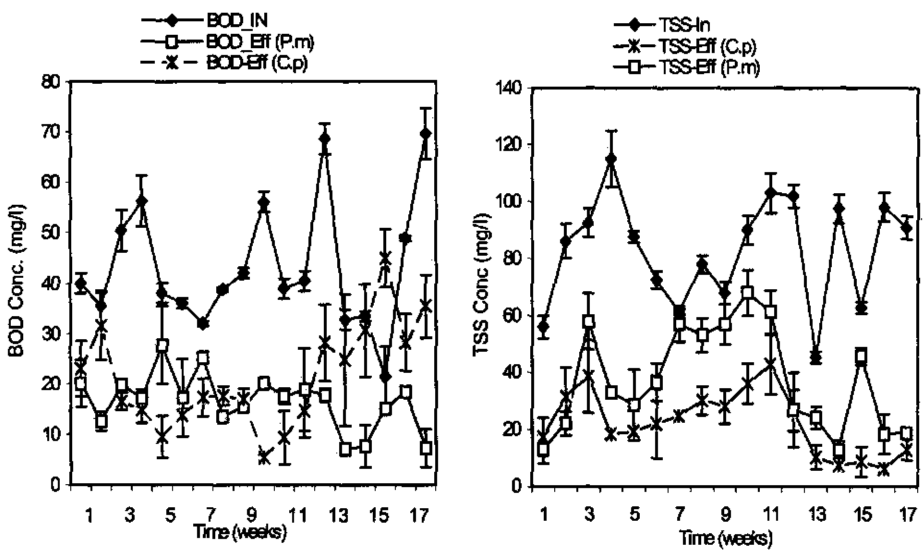
Figs. 3.5 a and b Dissolved oxygen (DO) concentration profiles along the serial pond length of papyrus series S 6-3-5 and S 1-2. Black sections indicate open zones without plants.(N = 22, vertical bars = standard error of the mean).

3.3.3 Removal of organic matter and suspended solids

Phase 1

(i) Concentration trends

The reduction of BOD, COD and suspended solids concentrations by the wetland systems was variable. BOD and TSS effluent concentration from papyrus units ranged from 5 - 45 mg/l and 6 - 62 mg/l respectively; 8 - 64 mg/l and 13 - 80 mg/l in *Phragmites* systems. In the control ponds, effluent BOD ranged from 17 - 58 mg/l and 24 - 56 mg/l in pond 3 and pond. Respective effluent TSS ranged from 33- 210 mg/l in pond 3 and 55 - 262 mg/l in pond 4. The influent concentrations of both parameters showed very strong fluctuation as illustrated in Figs. 3.6.



(a) BOD concenttration trends (b) TSS concentration trends

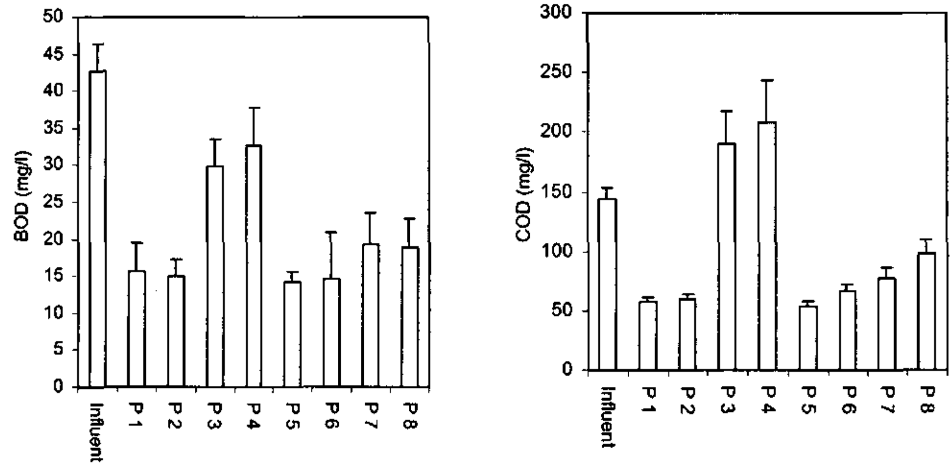
Fig. 3.6 a and b Concentration trends of BOD (a) and TSS (b) in *C. papyrus* and *Phragmites* wetland units as measured from November 1996 to March 1997.

The averaged influent and effluent concentrations of BOD, COD and TSS for each pond over the entire phase are illustrated in Fig 3.7 a -c. From this diagram it is clear that the effluent from the individual vegetated wetland units were significantly lower than in the control ponds. The lowest concentrations for all these parameters were obtained in the papyrus units. Pond 5 effluent did not show a significant variation from the others despite the different feed cycles.

(ii) Mass loading - removal relationships

The influence of the mass loading of COD, BOD and TSS on their removal rates in each of the ponds was evaluated graphically (Fig.3.8). Mass loading rates were computed by dividing the product of weekly volumes and influent concentrations by the actual number of days in between loading. The loading rates strongly correlated with the removal rates in both the *Phragmites* and papyrus systems; in the latter systems, $R^2 = 0.95$ for both COD and BOD and 0.87 for TSS.

Similar relationships were derived in *Phragmites* systems but no defined relationship existed in the control ponds.



(a) BOD (b) COD

Fig.3.7 a and b BOD and COD influent and effluent concentrations from different wetland units, vertical bars are standard errors of the mean of n = 40 except for P 5 with n = 75.

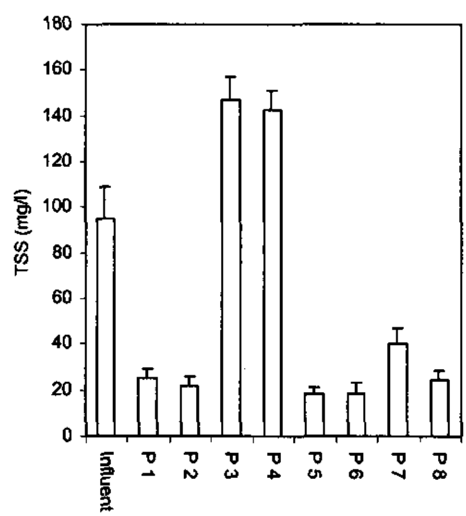
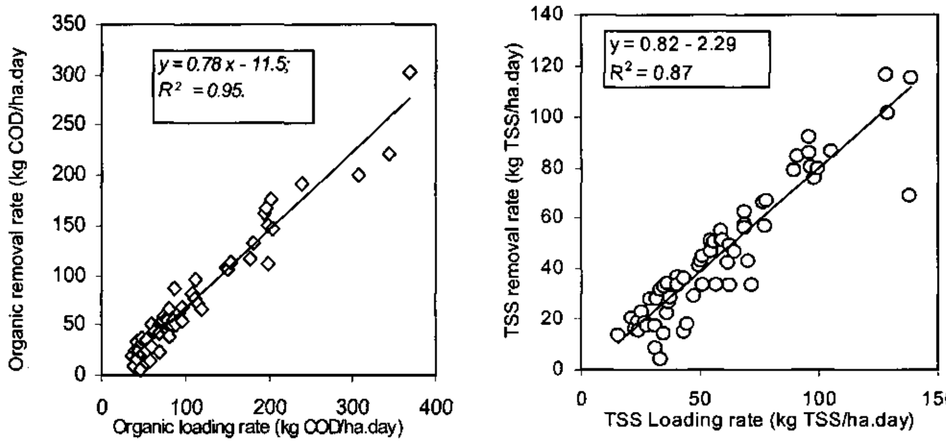


Fig. 3.7 c Total suspended solids (TSS) influent and effluent concentrations from different wetland units, vertical bars are standard errors of the mean of n = 30.



(a) COD (b) TSS

Fig. 3.8 a and b The effect of mass loading of COD and TSS on the removal rates in *C. papyrus* systems in Phase 1. (Data pooled from all the papyrus wetland units).

(iii) Effect of retention time and hydraulic loading

The influence of hydraulic loading rates (HLR) was evaluated graphically by plots of mean weekly HLR and the corresponding mass reduction of BOD in papyrus wetland units. Removal of BOD did not show a good correlation with hydraulic loading rate as shown by a low r^2 (Fig. 3.9 a) but it generally tended to decrease with increasing hydraulic loading rate with a maximum reduction was observed at HLR between 3 - 5 cm/day. The evaluation was not extended to *Phragmites* wetland units since they were operated over a narrow HLR range (between 1.3 to 2.8 cm/d) during this phase.

The actual optimal hydraulic retention time (HRT) under the operating conditions could not be determined because of the combined uncertainties of wetland porosity and variable depth. The nominal retention time is higher than actual but it was nevertheless used to gauge the impact on BOD removal under the batch feed conditions. Nominal retention time was computed using equation 3.6 for each feed cycle.

$$HRT = V_T \, x \, \frac{n}{V} \tag{3.6}$$

Where: V_T = total volume of pond at the operating maximum loads
n = feeding interval (days).
V = Volume of influent.

The retention times computed for each week in all the papyrus units were combined and the values obtained were plotted against the corresponding mass BOD reduction (Fig. 3.9 b). Maximum BOD reduction in papyrus systems were obtained at retention times between 4 -5 days; at longer retention times, the efficacy levels reduced and became constant.

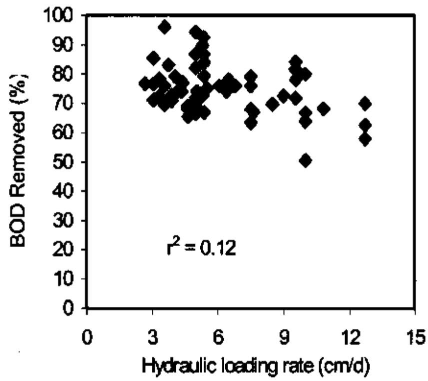


Fig. 3.9 a The effect of hydraulic loading rate BOD removal in *C. papyrus* systems in Phase 1 (pooled data).

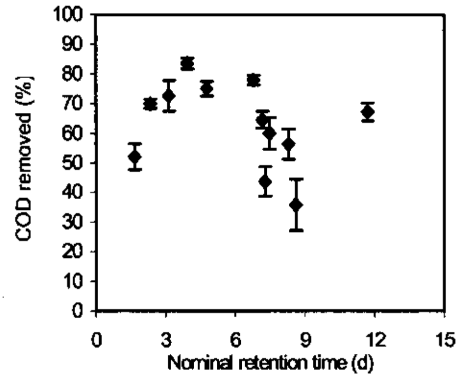


Fig. 3.9 b The effect of hydraulic retention on the time on COD removal in *C. papyrus* systems in Phase 1 (pooled data).

Phase 2

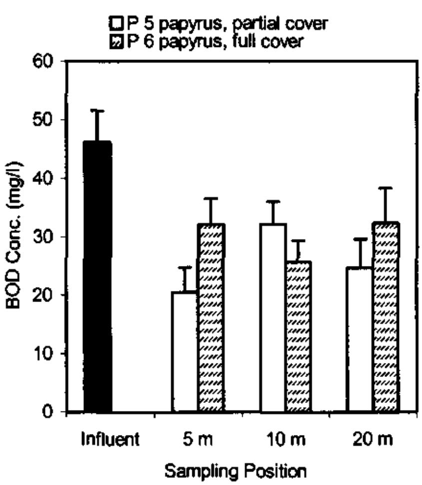
Experimental work in this phase was restricted to BOD; the main target for modifying the systems was on increasing ammonium removal rather than removal of organic matter or suspended solids. Consequently, the organic surface loading rates into the units were reduced by nearly 3-5 times. Surface organic loads of 10 and 14 kg BOD ha^{-1} day^{-1} were applied to ponds 1 -3 and 4 - 8 respectively.

In the first half of the study during exceptionally heavy rain period (Oct- Dec 97), BOD reduction was low (Table 3.3). This was associated with excess volumes of water in the pond at the end of the week feed cycle input by overflows through the surface of the pond (The drain flexible weirs in the eight units were maintained at the same level heights). As a consequence, reduced BOD mass removal was realised in these wetland units. Relatively, higher BOD removals were obtained in *Phragmites* and papyrus units that were operated at elevated volumetric loadings.

Table 3.3. Mean BOD concentrations in the influent and effluent and removal efficiency based on mass budgets during the first half of phase 2 operation (weekly load - drain format).

Pond	Influent volume (m^3 / wk)	Effluent volume (m^3 /wk)	BOD-In (mg/l)	BOD-Out (mg/l)	Reduction (%)
papyrus 1	5	6.07 ± 0.26	47.58 ± 4.7	32.52 ± 2.5	17.1 ± 4.6
papyrus 2	5	6.84 ± 0.51	47.58 ± 4.7	30.71 ± 5.1	11.8 ± 6.5
control 3	5	7.72 ± 0.77	47.58 ± 4.7	27.79 ± 7.7	9.8 ± 0.7
control 4	8	8.77 ± 0.58	41.12 ± 6.2	22.47 ± 3.9	40.1 ± 9.8
papyrus 5	8	8.30 ± 0.49	47.58 ± 4.7	21.95 ± 6.7	52.1 ± 7.2
papyrus 6	8	7.94 ± 0.20	47.58 ± 4.7	23.92 ± 5.8	50.1 ± 14.1
Phragmites 7	8	6.06 ± 1.11	41.12 ± 6.2	15.57 ± 4.4	71.3 ± 14.3
Phragmites 8	8	7.56 ± 0.96	41.12 ± 6.2	7.99 ± 3.5	81.6 ± 6.7

The effect of the created small open zones on the BOD removal during the 2nd half of phase 2 and when the rainfall decreased, is illustrated in Fig. 3.10. In both systems, the BOD concentration measured from units with open zones was lower than in the fully plant covered ponds. The improvement in the treatment performance in the partially vegetated wetland may be associated with aerobic BOD degradation since oxygen concentrations in these zones increased as compared to the other plant-covered sections (Fig. 3.1). As a result, in both *Phragmites* and *C. papyrus* systems, the overall BOD mass reduction measured was consistently higher in the wetlands where part of the vegetation was cleared (P 1, P 5 and P 7) than in the corresponding fully vegetated ones (P 2, P 6 and P 8) (Table 3.4).



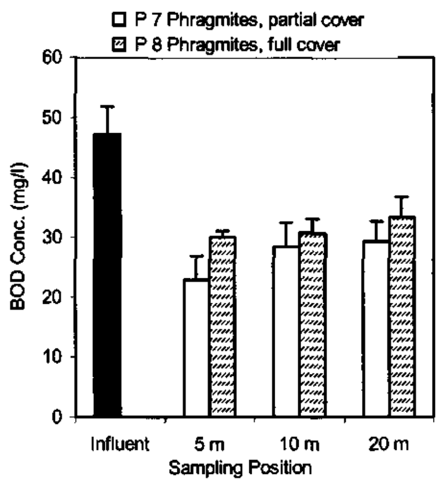


Fig.3.10 a and b BOD concentration at different position along the *C. papyrus* and *Phragmites* ponds which where partially vegetated (P 7 and P 5) and fully vegetated (P 8 and P 6) in Phase 2. Black section represents the open zone in P 5 & P 7. Vertical bars show the standard error of the mean of N = 50).

Table 3.4. Mean BOD concentrations in the influent and effluent and removal efficiency based on mass budgets during the 2nd half of phase 2 operation (daily load format).

Pond	Influent volume (m^3/d)	Effluent volume (m^3/d)	BOD-In (mg/l)	BOD-Out (mg/l)	Reduction (%)
papyrus 1	0.92 ± 0.1	0.61 ± 0.1	47.75 ± 4.5	24.82 ± 2.2	65.4 ± 5.2
papyrus 2	0.95 ± 0.1	0.70 ± 0.2	47.75 ± 4.5	23.68 ± 4.8	63.7 ± 7.4
control 3	1.00 ± 0.2	0.94 ± 0.2	46.42 ± 4.5	19.98 ± 3.0	57.2 ± 9.6
control 4	1.50 ± 0.1	0.89 ± 0.2	47.75 ± 4.5	31.97 ± 5.6	56.8 ± 11.3
papyrus 5	1.57 ± 0.1	0.81 ± 0.1	47.75 ± 4.5	29.75 ± 3.4	63.4 ± 8.6
papyrus 6	1.57 ± 0.1	0.98 ± 0.2	47.66 ± 4.5	33.69 ± 5.6	52.3 ± 11.8
Phragmites 7	1.57 ± 0.1	0.78 ± 0.2	47.69 ± 4.5	27.09 ± 5.2	72.3 ± 6.2
Phragmites 8	1.57 ± 0.1	0.83 ± 0.1	44.60 ± 4.1	28.08 ± 4.1	66.6 ± 6.5

The percent reduction in Tables 3.2 and 3.3 are calculated based on surface BOD influent and effluent loads.

Phase 3

(i) *Concentration trends*

BOD concentration profile along the series with small alternating open and planted zones, S 1-2 was exponential (Fig 3.11 a). The trend also shows the values derived from the first order kinetic model (k-C) given in equation 3.4. Average BOD concentration in the effluent over the time of the experiment was 14 mg/l. No specific concentration changes were observed in either the open or vegetated sections as probably was expected. The reduction trend correlated with the observed dissolved oxygen concentration changes along the same series, as reported earlier (Fig. 3.5 a).

The BOD transformations in the wetland series S 6-3-5 and S 8-4-7 illustrated in Fig. 3.11 b, correlated with the oxygen concentrations and pH induced factors (Fig. 3.4 b and Fig 3.5 b) in each of the individual units. The BOD concentration increase observed in the middle pond of the series (P 3 and P 4), characterised the secondary production from algal biomass. Most of the BOD removal occurred in the first pond of the series, the last pond acted as an independent system removing incoming BOD from the open ponds. Similar trends were recorded for total suspended solids in these two wetland serial combinations.

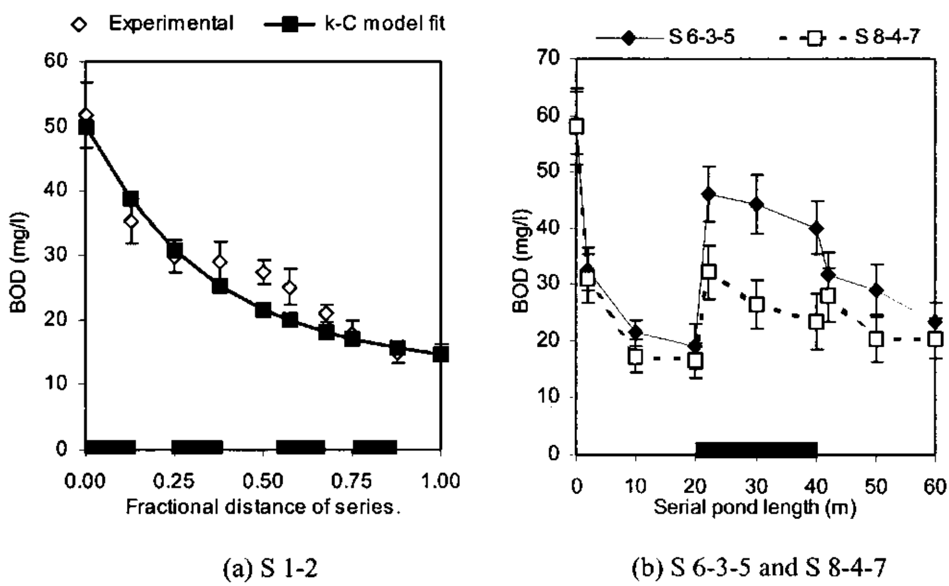


Fig. 3.11 a and b BOD longitudinal concentration profiles along the pond series S 1-2 (a) and pond series S 6-3-5 and S 8-4-7 (b) in Phase 3. The black sections show the zones without plants within the series.

(ii) *Mass loading effects.*

The influence of organic matter and suspended solids loading on their removal rates were similar to the observed relationship in phase 1, Fig. 3.8. In all wetland units combinations, linear relationships were derived for both BOD and TSS with the regression coefficient, $R^2 > 0.90$ (Fig. 3.12). The applied loading rates were however, lower than those used in the first phase; maximum BOD loading rates were 40 kg ha^{-1} day^{-1} and 70 kg ha^{-1} day^{-1} in series S 1-2 and S 6 -3-5, respectively. The linearity exhibited suggests that both the papyrus and *Phragmites* wetland units under this configurational setup, could be operated at elevated hydraulic loads and probably above the levels applied in phase 1. The applied mass loads at the Kirinya sewage lagoons over the time was in the range of 300 - 400 kg ha^{-1} day^{-1}.

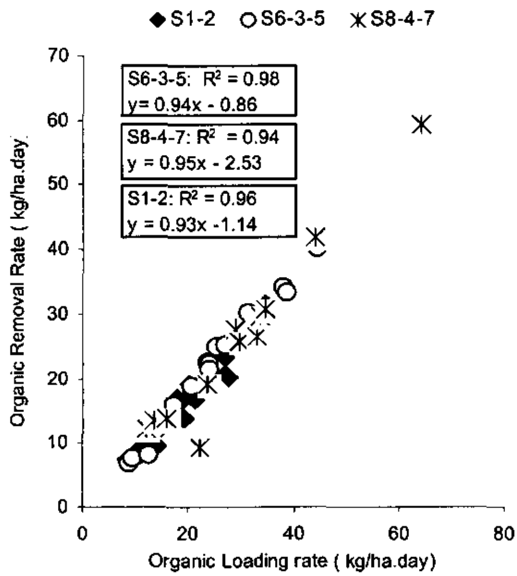


Fig. 3.12 The effect of BOD loading on BOD removal rates in papyrus series S 1-2 and S 6-3-5 and *Phragmites* series S 8-4-7 in Phase 3.

3.3.4 Residual BOD

In all the three operational phases, BOD concentrations measured in the effluent approached to a non-zero limit. During the whole experimental period, no effluent with zero BOD concentration was obtained in all the wetland units. This residual background concentration was calculated for papyrus and *Phragmites* systems using equations 3.3 and 3.4 and based on the experimental concentration data generated in phase 1 and in series S 1-2, phase 3. (Series S 6-3-5 and S 8-4-7 were not evaluated since no exponential relationship was sustained throughout the whole serial length).

A Globe calibration model developed by IHE Hydro Informatics Group was used for calibration and validation and, for determination of the residual BOD (non-zero background value) and the rate constant. Fig. 3.11 a shows typical BOD longitudinal concentration trends along the series, S 1-2 obtained from the experimental measurements and data predicted from the model.

A summary of the computed BOD non-zero background concentrations, C *, the BOD areal rate constants, k, and the net wetland BOD generation rates calculated by use of equation 3.4 are given in Table 3.5.

Table 3.5. Background BOD Concentrations, BOD areal rate constants and wetland BOD generation rate

System	Background BOD, C* (mg/l)	BOD areal rate constant (m/day)	Wetland BOD productivity rate (g m^{-2} day^{-1})
P 1	12.06	0.084	1.013
P 2	18.29	0.099	1.802
P 5	12.73	0.099	1.260
P 6	13.19	0.048	0.638
P 7	19.34	0.069	1.343
P 8	17.22	0.039	0.687
S 1-2	12.56	0.159	2.010

3.4 Discussion

BOD removal in constructed wetlands is generally considered to be more of a microbial mediated process specifically executed by attached aerobic and anaerobic bacteria (Wetzel, 1993; Cooper *et al.*, 1996; Wanner, 1997). In this study, dissolved oxygen concentrations showed a strong diel variability (0 - 27 mg/l) in open ponds but constantly low concentration (< 2 mg/l) prevailed in the fully vegetated units. Therefore, it seems that organic matter removal in the wetlands was influenced by both aerobic and anaerobic processes, but to varying extents in the different units.

(a) *Impacts of oxygen and pH*

Oxygen transfer into a fully developed and vegetated constructed wetland may be reduced to that trans-located by the macrophytes from the shoots to the rhizosphere and via surface diffusion or aeration when the shoot densities are very intense (Brix, 1990; Armstrong and Armstrong, 1988). This was apparently the case in the first phase of this study. The complete open sections created in the later phases were purposefully for augmenting oxygen supply into the wetland units. Oxygen in these zones was generated by photosynthesis performed by algae and cyanobacteria present in them (Muyima *et al.,* 1997) (Figs. 3.1 and 3.4 a). The high DO concentrations recorded in the large open ponds in this study is a typical characteristic of high-rate algal ponds, where oxygen residual concentrations of up to 33 mg/l have been reported (Picot *et al.,* 1993). High oxygen concentrations are linked to the high pH. The high pH values were related to shifts in the carbon dioxide, alkalinity and pH relationship as a result of algal consumption of inorganic carbon dioxide from the aqueous media (Goldman *et al.*, 1972; Sawyer *et al.*, 1994). Large pH shifts were typical of the large open ponds (Figs. 3.2 and 3.4 b), whereas in the wetland units with low algal metabolism, pH and alkalinity fluctuations were minimal.

The oxygen trends obtained in phase 3, series S 1-2 (Fig. 3.5 a) depicts a typical oxygen sag curve (characterised by depletion and recovery), which is a feature in rivers / streams loaded with high amounts of organic matter and nitrogen compounds (Chapman, 1997).The trend suggests that the wetland configuration adopted (small alternating open areas and planted areas) may have been in the range of the existing total oxygen demand of the system. The sustained BOD and ammonium N reduction obtained in this series (Fig. 3.11 a and Fig. 4.15, Chapter 4) supports these deductions. In these series, the surface area of the open zones tended to limit excessive algal growth as a result of the intermediate residual oxygen concentrations (maximum 12 mg/l) and neutral pH that prevailed. The pH and DO concentrations were within the suitable range for microbial degradation of both BOD and nitrogen. Meanwhile, in the other wetland series, S 6-3-5 and S 8-4-7, the large open ponds greatly influenced the oxygen concentration and subsequent BOD reduction (Fig.3.11 b). The BOD concentration increased in these open ponds despite the increased residual oxygen (Fig 3.5 b). Two explanations may be adduced to the observation. Firstly, the pH of the open ponds (> 9.5) during the day tends to inhibit the activity of heterotrophic bacteria, degraders of organic matter (Henze *et al.*, 1997) and thereby, causing a build-up of BOD in the systems. Secondly, and probably the most critical issue, was the proliferation of algae in these ponds (Fig. 3.3), which was responsible for generation of large amounts of secondary BOD from decaying algal biomass. The latter process was also the cause of the observed increase in suspended solids concentrations measured in the effluents from the open ponds in Phase 1 and Phase 3.

(b) *Organic and suspended matter removal*

The reduction of both BOD and TSS was consistently exhibited by *C. papyrus* and *Phragmites* wetland units in all phases (Fig 3.6, 3.10, 3.11). The effluent values were lower as compared to that derived from the existing Kirinya sewage lagoons (Table 3.1). Fig. 3.11 showed more reduction nearer the inlet. These observations are consistent with field observations obtained by Tchobanoglous (1987) in San Diego wetland and Gearheart *et al* (1989) in the study of the City of Arcata wetlands. Data from a batch fed wetland in Richmond in Australia indicated similar characteristics (Kadlec and Knight, 1996).The quiescent conditions prevailing in the wetlands are considered to be responsible for creating the ideal conditions for the observed rapid removal of these pollutants. The increased reduction of BOD concentration in partially planted pond as compared to fully planted pond, showed that the small open zones augmented aerobic BOD degradation. This was the desired effect for introduction of the pond configurational modifications during phase 2.

The direct correlation of organic loading on its removal rate has been suggested as one way for determining the maximum organic carrying load that a wetland system can bear (Hayes *et al.*, 1987). In this phase 1 of this study, a linear relationship of loading vs removal rates was sustained at maximum applied COD and TSS loading of 350 kg ha^{-1} day^{-1} and 150 kg ha^{-1} day^{-1}, respectively (Fig.3.8) and maximum BOD loading of 100 kg ha^{-1} day^{-1} . The US Environmental Protection Agency (EPA, 1988) recommends a maximum of BOD loading rate of 112 kg ha^{-1} day^{-1}, Knight (1993) recommend rates between 50 - 80 kg ha^{-1} day^{-1} as optimal, while WPCF (1990) recommends a maximum of 100 kg ha^{-1} day^{-1} for systems operated both as batch or continuous flow type. The results from this study applied to a wider linear range of organic loading and removal rates which suggests that the systems could still be operated beyond the maximum ranges applied since the temperatures prevailing favour relative high rates of BOD degradation. This is justified since at hydraulic loading rates between 5 - 10 cm/day, BOD removal efficiency was still above 60% (Fig. 3.9 a) and yet effluent concentrations were still below the design value of 30 mg/l.

The influence of the different phases on BOD and TSS removal efficiency was not evident except in the first half of the 2nd phase (Table. 3.2). In phase 1, BOD removal (on mass basis) in *Phragmites* systems ranged between 60 to 98% with a systems average of 82%. This was significantly different (p = 0.01) from that of the control open pond 4 (43%). Likewise, in papyrus systems, the removal ranged between 50 to 86% with an overall mean of 73%, which was also significantly higher (p = 0.01) than in the corresponding control open pond 3 (33%). Meanwhile, TSS removal in the same phase averaged to 80 % in both papyrus and *Phragmites* systems as compared to the negative removal of - 44 % in the control open ponds. Results from phase 3 registered BOD and TSS removal > 80% from all the three serial pond combinations. All the results of these three phases were on the upper limit of the wider bracket range (40 - 90 %) which is reported by several authors from different sized systems in temperate environments (Bastian and Benforado, 1983; Reed *et al.*, 1988; Gersberg *et al.*, 1989; Rijs and Veenstra, 1990; Bastian and Hammer, 1993; Steiner and Combs, 1993). Therefore a conclusion drawn from the results of the Kirinya wetland and the comparisons is that tropical wetlands using the two plants (*C. papyrus* and *Phragmites)* are suitable for application in the removal of organic matter and suspended solids and even at higher hydraulic loadings than in the temperate wetlands.

(c) *Wetland residuals*

The residual (non - zero) BOD concentration, C*, determined for the Kirinya wetlands (Table 3.4) were consistently greater than the values (average C* = 6.2 mg/l) reported by Kadlec and Knight (1996), for the North American treatment wetlands that have been in use for many years. The areal rate constants obtained (0.039 - 0.099 m/day) were within the ranges cited for same North American systems (average, k = 0.093 m/day) as well as for the United Kingdom systems (0.067 - 0.1 m/day) (Cooper *et al.*, 1996) and the Danish systems (0.083 ± 0.017 m/day) (Schierup *et al.*, 1990).

The BOD removal rates in this study from individual planted units were not similar and tended to be dependent on the hydraulic loading. The reason for this may be related to the variability in water depths and void fractions in each of the wetlands and the inherent short-comings of the plug-flow assumptions imbedded in the equation used in the calculations. The tropical environmental conditions at Kirinya were suitable for rapid decomposition and regeneration and therefore, creating a ready source of soluble organic matter within the systems as depicted by the high non-zero BOD concentration. The relatively high and steady temperatures throughout the year in particular, may be the most critical factor responsible for the accelerated decomposition rates in the Kirinya setting. Reports by Brix (1998) suggesting that reduced background BOD and rather increased reaction rates are derived as systems age, were not obvious from the results of this study as shown in Table 3.5. Total suspended solids exhibited similar characteristics of possessing an irreducible background concentration. This, although not quantified in this study, was generated from the several dynamic biological process that take place in wetlands which include death, litter fall and algal production. Based on the result from this study it may be concluded that background BOD in treatment wetlands in the tropical environments will be higher than in temperate environments. This will tend to limit the lowest effluent levels attainable and has to be considered in the wetland design phase.

3.5 Conclusions

The overall conclusion drawn from the results of this study is that the *C. papyrus* and *Phragmites mauritianus* constructed wetland systems have a potential for effective removal of organic matter and suspended solids from wastewater. The effluent concentrations of COD, BOD and TSS from these wetland units were consistently below the prescribed wastewater discharge regulations throughout the experimental period. Both plant types are therefore good candidates for larger scale application in wetland treatment in similar environments although papyrus wetland units performance was comparatively higher than that of the *Phragmites.*

Aerobic and anaerobic degradation pathways were both significant in the organic matter removal in the wetland units as illustrated by the extent of BOD removal in the fully vegetated units with low oxygen, and in the serial units with increased oxygen. Combined vegetated and non-vegetated (open) wetland units (but with the last unit vegetated) provides a suitable mix for an increased reduction of these pollutants than in isolated units. The pH, temperature and dissolved oxygen in the open zones and consequently, the amount of organic and suspended matter that can be removed was, dictated by the size of this open zone. Small sized open zones are preferred because of the intermediate amounts of oxygen and pH generated and sustained. These amounts minimised production of high concentrations of secondary BOD and TSS unlike in the case when these areas were large. Hydraulic loading rates ranging between 4 - 6 cm/day and retention times between 4 - 6 days were found to be optimal for the removal of organic and suspended matter from the wastewater. Finally, the results from this study have shown that complete removal of BOD to zero concentration in the effluent may not be achievable due to generation of BOD by the systems, the threshold value in the study was 12 mg/l.

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Chapter 4

Removal of Nutrients

Abstract 73

4.1 Introduction 73
- 4.1.1 Nitrogen 74
- 4.1.2 Phosphorus 76
- 4.1.3 Scope of the study 76

4.2 Materials and Methods 77
- 4.2.1 Screening 77
- 4.2.2 Substratum phosphorus retention potential 77
- 4.2.3 Plant Biomass: yields and nutrient storage capacity 78
- 4.2.4 Organic Carbon Accumulation 79
- 4.2.5 Quantitative Analysis 79
- 4.2.6 Statistical analysis 80

4.3 Results 80
- 4.3.1 General characteristics 80
- 4.3.2 Substratum phosphorus retention capacity 80
- 4.3.3 Biomass Yields 80
- 4.3.4 Nutrient distribution 84
- 4.3.5 Organic matter deposition 86
- 4.3.6 Nutrient Transformations 86

4.4 Discussion 94

4.5 Conclusions 102

References 103

Abstract

The use of constructed wetlands in the removal of nutrients from wastewater has been studied extensively using different types of wetland systems in the colder regions but only to a lesser extent, in the warmer regions. The objective of this study was to investigate this potential function of wetlands but under the warmer tropical environments and to quantify the contribution of plant uptake to the overall nutrient removal. The study was carried out using horizontal surface flow constructed wetlands in three different wetland configurations.

Removal of ammonium nitrogen (NH_4-N) in the vegetated and non-vegetated wetland units was influenced by the shoot density and surface areas of the open zones respectively. Ammonium reduction measured in the papyrus wetland units in phase 1 decreased from over 80% to 25 %, as the plant density increased from 20 shoots m^{-2} to 80 shoots m^{-2}. An improvement in the removal of ammonium was registered in the second phase but was not significant. In the third phase, over 90% of ammonium was removed in the two serial configuration applied and, in all the three sets, effluent concentration averaged less than 5 mg/l. Given the prevailing pH and temperature in the series with alternating open and planted zones of small surface areas, nitrification - denitrification pathway was considered to be a major pathway with estimated contribution up to 77% of ammonium removed. In the other series with larger surface areas of open zones, the volatilisation pathway is suggested to be the main pathway considering the high pH and temperature that prevailed. Phosphorus removal was higher in units with a substratum base than those without it. Mean removal efficiency of 45% was registered in the *Phragmites* units as compared to 34 % in the floating papyrus units.

Maximum standing biomass productivity of up to 108 ton ha^{-1} and 104 ton ha^{-1} were obtained during the exponential growth stage of papyrus and *Phragmites* plants, respectively. The rooting biomass yield for the respective plants was 66 ton ha^{-1} and 4 ton ha^{-1}. Nutrient uptake rates in the same period were: 7.1 kg N ha^{-1} day^{-1} and 0.24 kg P ha^{-1} day^{-1} in papyrus and 10.4 kg N ha^{-1} day^{-1} and 0.26 kg P ha^{-1} day^{-1} in *Phragmites*. From the mass balance computations, it was deduced that nutrient uptake into the standing biomass at the exponential growth stage contributed up to 15 % N and 10% P of total nutrient input into each of the papyrus wetland units and 58 N % and 37 %P in *Phragmites* units. The plant uptake however, declined in the next phases as the plants matured, to less than 4 % of the total input of both nutrients.

4.1 Introduction

Nitrogen and phosphorus are the macro-nutrients usually found in domestic and industrial wastewater. Nitrogen is found predominantly as ammonium and organic nitrogen (mainly proteins and amino acids, which hydrolyze to yield ammonia), while phosphorus is found in both soluble and insoluble forms. Effluents containing these constituents when discharged untreated into water bodies may cause a variety of problems which among others include the following:
(i) Depletion of dissolved oxygen arising from the nitrification process which could result in disturbance of the oxygen demands of other aquatic organisms; the effect may be fatal in some instances.
(ii) Toxic effects of ammonia (NH_3) on higher aquatic life like fish which is often the case where discharged wastewater has a high pH.
(iii) Stimulation of growth of aquatic life like algae and water hyacinths causing increase in water turbidity and associated problems.
All the above effects have both environmental and economic implications to the users and other

beneficiaries of the water body receiving nutrient loaded wastewater. It is therefore essential that nutrient levels in the effluents which are discharged into natural ecosystems are controlled so as not to compromise their trophics status.

4.1.1 Nitrogen

Nitrogen is an essential nutrient for biological growth, and it is available in different forms for different purposes. The major pathways in which nitrogen becomes available to the different array of organisms is represented by the nitrogen cycle detailed in Barnes and Bliss (1983) and Reddy and Patrick (1984). The transformations that have been directly related to the removal of nitrogen compounds in wastewater by treatment wetlands include the sequential nitrification-denitrification process, assimilation by periphyton and phytoplankton and volatilisation of ammonia gas. The contribution of each of these processes is variable and is dependent on the factors that control them.

Sequential nitrification-denitrification is considered to be the main route for nitrogen removal (export) during wastewater treatment by any process including constructed wetlands (Bowmer, 1986; Patrick, 1990). The initial transformation is nitrification which is microbially mediated in two distinct steps: ammonium is oxidized sequentially to nitrite and nitrate by organisms of the genera *Nitrosomonas* and *Nitrobacter* (Kuenen and Robertson, 1988). These steps are represented by the equations:

$$NH_4^+ + 1.5\ O_2 \rightarrow NO_2^- + H_2O + 2H^+ + (240 - 350\ KJ) \qquad (4.1)$$

$$NO_2^- + 0.5\ O_2 \rightarrow NO_3^- + (65 - 90\ KJ) \qquad (4.2)$$

The rate of oxidation of nitrite is faster than that of ammonium and consequently, nitrite does not accumulate in the systems. In this study, nitrite concentrations in the wastewater taken from within the wetland units and the effluent was negligible.

The nitrification process is usually accompanied by a decrease in the pH. The overall equation illustrating this effect including the nitrifier synthesis is given as (Barnes and Bliss, 1983 and Matsumura *et al.*, 1997):

$$NH_4^+ + 1.5\ O_2 + 1.98\ HCO_3^- \rightarrow 0.021\ C_5H_7NO_2 + 1.041\ H_2O + 0.98\ NO_3^- + 1.88\ H_2CO_3 \qquad (4.3)$$

From these equations, it may be noted that large amounts of oxygen (4.2 g of oxygen) are required for oxidation of 1 g of NH_4^+-N. In treatment wetlands, the availability of sufficient amounts of oxygen is often the limiting factor for ammonium removal.

The nitrate produced as described above gets reduced to elementary nitrogen gas through the process of denitrification. This process in contrast, mainly occurs under anaerobic conditions. It is through this process that actual export of N from wastewater treatment systems takes place. An external and readily biodegradable carbon substrate is essential for the transformation, although the process can also take place at reduced rates by slow endogenous respiration (Stensel and Barnard, 1992, Wanner, 1997) when the substrate is exhausted. In treatment wetlands the carbon source is not limiting as the required carbon is readily derived from decaying organic plant material. It is nitrate which is usually the limiting factor.

In this study, nitrification-denitrification processes were maximized in one operational mode through the creation of small alternating open and planted zones. The controlling environmental factors namely pH, dissolved oxygen concentration and temperature in the wetland units were in the ranges that stimulate nitrate production.

A simplified equation using methanol as the organic source for denitrification (Barnes and Bliss, 1983) is given as:

$$NO_3^- + 0.83\ CH_3OH \rightarrow 0.5\ N_2 + 0.83\ CO_2 + 1.17\ H_2O + OH^- \qquad (4.4)$$

The equation indicates that this process is always accompanied by an increase in alkalinity which counteracts the acidity production during the nitrification process. The pH in treatment wetlands is well buffered as a result of this counteractive effect.

The second route identified for removal of nitrogen compounds from wastewater is via assimilation of ammonium or nitrate by bacteria during growth and by periphyton production. Plant uptake is also suggested as one possible route for nitrogen removal. However, contrasting views on its significance on the overall treatment process have been expressed in literature. Gersberg *et al* (1986), Haberl and Perfler (1991) and Brix (1994), urge that plant uptake plays an insignificant contribution in the overall treatment by wetland systems. Geller (1997), using data derived from a 4 year mass balance study of a *Phragmites* treatment wetland, obtained N and P uptake values of only 4 and 2 %, respectively. He concludes that harvesting the reeds does not make sense and can only lead to destruction of plants. Howard-William (1985) and Brix (1997) further suggest that the nutrient uptake route provides only a temporary storage, which is subject to saturation and eventual release of stored nutrients when the plants die.

In the studies by Breen (1990) on a laboratory scale with cattail (*Typha* spp), plant uptake accounted for up to 50 % of N input. Rogers *et al* (1991), reported a higher level of 90 % N removal via uptake in their study with *Scirpus validus* plants. Koottatep and Polprasert (1997), also found that N uptake accounted for 43 - 75 % of total nitrogen (TN) input to the laboratory and pilot scale constructed wetland units located in a tropical environment. The significance of the uptake route is further reported in separate studies conducted in natural wetlands in Kenya (Muthuri *et al.*, 1989), in Australia (Greenway, 1997) and in the Netherlands (Meuleman, 1999). The nutrient uptake capacity of the native plants used was found in all cases to be high in environments receiving wastewater as compared to their natural location. The extent of nutrient uptake seems to be controlled by the climate and plant type. Generally, in climates where plant growth is conducive for the whole year, nutrient storage may be limited, while in areas which are subject to climatic fluctuations, storage of nutrients especially in the rooting biomass is a survival strategy.

Nitrogen export via volatilisation is a physico - chemical phenomenon which is a function of pH, temperature and ammonium concentration. It only becomes significant when pH > 9.3 (Emerson *et al.*, 1975, Reddy and Patrick, 1984). Vymazal *et al* (1998) suggest wind velocity, solar radiation and nature of the aquatic plants as other factors that can determine the rate of volatilisation in a constructed wetland. Most wetlands have a well buffered pH usually < 8, and this export route is therefore generally insignificant. However, in cases where photosynthesis by algae or free floating and submerged macrophytes becomes prevalent (unshaded zones), pH values can rise above 10. The pH increase is due to the consumption of inorganic carbon dioxide from the aqueous media by algae resulting in a shift in the bicarbonate equilibria in the water. During this study, volatilisation was identified as a major export route in the control (open) ponds.

4.1.2 Phosphorus

Phosphorus is commonly present in wastewater in soluble and insoluble forms. The soluble forms are orthophosphates which may exist as three different ionic species viz: $H_2PO_4^-$, HPO_4^{2-} and PO_4^{3-}, depending on the pH. The interaction between the soluble and insoluble forms and the controlling parameters in wetlands are detailed by Kadlec and Knight (1996). Stumm and Morgan (1996), Faulkner and Richardson (1989), Lijklema (1990), Mitsch and Gosselink (1993), Reddy and D'Angelo (1997), have all outlined the processes and factors that control phosphorus retention in wetlands together with limitations of phosphorus removal in the treatment wetlands. The key controlling factors identified involve interaction with the redox potential, pH, Fe, Al and Ca minerals in the soils or substrata.

In acid soils, inorganic phosphorus is adsorbed on hydrous oxides of Fe and Al. It can also be precipitated as insoluble Fe-phosphate and Al-phosphate but at pH > 7, it is mainly precipitated as Ca-phosphate. Steiner and Freeman (1989), Mann (1990), Wood(1990), Richardson and Craft (1993) suggest adsorption to be the major route for both short and long term removal of phosphorus from wastewater. However, sites for adsorption and precipitation are finite and are susceptible to saturation. When that stage is reached no further removal of loaded phosphorus can be realized.

As described for nitrogen, assimilation by phytoplankton and bacteria, and by plants play a role in phosphorous removal from wastewater. Algal phosphorus uptake contributes to the removal in two ways: assimilation of dissolved P to satisfy cell synthesis and by precipitation of insoluble phosphate (Muyima *et al.*, 1977). However, all the P removal via these routes offer only a short term storage of P (Richard and Craft, 1993) as they have fast recycle rates in which stored P gets returned to the water column. In this study, phosphorus assimilation by bacteria and phytoplankton seems to have been considerable in control (open) ponds.

4.1.3 Scope of the study

Previous studies cited in the literature have identified the potential of constructed wetlands in removing nutrients. However, since most of the results and conclusions are based on data generated from temperate regions, extrapolating them to conditions in the tropics must be done with caution. During this study, focus was centered on finding out the capacity of tropical constructed wetlands using indigenous plants, in stripping nutrients.The key objectives of this study therefore were:

(i) to determine the potential of nutrient removal by a tropical constructed wetland under different operating conditions and system configurations (as described in chapter 2).
(ii) to determine the nutrient uptake capacity and storage in the wetland plants and conditions influencing it.

Several activities were carried out to achieve the stated objectives during the different operational phases although some experiments were limited to the first phase only. Summary key activities for all the phases involved:

1. measuring vertical and horizontal profiles of physico-chemical and environmental factors: pH, oxygen, temperature and conductivity.
2. measuring the concentrations of N and P (different forms) for samples taken from the selected sites (chapter 2).
3. measuring N & P content in plant tissues
4. measuring the potential of the substratum used to remove P.
5. measuring rooting biomass volumetric changes.

4.2 Materials and Methods

4.2.1 Screening

Nitrogen and phosphorus concentrations and general forms in the raw sewage received at the Kirinya sewage treatment works was obtained from the 4 years' database of National Water and Sewerage Corporation (NWSC, 1998), Jinja Area. In addition, concentration measurements were carried out as described below during storm events in order to assess the impact of these events on the influent concentrations pumped into the pilot constructed wetland.

4.2.2 Substratum phosphorus retention potential

Two laboratory experimental set-ups were used to estimate the phosphorus removal potential of the gravel used as substratum in the constructed wetland in ponds 4, 7 and 8.

4.2.2.1 Retention - release potential

Samples of gravel originally obtained from the nearby Kirinya Prison quarry had a variable size range that was graded into three diameter sizes: 5 - 10 mm, 15 - 25 mm and 30 - 50 mm The gravel (1000 g) was packed in separate glass columns. A potassium hydrogen phosphate solution (10 mg/l P in 0.01 M KCl) was added into each of the columns at flow rates of 20 ml/ hr (an estimate of the flow rates that were applied at wetland units). 20 ml aliquots of the eluate were collected at one hour intervals and the concentration of total reactive phosphorous measured as described in 4.2.5 (ii). This was repeated until the leachate concentration equaled 10 mg/l indicating complete saturation of phosphate removal routes. The columns were then flushed with a 0.01M KCl until the concentration in the eluate remained constant. This step was used in order to determine the percentage of phosphate strongly bound to the substratum. The underlying assumptions for equilibrium conditions were that adsorption to the walls of the columns were negligible as well as the change in the volumes of overlying water during the successive sample removal.

4.2.2.2 Precipitation potential

The chemical composition of the gravel with respect to aluminum, iron, and calcium was measured by an extractable ions leaching procedure (Mann, 1990). 20 g of gravel samples was placed in three erlenmeyer flasks to which either 200 ml of distilled water, 0.01 M KCl or 0.5 M HCl was added.The flasks were shaken on an orbital shaker (800 rpm) for 16 hrs at room temperature. Thereafter, the concentration of Fe^{2+}, Al^{3+}, and Ca^{2+} in each of the filtrates was determined using an Atomic Absorption Spectrophotometer at Makerere University, Chemistry Department. A total acid (HCl) digestion experiment was also carried out to determine the amounts of iron oxide and aluminium in the substrata.

4.2.3 Plant Biomass: yields and nutrient storage capacity

The plant growth rates and nutrient storage were measured continuously using defined quadrats, each 1 m^2, that were established near the inlet, the mid-positions and near the outlet of each planted wetland unit.

4.2.3.1 Plant growth: - Rooting biomass

(a) Volumetric method

The experiment was carried out starting from Oct. 96 to Oct. 97 and only in the papyrus ponds. Initial attempts to extend the study to the *Phragmites* ponds were not successful because of substratum complications. The experiment essentially involved draining each of the units completely, followed by addition of 2 m^3 of wastewater at intervals, each time recording the corresponding water level at the mid-pond point until water just covered the rhizome/mat complex. The volume of wastewater added up to that level was recorded and was referred to as Effective Volume, V_{eff} .Using the same water level, the volume of the pond was computed as if it were empty (the scenario when there are no plants in the ponds); this was referred to as Ideal Volume, $V_{i.}$.

The volume of the rooting biomass; V_b, was taken as equal to the difference between the ideal and effective volumes: $V_b = V_i - V_{eff}$

(b) *Dry weight method*

In the second approach, rooting biomass was measured by an indirect method using the wet - dry weight ratios. Wet weight determination by the direct method involving complete harvest or removal of plant biomass in a given segment was avoided as it would have resulted into destruction of the plant root/rhizome mat complex at each time of measurement. Instead, small but representative wet biomass samples of rhizome/roots were harvested on each sampling schedule from each of the established quadrats. Known weights ($\approx$ 200 g) of the wet samples were immersed in water (500 ml) in a measuring cylinder, the amount of water displaced gave the corresponding volumes of the wet samples. The density of the sample was computed and extrapolated to determine the total wet biomass for the whole pond using the rooting biomass volume, V_b, measured in (a) above. Dry weights of same samples were obtained after oven drying at 80 °C and 105 ° C. Ratio's of dry weight / wet weight were computed accordingly. Total dry weight biomass for the whole pond was obtained by multiplying the total wet biomass with this ratio.

4.2.3.2 Plant growth:- Standing biomass.

(a) *Shoot density*

On each experimental date the number of shoots in each quadrant was counted. The initial shoot count was carried out after equilibration of the systems following transplanting of propagules. Counts were cross-checked by repeated counting. The numbers obtained from each transect were pooled together and extrapolated to estimate the total plant shoot density per unit area and for the whole pond.

(b) *Dry weight*
Plant samples were collected during each experimental day. Two plant shoots - one young and one mature - were randomly selected and harvested from each quadrat. Their fresh weights were measured immediately at Kirinya before taking them to the laboratory. The umbels and culms were separated before each set was chopped into smaller parts, then sun dried for 1 week prior to oven drying at 80 °C for at least 24 hours. Their dry weights were determined after cooling. The dry/wet weight ratios were computed for each set of shoots and these were used to extrapolate equivalent shoot dry weights per unit area (g DW/m^{-2}).

4.2.3.3 Nutrient storage

The content of nutrients in the plant biomass was computed as a product of dry weight biomass (rooting or standing) and the concentration of nitrogen or phosphorus in plant tissues. The determination of the latter is described in section 4.2.5.

4.2.4 Organic Carbon Accumulation

This experiment was restricted to *Phragmites* systems and control pond 4 only. Substratum samples were removed randomly at defined time intervals from these ponds at three positions. They were sun dried together with the control samples consisting of unused gravel to near constant weight. Samples were weighed before firing in a furnace at 550°C (Chemistry Department, Makerere University). The loss on ignition was determined accordingly and it represented the amount of the accumulated organic matter.

4.2.5 Quantitative Analysis

Wastewater and plant samples at the laboratory were pre-treated and handled as described in standard methods (APHA, 1992). A HACH hand book for DR 2010 and a CECIL CE 1000 spectrophotometers were referred to when carrying out specific measurements using these instruments.

4.2.5.1 Nitrogen Compounds.

Ammonium-N was measured within six hours of sampling following a Nesslerization method with a DR 2000 spectrophotometer at 425 nm wavelength. Nitrate (NO_3^-) was determined using the Cadmium reduction method and spectrophotometric measurement using a CECIL CE 1000 Series instrument at 543 nm. The glass columns used were fabricated to the required specifications at the glass blowing laboratory of Kyambogo Curriculum Development Centre. Total nitrogen in plant digests was determined as ammonium after sample digestion with selenium, sulphuric and salicylic acids and hydrogen peroxide mixture on an aluminium digestion block using the procedure of Novozamsky *et al* (1983).

4.2.5.2 Phosphorus compounds

Ortho-phosphate (o-PO4) was determined using the Ascorbic Acid method and a CECIL CE 1000 for spectrophotometric measurements at 880 nm. Total phosphate (TP) in water samples was determined via persulphate digestion in an autoclave and the ascorbic acid method. Total phosphate in plant samples was determined after digestion described in (i) using the ascorbic acid method, after making necessary dilutions.

4.2.6 Statistical analyses

Data analyses were carried out using MINITAB Release 10 for Windows software. Comparison of variables was performed using analysis of variance technique and Tukeys and Dunnets multiple comparisons to test differences in the means (ANOVA). The family error rate was set at 5 % and occasionally at 1 %, the individual error rate was automatically set by the programme in response to the number of comparisons.

All treatment performance evaluation was based on mass budgets rather than concentration unless otherwise stated.

4.3 Results

4.3.1 General characteristics

The concentration of nitrogen and phosphorus compounds in the wastewater received at Kirinya station and the level of treatment by the existing stabilization ponds at different stages is given in Table 3.1 (Chapter 3). The data show that the level of nutrient removal is relatively low under the present treatment regime.

4.3.2 Substratum phosphorus retention capacity

The phosphorus retention potential (as o-PO_4) of the three gravel sizes 5 - 10 mm, 30- 50 mm and 15 - 25 mm were on average 19%, 35% and 28%, respectively. The smaller sized gravel did not retain more phosphorus as expected given their large surface area, probably due to persistent algal interference which was more visible in the column packed with this sized gravel. Besides, the gravel in all the columns tended to break into fine particles but the disintegration was more in the column packed with the small sized gravel. The fine particles that leached out during experimentation may have contained adsorbed phosphorus which could not be measured and consequently reduced the contribution of P removal by the different gravel sizes. At the Kirinya pilot wetland units, this tendency of disintegration of the substratum was only observed at the beginning of the wetland operation but it ceased with accumulation of organic matter and as plants established a root net work that bound the substrata together.

The gravel chemical composition was determined as: Silica, 52%, Iron(III) Oxide, 32%, alumina, 9% and CaO, 5%.

4.3.3 Biomass Yields

4.3.3.1 Rooting biomass.

The papyrus rooting biomass estimated by volumetric methods in each of the ponds fluctuated. Overall, there was a progressive volumetric increase of up to 30 % of the effective pond volumes (Fig 4.1) in an 18 month period and up to 40% after 2.5 years (end of Phase 3). The volumetric fluctuations were a result of growth changes, drying of parts of rooting biomass exposed by wind overturns, and continuous rotting / decay and accumulation of litter at the pond bottom. In October 1997, while starting phase 2, ground biomass of *Phragmites* units was measured and found to contribute only 10 % to the total biomass.

The rooting biomass (dry weight per unit area) yield increase was rapid during the exponential growth stage with average growth rates of 23 g m^{-2} day^{-1}. However, the growth rate declined to only 2.5 g m^{-2} day $^{-1}$ after the first 8 months. The overall biomass yield averaged to 6.6 kg m^{-2} over the 18 months' experimental period.

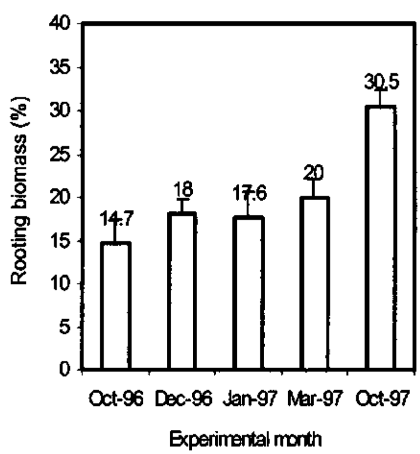


Fig. 4.1 a Pond volumes (%) occupied by papyrus rooting biomass from Oct 96 - Oct 97.

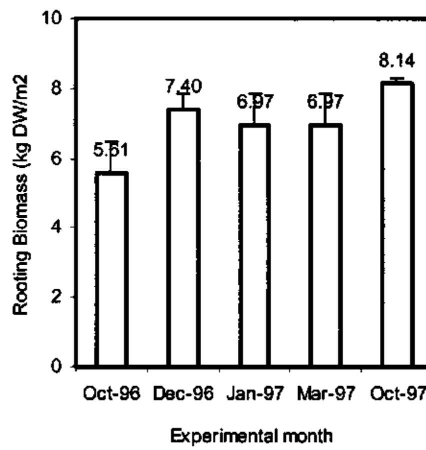


Fig. 4.1. b Papyrus rooting biomass (dry weight/m^2) changes from Oct 96 to Oct 97.

4.3.3.2 Standing biomass.

(a) *Shoot density*

The shoot density in papyrus units rapidly increased from 17 shoots m^{-2} in May 1996 to a maximum of 117 shoots m^{-2} within ten months and reached shoot heights of 3 m (Plate 4. 1). This was followed by a gradual decline to an average of 80 shoots m^{-2} by the 18^{th} month (Fig. 4.2 a). In *Phragmites* units, the lag-phase (Aug- Nov 96) was rather long probably due to the attack by aphids in the second month after planting. The exponential growth stage is shown by an increase in shoot density from 82 to 282 shoots m^{-2} within four months with shoot heights of 4 m (plate 4.2). The shoot density declined to an average of 140 shoots/m^2 by the 11^{th} month (Fig. 4.2 b). The decrease in shoot density was due to senescence of shoots and the occasional physical destruction by vermin monkeys and wind.

Plate 4.1 Relative shoot height of *C. papyrus* 16 months after planting

Plate 4.2 Relative shoot height of *P. mauritianus* 13 months after planting

The shoot density fluctuations were not correlated to seasonal variations as reported by Meuleman (1999) in *Phragmites* plants in temperate climates.

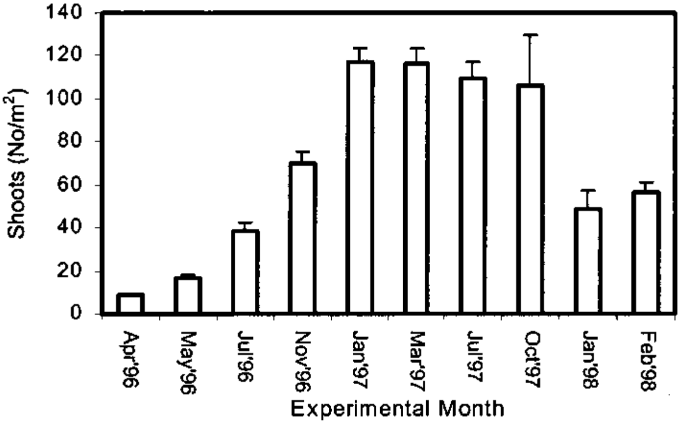


Fig. 4.2. a The variation of papyrus shoot density over a 22 month period; vertical bars indicate standard error of the mean (n = 12) from the four ponds.

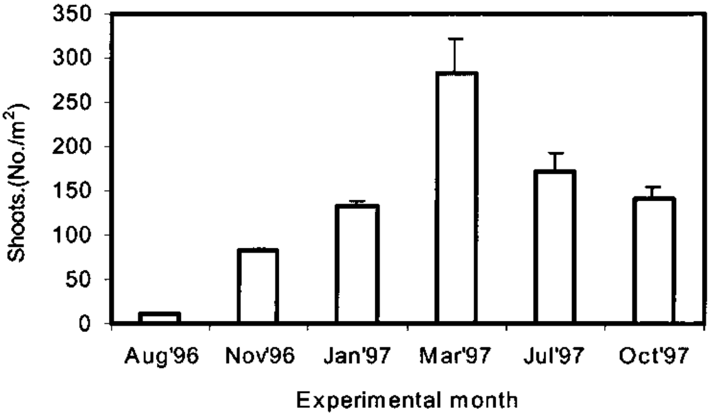


Fig. 4.2. b The variation of *Phragmites* shoot density over a 11 month period; vertical bars indicate standard error of the mean (n = 6) from the two ponds.

(b) *Dry weight*

Standing biomass production as determined by the dry weight method was directly correlated with the shoot density trend. Maximum biomass yields of 10.82 kg m^{-2} and 10.43 kg m^{-2} were recorded in papyrus and *Phragmites* wetland units respectively (Fig 4.2 c). However, over the whole study period, the biomass production yields in the same systems averaged to 5.76 kg m^{-2} for papyrus and 3.44 kg m^{-2} for *Phragmites*. Standing biomass in papyrus systems accounted for 45.7% of the total biomass, while it represented 90% in *Phragmites*. No statistical significance ($p > 0.05$) was found between the biomass production yields in the individual wetland units over the 22 months measurement period. It may therefore be concluded that the growth conditions in the ponds were similar.

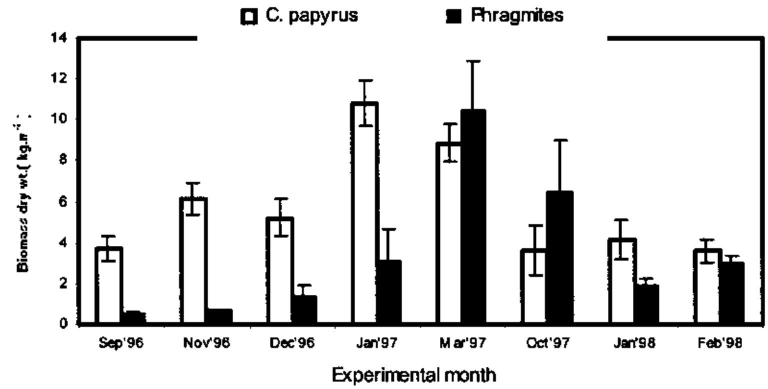


Fig. 4.2. c Standing biomass yields of papyrus and *Phragmites* for the period Oct 96 - Feb 98, vertical bars show standard error of the mean (n = 12 for papyrus and n = 6 for *Phragmites*)

4.3.4 Nutrient Distribution

4.3.4.1 Plant organs

The nutrient allocation in the different plant organs was variable. In papyrus plants, the highest nutrient (both N and P) concentration was in the roots and least in the culm. In contrast, the roots of *Phragmites* had rather low values when compared with the stems / leaves (Table 4.1). Results obtained did not show defined seasonal variation in nutrient content although higher concentrations were recorded when all the plants were young (Fig. 4.3 a).

Table 4.1: Nutrient distribution in different plant parts (dry weight, DW ± SE, n = 32).

Plant	Plant part	TN (mg g^{-1})	TP (mg g^{-1})
papyrus	rhizome	21.19 ± 2.2	5.89 ± 0.6
	roots	27.18 ± 1.2	9.51 ± 0.7
	umbel	22.40 ± 3.4	7.00 ± 1.6
	culm	11.60 ± 1.7	3.54 ± 0.8
Phragmites	stem and leaves	14.34 ± 2.0	3.91 ± 1.4
	roots	13.97 ± 0.9	2.60 ± 0.2

4.3.4.2 Shoot biomass

The nutrient content of the standing biomass per unit area was computed by multiplying the unit area biomass yields with the corresponding nutrient concentration. The nutrient content obtained was more or less a function of plant biomass rather than the nutrient concentration when the latter became steady after 6 months. The nitrogen yields in papyrus biomass showed no significant difference with that of *Phragmites* ($p = 0.34$) with overall averages

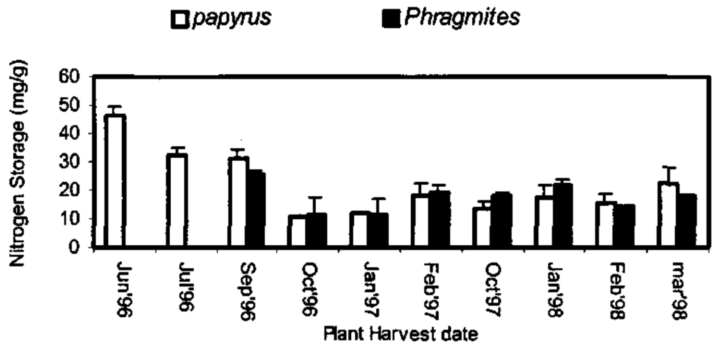


Fig. 4.3. a Nitrogen content (mg/g) in shoot biomass of papyrus and *Phragmites* from June 1996 to March 1998, vertical bars show standard error of mean (n=8).

of 89.12 g N m^{-2} and 63.7 g N m^{-2}, respectively (Fig. 4.3 b). Both nutrient concentration and biomass influenced the unit area phosphorous yields especially in October 97 and November 97, during the heavy rains (Fig.4.4). The mean P content in papyrus and *Phragmites* biomass were 19.1 g P m^{-2} and 13 g P m^{-2} respectively.

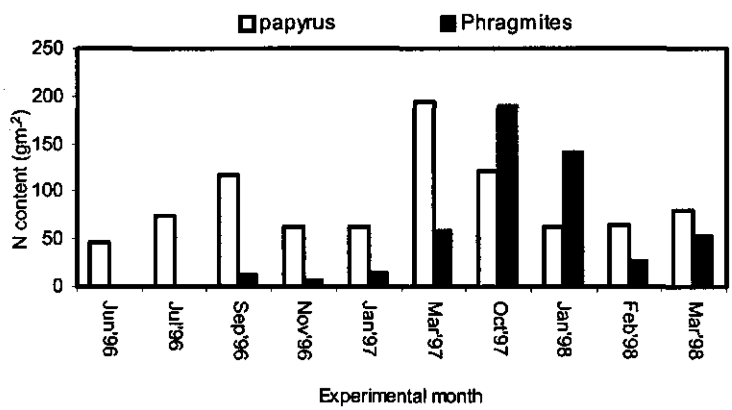


Fig. 4.3. b Variation of nitrogen content per unit area in papyrus and *Phragmites* biomass (g/m^2) from 1996 to 1998.

4.3.4.3 Rooting biomass

The average nitrogen and phosphorus contents (yields) in the rooting biomass computed in a similar manner (using biomass and concentration data in Table 4.1) were as follows: 159.6 g N m^{-2} and 50.8 g P m^{-2} in papyrus and 3.1 g N m^{-2} and 0.57 g P m^{-2} *Phragmites* plants. Considering the whole plant, it is deduced that rooting biomass in papyrus systems accounted for 64% of N and 72 % of P, while in *Phragmites* it accounted for less than 10 % for both nitrogen and phosphorus. This allocation of nutrients between the standing and rooting biomass is critical when periodic harvesting of the biomass is carried out as a management strategy.

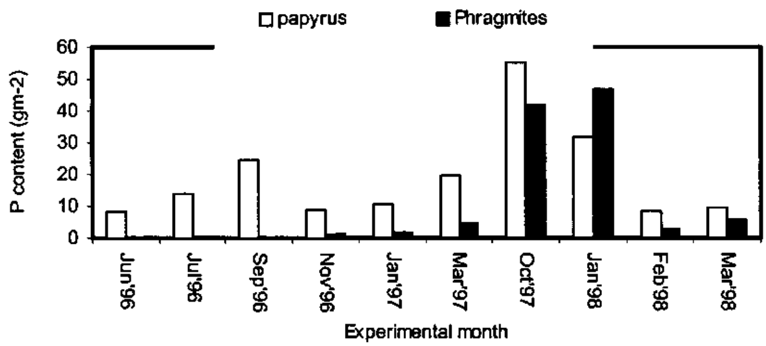


Fig. 4.4 Variation of phosphorus content in papyrus and *Phragmites* shoot biomass (g/m^2) from 1996 to 1998.

4.3.5 Organic matter deposition

The rate of organic matter accumulation to the substratum was only evident in the control (open) pond where it progressively increased from 0.15 % to 0.35 % (weight/weight) within 3 months. There was no observed increase *in Phragmites* ponds. The organic source in the open pond was mainly derived from algal biomass while in the planted ponds, decaying plant parts were the potential source. However, at the time of measurement plant senescence rates were still negligible.

4.3.6 Nutrient Transformations

Phase 1

During this phase (May 96 - March 97), the wetland units were loaded batchwise on a weekly basis and with no disturbance on the vegetation in both papyrus and *Phragmites* ponds. The results from this phase provided basic information on the potential and limitation of nutrient transformations by the wetland systems under the design conditions. The results were used to modify the systems in the subsequent phases. Nitrate and nitrite concentration in the influent, within the wetland and in the effluent were over ten times lower than for ammonium.

4.3.6.1 Nitrogen-ammonium

(a) *Concentration variations*

Ammonium concentration in the influent wastewater was relatively high and variable. The concentrations ranged from 10 - 90 mg/l with a mean of 60 mg/l. The monthly mean concentrations in the effluent from papyrus ponds were low for the 1st three months from the start of the operation but thereafter increased progressively irrespective of the different applied loading rates (Fig. 4.5). The mean effluent concentrations in the individual papyrus wetland units ranged from 35.3 to 47 mg/l, and 19.7 mg/l and 26.4 mg/l in the two *Phragmites* wetland units. The values for the control (open) units were 7.86 mg/l and 16.8 mg/l in P 4 and P 3, respectively. Monthly trends are illustrated in Fig. 4.6. In all the variations illustrated in Figs. 4.5 and 4.6, it is evident that the ammonium concentrations in the effluents from the vegetated units were influenced by shoot density and hydraulic loading to some extent. Dissolved oxygen concentrations and pH in these units decreased with increase in the shoot density.

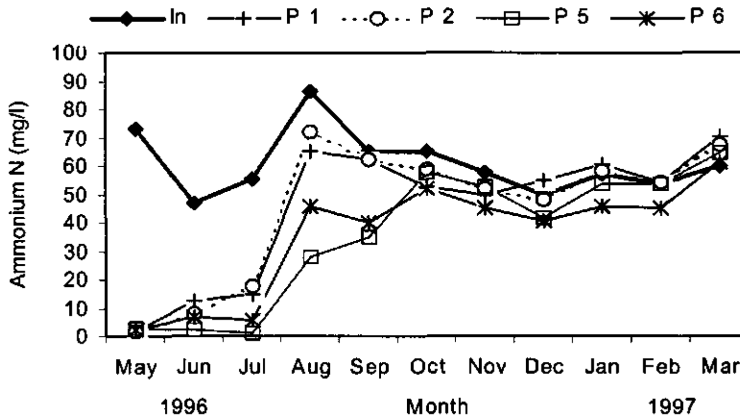


Fig. 4.5 Mean monthly influent (In) and effluent ammonium N concentrations from papyrus wetland units no. 1, 2, 5 & 6 operated at hydraulic loading rates of 5.7, 4.8, 8.4 and 3.2 cm/day, respectively. Each data point is a mean value of 8 data values per month.

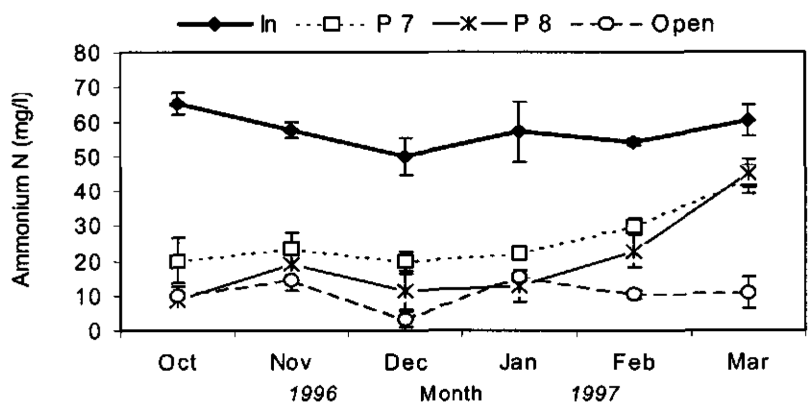


Fig. 4.6 Mean monthly influent (In) and effluent ammonium N concentrations from *Phragmites* wetland units 7 & 8 and the control ponds. Each data point is a mean value of 4 data values per month.

(b) *Mass budgets*

The surface ammonium loading rates applied to the different ponds were in the following ranges: papyrus ponds 1, 2 & 6: 1.1 - 47.4 kg N ha^{-1} day^{-1} and P 5, 1.1 - 99.1 kg N ha^{-1} day^{-1}. *Phragmites* pond 7 & 8 P 7, 3.2 - 17.5 kg N ha^{-1} day^{-1}. For the control ponds 3 &4, the rates were in the range: 3.1 - 21.9 kg N ha^{-1} day^{-1}. The corresponding reduction over the 11 months study period from the individual papyrus ponds 1,2,5 & 6 was: 35.5 %, 31%, 29.2% and 51.7 %, respectively. *Phragmites* units on the other hand attained 63 % and 80 % reduction in ponds 7 and 8, respectively while in the control (open) ponds, reductions of up to 80 % were recorded. This relationship between N surface loading rates and treatment efficiency is illustrated in Fig. 4.7 b where it is evident that high removal of N was attained at lower surface loadings. Meanwhile, Fig. 4.7 a shows the relationship between the ammonium reduction derived from all the papyrus units in the experimental period with shoot density (Fig. 4.7 b). The two are negatively correlated.

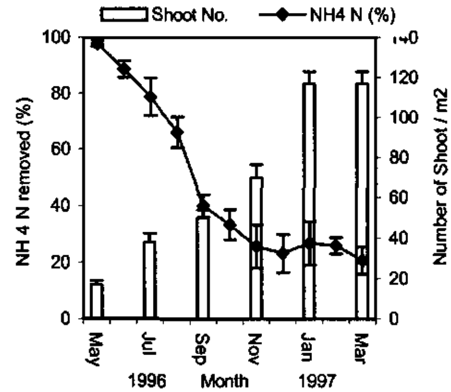


Fig. 4.7 a The effect of shoot density cover on the removal of NH4 N from papyrus units; bars show standard error of the mean, N=16.

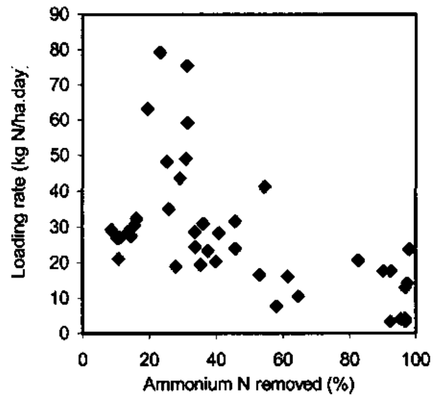


Fig 4.7 b The relationship between surface Loading rates of NH4 N with the percent removal attained in papyrus wetland units

4.3.6.3 Phosphorus (o-phosphate)

(a) *Concentration trends*

Phosphorus concentrations in the influent wastewater were low as compared to the ammonium. Concentrations ranged from 1.4 to 6.5 mg/l. Concentrations of phosphate in the effluent from the papyrus wetland units were relatively low in the first 4 months but gradually increased in all units in the subsequent months (Fig. 4.8). *Phragmites* systems on the other hand, produced effluents with rather steady concentrations as in the open ponds and were consistently lower than the input concentrations (Fig. 4.9). The different hydraulic loading rates applied seem not to have influenced the phosphate effluent concentrations from both vegetated and non vegetated ponds.

(b) *Mass budgets*

Reduction of phosphorus obtained from mass budgets in papyrus systems was characterized by initial elevated removals, maximum values of up to 80% were achieved by the 5th month of operation. Decline followed with the eventual release (minimum values of - 24%) in the 9th operational month (Fig. 4.10). This trend suggested saturation of removal routes within the papyrus units which included adsorption by the walls of the wetlands. In contrast, no release trend was derived from either *Phragmites* or control ponds during this phase which, indicated that its removal routes had not been saturated. The phosphate removed from respective individual papyrus ponds 1,2,5 & 6 for the whole experimental period averaged to: 32 %, 20 %, 34 % and 45 %, respectively. In *Phragmites* ponds 7 and 8, reductions averaged to 41 % and 56%, respectively. It was 48 % and 51% in the open ponds 3 & 4. The effect of substratum on phosphate removal is reflected in these reductions.

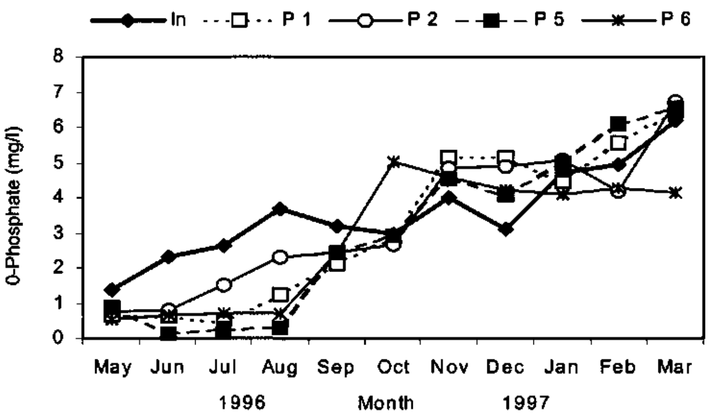


Fig. 4.8 Monthly Influent (In) and effluent o-Phosphate concentrations from papyrus ponds operated at different hydraulic loading rates.

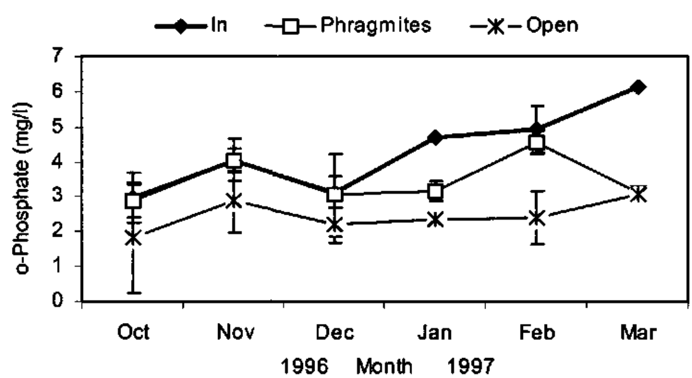


Fig. 4.9 Monthly influent (In) and effluent o-Phosphate concentrations from *Phragmites* and control open ponds.

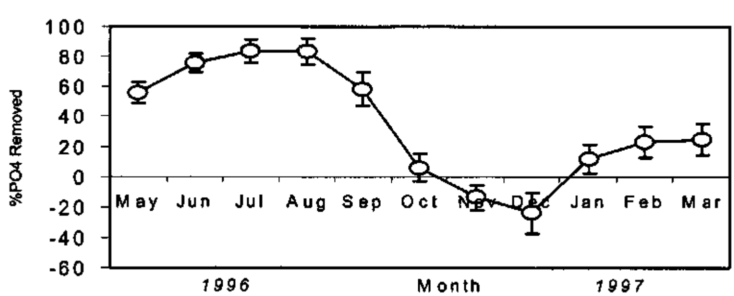


Fig. 4.10 Monthly phosphate mass budgets in *Cyperus papyrus* wetlands; removal and release trends. Vertical bars show the standard error of the mean (n = 12).

Phase 2

In this operational phase (September 97 - March 97), plant biomass was removed from a 10 m^2 area in papyrus P 1 and P 5 and *Phragmites* P 7. A weekly wastewater load format was applied in the first half and a daily one in the second half of this phase.

4.3.6.4 Nitrogen

(a) *Concentration profiles*

The modifications introduced in this phase were intended to increase oxygen transfer into the vegetated systems through surface aeration and algal photosynthesis. Results obtained showed that this objective was fulfilled.

Residual oxygen concentrations in the non vegetated sections of ponds P 1, P 5 and P 7 were significantly higher than in the fully vegetated units (P 2, P 6 & P 8) in the day time (Fig. 3.1, Chapter 3). This created an impact in the ammonium transformation in those sections. Sharp ammonium concentration drops were registered in these non-vegetated sections as compared to the fully vegetated ones as demonstrated in Figs. 4.11 a (papyrus units) and 4.12 a (*Phragmites* units). Similarly, residual nitrate concentrations measured in the open sections were consistently higher than that in the fully covered vegetated zones in both plant systems (Fig. 4.11 b and 4.12 b).

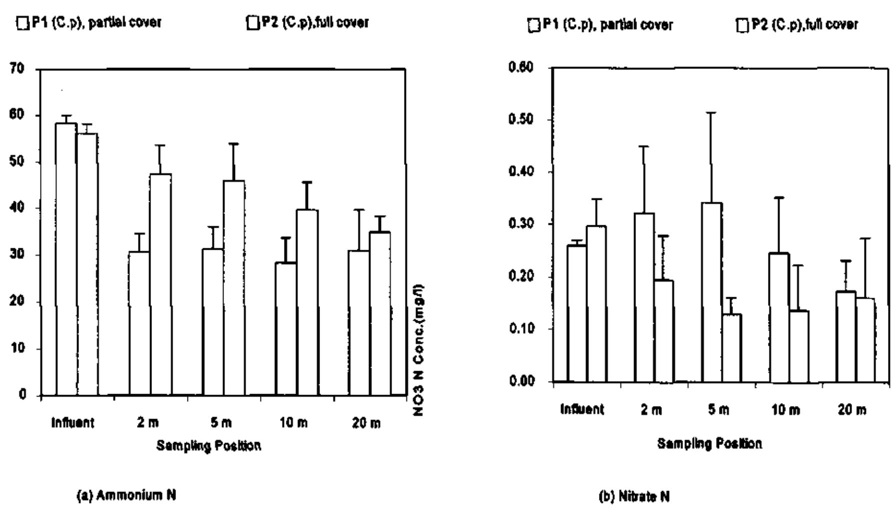


Fig. 4.11 (a) Ammonium N and (b) Nitrate N concentration measured at different positions along the transects in fully vegetated (P 2) and partially vegetated (unplanted 1st 5 m) (P 1) papyrus units. Vertical bars show the standard error of the mean (N = 50).

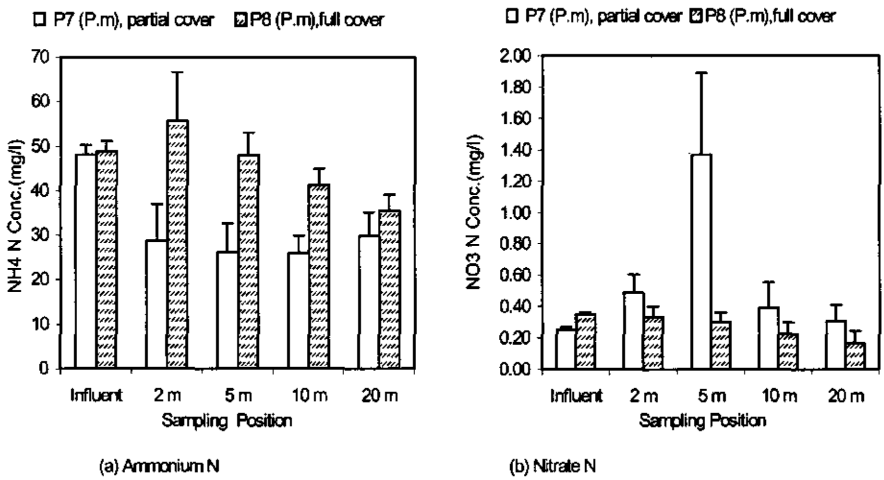


Fig. 4.12 (a) Ammonium N and (b) Nitrate N concentration measured at different position along the transects in fully vegetated (P 8) and partially vegetated (unplanted 1st 5 m) (P 7) *Phragmites* units.

(b) *Mass budgets*

The nitrogen surface loading applied in the two halves of this phase were rather different. In the 1st half (Sep. - Dec. 1997), the loading rates applied were in the ranges: 5.4 - 17 kg N ha^{-1} day-$^{-1}$ in ponds 1, 2 & 3; 6.1 - 13.3 kg N ha^{-1} day-$^{-1}$ in ponds 4 to 8. The reduction attained in each pond is given in Table 4.2. Inspection of these result, reveals that ammonium reductions were consistently higher in partially vegetated ponds than in the fully vegetated ponds operated for the same surface loads; (P 1 & P 2), (P 5 & P 6) and (P 7 & P 8). The difference between the pond pairs demonstrates the effect of the open zones in the processes that control ammonium removal from the systems.

Table 4.2: Mean ammonium concentrations in the influent and effluent and the reduction levels attained in the 1st half of phase 2; vegetation was removed from sections of papyrus ponds 1 & 5 and *Phragmites* pond 7, n = 24.

Pond	Vol. Added (m*3)	Vol. Out (m*3)	NH_4 N. In (mg/l)	NH_4 N Out (mg/l)	HLR (cm/day)	Reduction (%)
C. papyrus 1	5	6.07 ± 0.26	42.31 ± 12.68	18.69 ± 2.99	1.8	46.38 ± 8.3
C. papyrus 2	5	6.84 ± 0.55	42.31 ± 12.68	19.94 ± 4.06	1.8	35.58 ± 12.0
Control 3	5	8.43 ± 0.77	42.31 ± 12.68	6.44 ± 2.40	1.8	74.35 ± 10.9
Control 4	10	8.77 ± 0.58	34.44 ± 5.20	7.69 ± 1.58	3.6	75.53 ± 2.7
C. papyrus 5	10	8.3 ± 0.49	33.94 ± 1.75	24.88 ± 2.81	3.6	23.93 ± 8.3
C. papyrus 6	10	7.94 ± 0.20	33.94 ± 1.75	29.31 ± 3.69	3.6	14.28 ± 9.5
Phragmites 7	10	6.06 ± 1.10	34.44 ± 5.20	20.44 ± 2.64	3.6	55.06 ± 6.9
Phragmites 8	10	8.02 ± 0.95	35.04 ± 5.20	21.81 ± 1.69	3.6	37.58 ± 11.7

The surface loads in the 2nd half although in the same range as those applied in Phase 1, the removal seemed to have been influenced by the daily batch feed format instead. The mass load ranges were 10 - 23 kg N ha^{-1} day-$^{-1}$ for ponds 1, 2 & 3; 20 - 40 kg N ha^{-1} day-$^{-1}$ for ponds 5, 6, 7 & 8. The corresponding mean reductions attained in the individual ponds were: 64%, 53 %, 61 % and 58 % for papyrus ponds 1,2,5 and 6 respectively; 65 % and 66 % for *Phragmites* pond

7 & 8. The control ponds maintained removal above 75 %. For the planted systems, there was a marked improvement in performance in this half as compared to the first. Reductions attained in papyrus ponds 5 and 6 were even significantly different ($p< 0.05$) from that in the first phase.

4.3.6.5 Phosphorus.

Phosphorus concentration changes in papyrus systems continued to fluctuate with export occurring more in Nov- Dec 97 (Phosphorus content in the plant biomass at the same period was high (Fig. 4.3)), as shown for papyrus in Fig. 4.13. In the *Phragmites* systems and the open ponds, no fluxes were recorded and the phosphate reduction attained remained almost unchanged from that obtained in phase 1 with results: 50-52 % and 42-52 %, respectively. No significant differences existed in treatment performance between the different individual ponds in each system and in different systems. Similarly, the phosphorus removal efficiency derived from this phase was not significantly different from that obtained in phase 1.

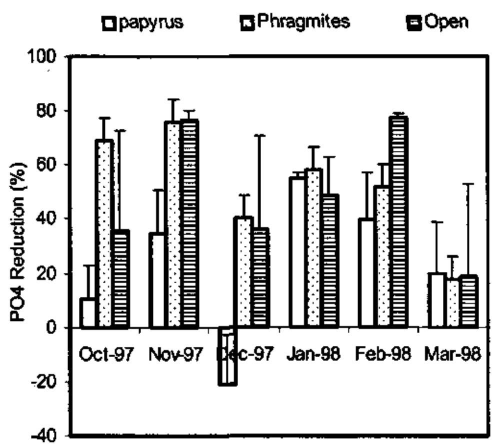


Fig. 4.13. Phosphorus reduction efficiency in papyrus, *Phragmites* and open ponds during phase 2.

Phase 3

In this operational phase, the total pond length of the ponds was increased by joining ponds in series. Ponds 1 & 2 were joined in series (S 1-2) and had a pond configuration with small areas of alternating vegetated and non-vegetated zones. *Phragmites* ponds 7 & 8 were joined with the control open pond 4 to form a series (S 8-4-7) and in a similar way papyrus ponds 5 & 6 were joined with open pond 3 to form series (S 6-3-5). The above system configurations were aimed at deriving optimum conditions that could maximize nitrogen removal as well as other parameters at the same time. Results from the previous studies indicated this possibility. The focus was on stabilizing the critical controlling parameters namely dissolved oxygen concentration, pH, temperature and ammonium concentration.

4.3.6.6 Concentration trends

The environmental factors created and maintained during this format are well described in chapter 3. Their impact in this phase becomes evident with concentration profiles and reductions obtained in the separate pond series. The papyrus series S 1-2, exhibited a sustained reduction in ammonium concentration through the whole length of the system, producing a final effluent which was less than the target 5 mg/l NH_4 N (Fig. 4.14).

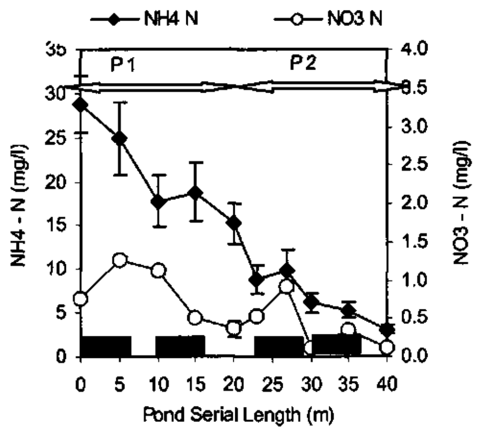
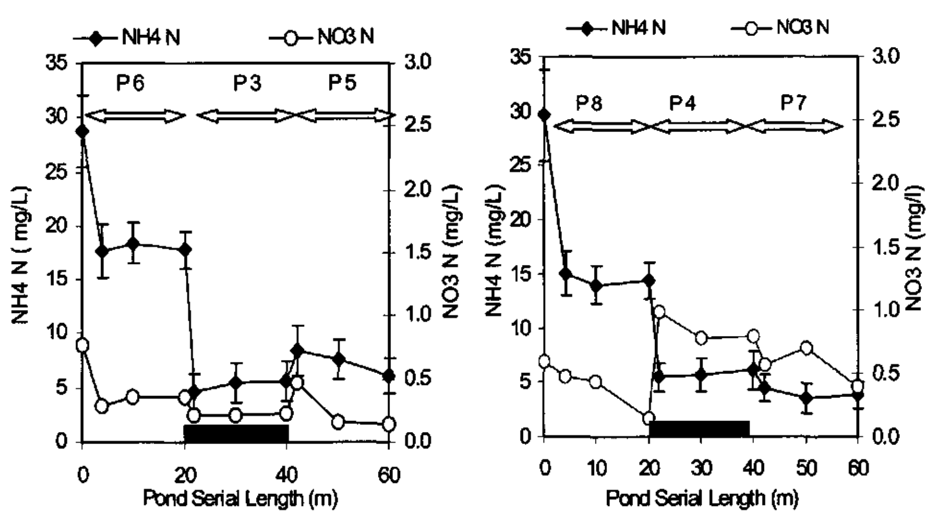


Fig. 4.14 Ammonium and nitrate concentration transect data along the two papyrus ponds 1 and 2 in the series S 1-2; vertical bars show standard error of mean (n =20). Black sections show the non vegetated zones.



(a) (b)

Fig. 4.15 Ammonium and nitrate concentration transect data along (a) papyrus P 6, open pond 3 and papyrus pond 5 in series S 6-3-5 and (b) *Phragmites* ponds 8, open pond 4 and *Phragmites* pond 7 in series S 8-4-7; vertical bars show standard error of mean (n = 15). Black sections show the non-vegetated zones.

The decay profile correlated with the trend of residual oxygen concentrations (Chapter 3, Fig. 3.4). Evidence of increased nitrification was observed in the general increase in nitrate concentration in the small open zones in the series (Fig. 4.14). Since pH values along the series remained nearly constant (7 to 8) depicted in Fig. 3.3, the most probable dominant route for ammonium reduction observed was via the microbial mediated sequential nitrification - denitrification.

Ammonium transformation in the other series S 6-3-5 and S 8-4-7 was characterized by processes that predominated in each of the individual ponds (Fig. 4.15 a and b).The serial combination however, resulted in a final effluent of less than 5 mg/l NH_4- N. Much of the ammonium reduction thus occurred in the first and the second (open) pond of the series.

4.3.6.7 Mass rates

The systems configuration modification introduced in this operational phase resulted into a significant improvement in the removal of ammonium. A graphical plot of loading rate versus corresponding removal rates was linear for all series S 1-2, S 8-4-7 and S 6-3-5 (Fig.4.16). This relationship suggests that the serial systems had a potential to be operated at increased loading (concentration or volume).

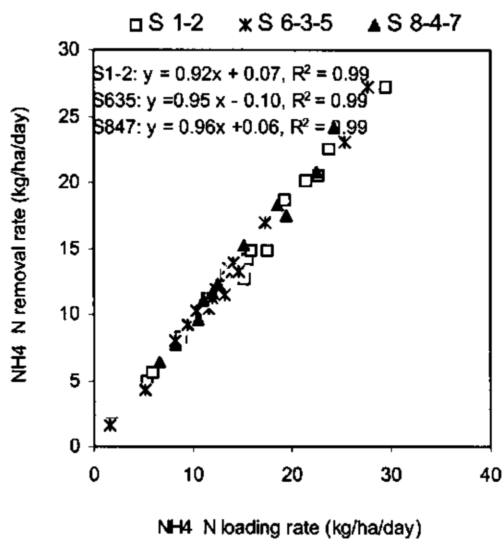


Fig. 4.16 Plots of mass loading of NH_4 N against its removal rate in pond series (S 1-2) , S 6-3-5 and S 8-4-7.

4.4 Discussion

(i) *Nitrogen Removal Capacities*

The main mechanism by which treatment wetlands remove nitrogen from wastewater is identified by many authors to be the microbial mediated sequential nitrification - denitrification (Bowmer, 1986, Patrick, 1990, Hammer and Knight, 1994, Cooper *et al.*, 1996, Vymazal *et al.*, 1998). The extent to which this process can progress is governed by the availability of oxygen (Wittgren and Tobiason , 1995, Kadlec and Knight, 1996). Recent wetland designs and operational modes applied to wetlands referenced in literature have been targeted on ways of increasing oxygen transfer into the wetlands. Direct aeration has been applied in some cases. Oxygen transfer may take place via three natural routes: surface aeration (Brezonik, 1994), translocation by the macrophytes from the shoots to the rhizosphere (Brix, 1990; Armstrong and Armstrong, 1988) and photosynthetic generation by phytoplankton (Muyima *et al.,* 1997).

In this study phase, ammonium removal capacity of the papyrus wetland units decreased from over 80% to an average of 25% within 6 months of operation (Fig.4.7). Residual oxygen concentration correspondingly also decreased from 6 mg/l to less than 1 mg/l. The oxygen supply to the rhizosphere over that duration (Oct 96 - Mar 97) was most probably restricted to the amount translocated by the shoots, since surface aeration and photosynthetic transfer were suppressed by the intense plant cover.

This scenario can be evaluated by assuming that the decrease in the concentration of ammonium in this period was solely due to oxidation (4.2 mg O_2 per 1 mg NH_4N). Using the monthly hydraulic loads, influent and effluent concentrations, the corresponding amount of oxygen that was required for the observed reduction is computed to be in the range of 0.56 to 12.78 g m^{-2} day^{-1} and 1.26 to 5.31 m^{-2} day^{-1} in papyrus and *Phragmites* units, respectively. Oxygen transfer rates by *Phragmites australis* ranging from 0.02 - 12 g m^{-2} day^{-1} have been reported in literature (Armstrong *et al.*, 1990; Brix, 1990). Assuming, the transfer rates are of the same order of magnitude in *Phragmites mauritianus,* then range of oxygen concentrations derived from plant roots could account for the ammonium removed. However, in the papyrus wetland units, the oxygen consumption requirement is much higher than that can possibly be transferred by plant roots considering the oxygen release rates of only 0.017 g m^{-2} day^{-1} reported by Kansiime and Nalubega (1999). Given that the oxygen flux from the shoots is mostly utilized by roots and rhizomes for their respiratory needs and it is only the remainder that is competed for by heterotrophic and nitrifying bacteria which are even slow growers, other routes must have therefore contributed to the observed nitrogen removal. It is suggested that other mechanisms such as matrix adsorption and uptake of ammonium and nitrate ions by plants were playing a substantial role in N removal. The latter aspect was demonstrated in this phase.

Treatment capacities obtained in the first half of the 2nd phase, were particularly low (Table 4.2). The cause is thought to be the high rainfall and reduced evapotranspiration rates experienced over the entire period of measurement. The high rainfall and the localized flooding created contributed to a drastic reduction of the retention time of wastewater in the wetland which in effect reduced the effectiveness of the treatment. This effect was also augmented by the reduced evapotranspiration rates that prevailed at the time (Chapter 2, Fig. 2.4). From these results, it is suggested that the operation procedure adopted in a constructed wetland should be responsive to rainfall effects. For instance, raising the weir levels so that the risk of flushing of the systems in the event of a storm is reduced. An allowance for elevated embankments may also be necessary in the design so that effects of local flooding are minimized.

During the operation in phase 3, volumes transferred between the contributing ponds in the series were recorded and it was possible to compute treatment efficiency of ammonium removal achieved after flow through each of them (Fig.4.17). Removals > 90 % were derived from papyrus series S 1-2 an S 6-3-5 and *Phragmites* series S 8-4-7. Performance of series S 1-2 and that based on sub-series with the same surface areas (S 6-3 & S 8-4) showed no significant difference ($p= 0.95$) within each category or between categories. Both were significantly much higher than removals derived from individual ponds in the series (S 1, S 6 & S 8). Such high removals in wetland systems have only been recorded in advanced wetland treatment using vertical down flow reed beds which tend to promote faster nitrification (Watson and Danzig, 1993; Laber *et al.*, 1997; Felde and Kunst, 1997).

The serial pond configuration with alternating small vegetated and non-vegetated areas (series

S 1-2) promoted oxygen transfer through surface aeration, plant translocation and, to a limited extent, by photosynthesis. Therefore oxygen demands for both the organic matter and ammonium were met. The calculated oxygen demand for ammonium oxidation to give concentrations of NH_4 N in the effluent of 5 mg/l using the same assumptions above, was 3.17 g m^{-2} day^{-1}. In fact, ammonium reduction achieved was lower (3 mg/l) suggesting that oxygen supply by the systems was satisfied even disregarding the competitive requirements of the microbiota and organic carbon degradation. However, when only one pond of the series is considered, S 1, an oxygen demand of 6.33 g m^{-2} day^{-1} is calculated but the effluent concentration obtained (17 mg/l), suggested an oxygen deficit of 3.6 gm^{-2} day^{-1} Therefore, the size (surface area) of the wetland (planted and open zones) is also a critical factor in the design.

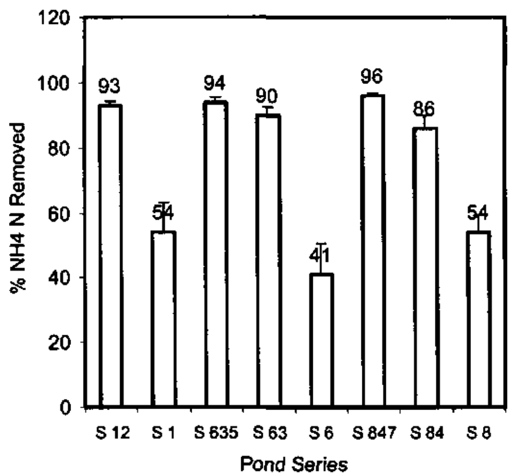


Fig. 4.17 Percentage reduction of ammonium removed based on the effluent derived from the end of each pond serial combination and/or individual ponds within the series.

The observed ammonium reduction in the first vegetated papyrus pond 6 and *Phragmites* pond 8 of series S 6-3-5 and S 8-4-7 suggests that oxygen transfer by shoots was still occurring and therefore ammonia loss via nitrification - denitrification route was possible. At the prevailing day pH (> 10) and temperature (27 ^{0}C) in the open ponds, over 87 % ammonium is present in form of ammonia gas (NH_3) (Emerson *et al.*, 1975). Therefore, the ammonium loss in these zones was predominantly via volatilization of ammonia gas. Nitrate concentration might have been expected to build up in the open ponds and open sections of S 6-3-5 and S 8-4-7 due to the high dissolved oxygen (> 10 mg/l) available, but this was not recorded. Instead, low (< 1 mg /l) and steady nitrate concentrations existed (Fig. 4.15 a and b).This trend was probably a result of two factors that are found to limit or inhibit the nitrification process. The first is the low residual substrate (ammonium) concentration, most of it was lost rapidly through volatilisation and secondly, the high pH that prevailed in the day inhibited the activity of the nitrifiers (Henze *et al.*, 1977).

In the papyrus series S 1-2 (small areas of alternating vegetation / non vegetation), at the pH range of 7 - 8.5 and water temperatures of 23 ° C, less than 4.4 % ammonium N is present in form of ammonia gas (NH_3) (Emerson *et al.*, 1975). Therefore, volatilisation loss was not significant. Instead, ammonium removal that occurred was associated with nitrification-denitrification route and plant uptake. The ammonium concentration decline along this series was exponential (Fig. 4.18) and therefore could be described by the first order kinetic model suggested by Reed *et al* (1995). The model is based on the assumption that all the Kjeldahl nitrogen (TKN) is converted to ammonium which is then removed via the nitrification-denitrifcation route.

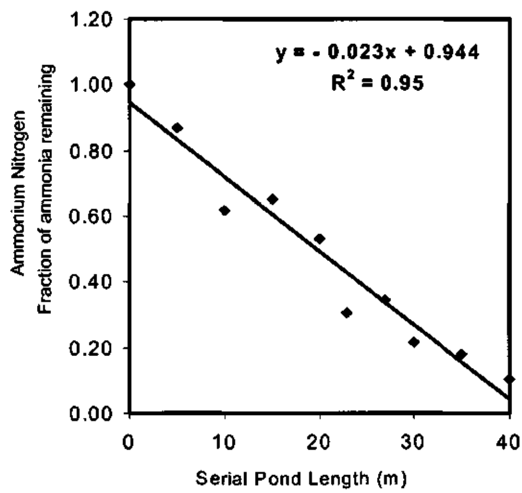


Fig. 4.18 Transect data for ammonium concentration in the papyrus wetlands units 1 and 2 in series (S 1-2).

The actual contribution of the nitrification-denitrification route to N export in series S 1- 2 was estimated using this first order equation given as:

$$\ln \frac{C_i}{C_o} = A_s \cdot \frac{k.h.n}{Q} \tag{4.5}$$

where:
A_s = surface area of wetland (m^2)
Q = average flow through the system (m^3/day.)
C_o = effluent ammonium (mg/l)
C_i = influent TKN (mg/l)
k = temperature dependent rate constants (day^{-1})
= $0.2187(1.048)^{(T-20)}$
h = depth of water in the wetland (m)
n = wetland porosity (volume occupied water), as decimal fraction
T = temperature of the water in the wetland (^{0}C)

When the operational data applied to series S 1-2 (Q = 3.8 m^3/day, n = 0.70 and at a depth of 40 cm, Ci = 31.3 mg/l and at a temperature of 23 ° C) are substituted to equation 4.5, the effluent concentration expected and assuming that all is removed via nitrification-denitrification route was derived as 7.64 mg/l. This value is close to the average effluent concentration obtained from experimental measurements in this series (5 mg/l).The difference with the measured concentration can be attributed to other ammonium removal mechanisms such as adsorption and plant uptake as explained later in sections (iii) and (iv). From mass budget computation, at this effluent concentration (7.64 mg/l), 77 % ammonium removal is calculated, which can all be associated with the nitrification-denitrification pathway.

(ii) *Phosphorus.*

Phosphorus removal from wastewater by wetlands is to a large extent controlled by biogeochemical processes (Kadlec and Knight 1996). Adsorption to soil or substrata is suggested to be the principal mechanism that contributes to much of the removal (Wood, 1990, Mann, 1990). Other routes such as uptake by phytoplankton, other algae and plants are recognized to offer only temporal phosphorus storage (Richardson and Craft, 1993, Schreijer *et al.*, 1997).

In this study, phosphorus dynamics were evaluated based on the controlling conditions set for optimizing nitrogen and organic matter removal in the wetland units. An earlier preliminary study in Uganda by Sekiranda (1996) using buckets planted with either *Phragmites* or papyrus on a gravel substrate indicated a phosphorus removal of up to (93%). This removal was partitioned between plant uptake (37%) and sorption (58%). This finding suggested that the Kirinya quarry gravel had a considerable potential for P removal in a bigger system. However, results from the laboratory sorption studies, showed that P sorption by the substrata was rather low (average 27%). Nevertheless, phosphorus removal rates recorded in *Phragmites* systems (≈ 45%) were consistently high in comparison to that obtained from papyrus (without substrata) (34 %) (Fig. 4.14).

The performance achieved in both systems was within the range in temperate wetlands that are operated at a large scale as indicated by the results of Wood and Hensman (1989), 31 % and, Brix and Schierup (1989), 20 - 40 %. In contrast, P removal derived from small sized wetland systems, tends to be high (60 - 100 %) (Drizo *et al.*, 1997; Sekiranda, 1996; Burgoon *et al.*, 1991). Initial optimism cited in early literature on the wetland P removal capacity was probably based on result generated from such systems but which are now not reflected in actual field conditions.

The removal trend described in papyrus systems (Fig. 4.10) was indicative of saturation of P removal routes which in this case (no substrata) were limited to uptake and sorption by the heavy root - rhizome mat structure and walls of the wetland. The decrease in phosphate reduction in this study can be associated with two main factors; release of stored phosphorus by the decaying litter, phytoplankton and peripyton and the blocking of sorption sites by a biological slime layer that was visibly covering the substratum as well as the dead rooting biomass. Organic anion competition is suggested by Johansson (1997), as another contributory factor in the reduction of P sorption sites. In the study, the effect of this factor was not quantified but given the high rates of organic decomposition in these wetlands (Chapter 3), its effect on P removal was probably significant.

The relatively high phosphorus reduction obtained in the open ponds can be associated with algal uptake; this evidence is derived from earlier summer studies in lagoons in the temperate regions (Picot *et al.*, 1993;, Reddy and D'Angelo, 1997; Muyima *et al.*, 1997). The phosphorus up-taken and stored in the algal biomass gets exported from the wetland when the algae is discharged as effluent. This in effect, is counterproductive if the effluent enters a water body where the phosphorus can become available when the algae die.

(iii) *Biomass yields*

The role of the plants as stores of nutrients was evaluated using data generated in two phases which spanned through both the growing and maturation of the plants. The rooting biomass of papyrus was substantial (53% of the total biomass) as compared to 10 % in *Phragmites* and its contribution as a storage compartment for nutrients was significant. The rooting biomass growth rates of papyrus were apparently different from that of the standing biomass. Biomass growth rates of 470 kg ha^{-1} day^{-1} and 310 kg ha^{-1} day^{-1} were determined for the rooting and standing biomass respectively over the first five months after planting. The rooting biomass growth rate declined to 200 kg ha^{-1} day^{-1} in the subsequent months but the standing biomass increased exponentially instead. The standing biomass growth rate for *Phragmites* was comparatively low (113 kg ha^{-1} day^{-1}) by the 4^{th} month after planting.

The growth rates (standing biomass) of 654 and 326 kg ha^{-1} day^{-1} for papyrus and 1300 and 256 kg ha^{-1} day^{-1} for *Phragmites* were computed when the plants exhibited exponential (five months after planting) and stationary growth (starting 15 -18 months after planting) trends, respectively. The growth rates for papyrus are greater than what is found in natural wetlands. Muthuri *et al* (1989) reported rates ranging from 124-155 kg ha^{-1} day^{-1} in a typical wetland and 410 kg ha^{-1} day^{-1} in papyrus growing in fertile soils. Kansiime and Nalubega (1999) obtained growth rate of 130 kg ha^{-1} day^{-1} in the Nakivubo wetland while Lizhiboa (1995) in the bucket experiments at Kirinya obtained doubtful high rates of up to 1560 kg ha^{-1} day^{-1}. Meuleman (1999), reports primary production rates of 192 kg ha^{-1} day^{-1} and 88 kg ha^{-1} day^{-1} for *Phragmites australis* growing in an infiltration and natural wetland, respectively. The high growth rates obtained at the Kirinya pilot wetland is a result of a constant nutrient and water availability which is not the case at all times in natural wetlands.

(iv) *Nutrient yields*

The varying biomass obtained in the different phases had a direct correlation with the total nutrient content stored and uptake rates by the plants. The nutrient storage capacity of the rooting biomass for the period examined averaged to 1600 kg N ha^{-1} and 508 kg P ha^{-1} in papyrus and 310 kg N ha^{-1} and 57 kg P ha^{-1} in *Phragmites*. Nutrient uptake rates averaged to 0.63 kg N ha^{-1} day^{-1} and 0.42 kg P ha^{-1} day^{-1} in papyrus. Different nutrient uptake rates were exhibited in the standing biomass at the exponential and stationary growth phases. Nutrient uptake rates of 7.1 kg N ha^{-1} day^{-1} and 0.24 kg P ha^{-1} day^{-1} in papyrus and 10.4 kg N ha^{-1} day^{-1} and 0.26 kg P ha^{-1} day^{-1} in *Phragmites* were obtained at the exponential phase. These rates reduced to 0.34 kg N ha^{-1} day^{-1} and 0.06 kg P ha^{-1} day^{-1} in papyrus and 0.18 kg N ha^{-1} day^{-1} and 0.015 kg P ha^{-1} day^{-1} in *Phragmites* during the stationary phase. These results demonstrate the importance of defining the timing of the nutrient uptake measurements. Uptake rates measured in the exponential growth phase give high values while low values tend to occur at steady state growth stage. Maintaining plant systems in the exponential growth phase may be a desired management strategy that can be applied in wetland systems in the tropical environments.

Nutrient uptake values obtained at stationary growth phase were comparable with those reported in natural systems. Reddy and DeBusk (1987) report uptake in *P. australis* of: N = 0.493 - 0.575 kg ha^{-1} day^{-1} and P = 0.055 - 0.082 kg ha^{-1} day^{-1}. For papyrus, Gaudet (1976, 1977) working in Kawaga wetland in Uganda reports uptake values of 1.30 and 0.06 kg ha^{-1} day^{-1} for N and P, respectively. Muthuri *et al* (1989) reported values of 1.35 and 0.192 kg ha^{-1} day^{-1} in the Lake Naivasha wetland. Kansiime and Nalubega (1999) obtained rates ranging from 0.468 to 2.08 kg

ha^{-1} day^{-1} for N and 0.055 to 0.34 kg ha^{-1} day^{-1} for P in different locations and growth stages in the Nakivubo wetland.

(v) *Mass balance*

The nutrient mass balance is worked out on the simplified generic mass balance equation (Kadlec and Knight, 1996) which is given as:

$$Q_i.C_{ki} - Q_o.C_{ko} - J.A = [V.C_{ks}]_t - [V.C_{ks}]_0$$

where:

Q_i	=	System inflow (m^3/d),
C_{ki}	=	concentration in inflow (gm^{-3}),
Q_o	=	system outflow (m^3/d),
C_{ko}	=	concentration in the outflow (gm^{-3}),
J_k	=	spatially averaged removal rates ($gm^{-2} d^{-1}$)
A	=	area of wetland (m^2)
V	=	storage or wetland volume (m^3),
C_{ks}	=	concentration in the wetland surface water (gm^{-3}),
t	=	time(day)

Figures 4.19 and 4.20 illustrate the mass balance variables for nitrogen and phosphorus over the two phases.

The underlying assumptions in the mass balance analysis include: no groundwater recharge or surface run- off into the wetland, production rate of nutrients within the wetland and the concentration of ammonium in the rainfall are negligible. The concentrations and flow in the equation are weighted averages which gives a more accurate interpretation of the variability of the data for set time intervals.

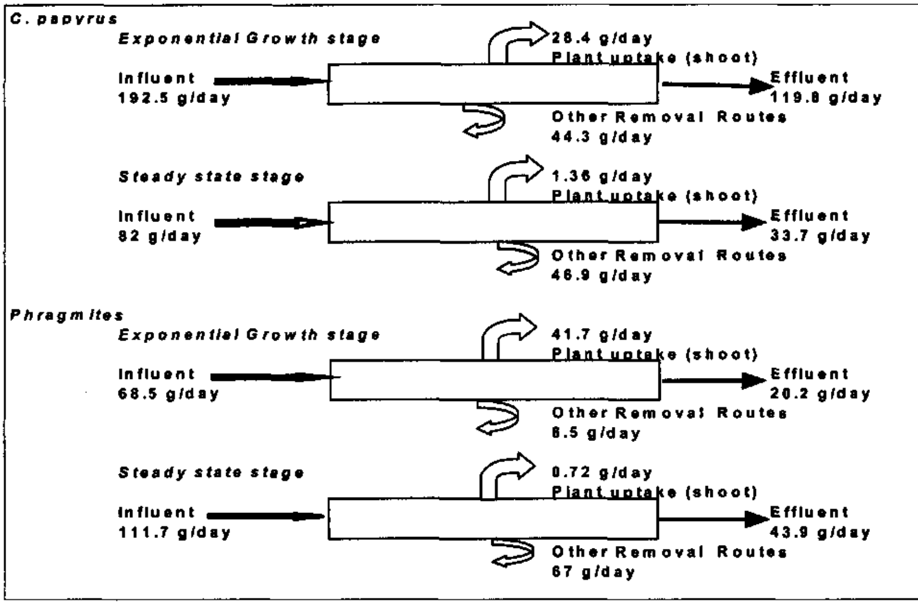


Fig. 4.19 Nitrogen mass balance during phase 1 and phase 2 modes in an interval of 70 days in papyrus and *Phragmites* wetland systems; all values expressed as gram N.

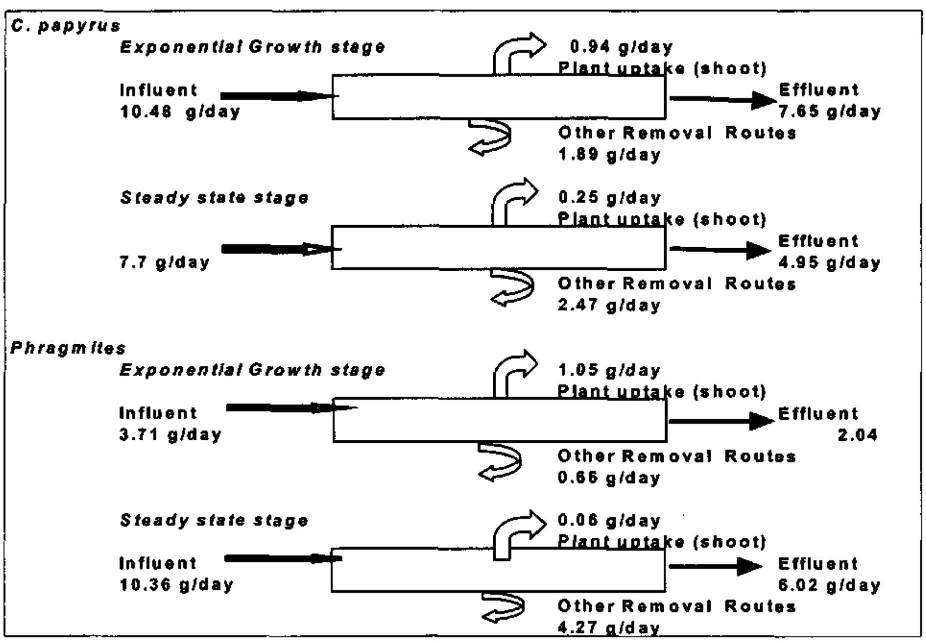


Fig. 4.20 Phosphorus mass balance during phase 1 and phase 2 modes in an interval of 70 days in papyrus and *Phragmites* wetland systems, all values expressed as gram P.

Note: (i) pond water = change in storage of the water volume in the wetland. (ii) Other removal routes = root biomass uptake plus other routes by which N or P is removed.

The terms on the right hand of the equation represent storage of the wetland (what is present in the wetland water). Input nitrogen concentration was presented as total Kjeldahl nitrogen (TKN), assuming TKN = [NH4 N]/0.6. The contribution of nitrate and nitrite was considered negligible given that their concentration was more than ten times lower than that of ammonium.

The mass balance values show a consistent contribution of plant uptake to the overall nutrient removal from wastewater during the exponential growth phase. Nutrient removal via plant uptake (standing biomass) pathway contributed 15 % N and 10 % P of the total input into the papyrus wetland units and 58% N and 37% P in *Phragmites* systems. In the stationary phase, uptake contribution reduced to 1.83 % N and 3.2 % P in papyrus and to 0.6 % N and 0.58 % P in *Phragmites* systems. The amount of nutrients taken into the rooting biomass was variable but the contribution was more in the papyrus systems than in the *Phragmites* based on the biomass yields of the two systems.

These findings highlight the source and reasons for conflicting data on the role of plant nutrient uptake in wastewater treatment cited in literature. Previous reports by Gersberg, *et al* (1986), Wathugala *et al* (1987), Herskowitz *et al* (1987), Haberl and Perfler (1991), Brix (1994), Kadlec and Knight (1996) and Geller (1997) have indicated that the contribution of nutrient uptake to the overall removal of nutrients from wastewater flows as insignificant. The deductions from this study show that this is not always the case. Nutrient uptake pathway is significant at the exponential growth stage. In the tropics, plant growth is not subject to seasonal weather variations as is the case in temperate climates and it is possible to maintain the plants in the exponential growing stage by a well planned harvesting regime. Koottatep and Polprasert (1997) working in tropical environments report, achieved consistent nitrogen removal by this method. The storage of nutrients in the rooting biomass is often not considered in most of the evaluations cited in literature. However, for harvesting and re-growth exploitation, this component is important as it will influence the rate at which the plants re-establish after harvesting. Papyrus plants with a high nutrient storage capacity (section iv) have an advantage in this regard over *Phragmites* with a small rooting biomass.

4.5 Conclusions

This study has demonstrated both the capacity and limitations of tropical constructed wetlands with respect to nutrient removal from wastewater. The following specific aspects were deduced;

(1) The substratum (gravel) utilised had a low phosphate retention capacity since phosphorus mass reduction in the *Phragmites* wetland units (45 %) was not significantly different from that in the papyrus units (34%). The systems were susceptible to phosphorus saturation and this limited their ability to remove phosphorus on a continuous basis.

(2) Ammonium removal in fully developed wetlands was minimal because of the reduced oxygen availability. The removal was enhanced by creating open surface areas adjacent to the vegetated sections.

(3) Ammonia (NH3) volatilisation was a significant route for nitrogen loss in wetlands with large open surface areas. In a wetland arrangement comprising of several small alternating vegetated and non-vegetated zones, nitrification - denitrification pathway was the principle route for ammonium loss. This type of wetland configuration is the recommended option.

(4) The rooting biomass of papyrus is nearly 50% of the total biomass as compared to 10

(4) The rooting biomass of papyrus is nearly 50% of the total biomass as compared to 10 % of *Phragmites* and it is therefore an important nutrient storage compartment in the wetland which is critical for re-growth when plants are cut off.

(5) Nutrient uptake into the standing biomass was only a significant route for the removal of nutrients when the plants were in the exponential growth stage (young) but became insignificant when the plants reached the stationary state (mature). The removal route can be maximised by maintaining the plants in the exponential growth stage, for example by periodic harvesting.

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Chapter 5

Distribution and Removal of Faecal Coliforms

Abstract....109

5.1 Introduction....109

5.1.1 Removal of pathogens by constructed wetlands....111
5.1.2 Scope of this study....112

5.2 Materials and Methods....112

5.2.1 Vertical profiles of faecal coliforms....113
5.2.2 Physico-chemical measurements....113
5.2.3 Measurement of faecal coliform numbers....113
5.2.4 Influence of hydraulic retention time....114

5.3 Results....114

5.3.1 Phase 1....114
5.3.2 Phase 2....116
5.3.3 Phase 3....117

5.4 Discussion....118

5.5 Conclusions....121

References....121

Abstract

The capacity of faecal coliform removal by a tropical constructed wetland receiving pre-settled municipal wastewater was investigated at Kirinya, in Jinja from 1996 to 1999 under three different wetland operational phases. The influence of the hydraulic retention times on the faecal coliform removal in each phase was determined. Within the wetland, the faecal coliform population distribution was found to be uneven; high numbers were found in the waters at the free flow zone, below the mat but significantly reduced populations were in the root–mat zone and within the substratum matrix. Faecal coliform removal efficiencies above 3 log units were obtained in all the wetland units. Removals above 4 log units were only achieved in those units operated at retention times above 7 days. The faecal bacterial counts in the effluent from the different pond systems were on average less than 10,000/100ml, which is the Uganda National Standards limit for discharge of wastewater to water or land. Under the wetland configuration with alternating open water and planted zones applied in Phase three, faecal coliform counts in the effluent from all the three series were less than 2000/100ml. The wetland configurations adopted in the third phase are recommended for application in similar environments since high reductions of ammonium nitrogen, biochemical oxygen demand and total suspended solids were obtained as well.

5.1 Introduction

The fundamental purpose of collecting and treating wastewater is the protection of public health. All the different wastewater treatment methods and operational procedures applied target at achieving this central issue. However, this task is often not realised especially in the developing countries for several reasons but mainly because of social and economic constraints. The need for utilising alternative technologies such as constructed wetlands in wastewater management is essential in such circumstances. Several study reports have demonstrated the potential of natural and constructed wetlands in reducing the populations of various types of pathogens to very low concentrations and with reduced public health risk. The categories of pathogens, their exposure routes to the communities and the levels of treatment achieved under the conventional treatment methodologies are reviewed in the next paragraphs.

(a) *Pathogens in wastewater*

Human excreta (faeces and urine) are the main source of pathogens in domestic wastewater. The pathogens are in four categories, namely viruses, bacteria, protozoa and helminths. Their population in wastewater is highly variable and depends on the health of the population contributing to the wastewater (Krishnan and Smith, 1987 and Rowe and Abdel-Magid, 1995). Feachem *et al* (1983) suggested that the public health risk from untreated wastewater is directly linked to exposure of the community to these pathogens. Communities or individuals may get exposed to untreated or partially treated wastewater in a variety of ways which include among others, direct contact through abstraction of drinking water down stream and diversion of wastewater receiving streams for irrigation purposes. It may also occur indirectly via aerosol transport and the food chains (Krishnan and Smith, 1987 and Reed *et al.*, 1995).

(b) *Removal of pathogens by common wastewater treatment systems*

The removal of pathogens using the common wastewater treatment systems takes place in the different sections but the extent varies, depending on the pathogen type, wastewater characteristics, environmental conditions and sometimes the yearly seasons.

(i) *Septic tank systems*

The reduction of pathogens in septic tanks as documented by Feachem *et al* (1983), are effected through settlement of solids, protozoa and cysts to the sludge layer and by adsorption of bacteria onto the suspended solids which eventually also settle down. Retention time and design of the septic tank determine the extent of pathogen reduction that may be achieved. Faecal coliform bacteria removals in the range of 50 – 95 % are cited in the reports of Brades (1978) and Tebbut (1983).

(ii) *Conventional treatment systems*

Data from literature show that the extent of pathogen removal achieved in conventional treatment systems varies at different unit operations. The initial stage of pre-treatment by screening will have no effect on the pathogen population in the sewage. At the sedimentation stage, a few of the pathogens get removed, especially the larger and heavier ones such as eggs of helminths, cysts and protozoa (EPA Manual, 1992; Metcalf and Eddy, 1991). They tend to settle out together with particulate matter associated with the micro-organisms in a similar manner to septic tanks. Removal of bacteria of less than 24 % and between 50 – 90 % for protozoa are reported at this treatment stage (EPA, 1992). The secondary treatment stage is found to be where substantial reduction of faecal bacteria and other pathogens takes place (WPCF, 1984; EPA, 1992). Up to 90–95 % faecal coliform reductions have been reported in trickling filter units (Wheater *et al.*, 1980) and 80 – 99 % in activated sludge systems (EPA, 1992; Rowe and Abdel-Magid, 1995). Despite the large pathogen reduction obtained from these treatment systems, residual pathogen populations often remain high and still pose a public health risk when the wastewater gets discharged to receiving water bodies or through unrestricted reuse. In countries with sufficient resources, terminal disinfection with chlorine before discharge is carried out.

(c) *Waste stabilisation ponds*

A very high degree of pathogen reduction is achieved by use of the waste stabilisation treatment systems in the warmer regions of the world. The systems comprise a series of anaerobic, facultative and maturation ponds that provide conditions that are conducive for enhanced pathogen die-off. Processes such as sedimentation, predation and photolysis are the principal routes for reduction and these are augmented by the long retention times that are applied. The effluent from these systems has a low concentration of pathogens, especially helminths eggs and protozoa. Faecal coliform reductions of up to 99.99 % have been attained in waste stabilisation systems in Brazil (Feachem *et al.*, 1983). The application of waste stabilisation ponds in Eastern African regions as a simple low cost and highly efficient wastewater treatment option has been on the increase over the past 15 years (Mara *et al.*, 1992). In Uganda, apart from two existing conventional municipal wastewater treatment systems, the majority of the existing municipal treatment plants are waste stabilisation ponds. A summary of the extent of removal of selected pathogenic organisms by the common wastewater treatment processes is given in Table 5.1

Table 5.1. Summary of the percent removal of selected pathogenic micro-organisms by various wastewater treatment processes.

Organism	Primary Treatment	Activated sludge	Trickling filters	Stabilisation pond
Bacteria coliform	48	91 - 98	97	99.96
Faecal coliform	-	99 – 99.9	95 – 97	99.6
Salmonella spp	15	85 – 9	70 – 99	99.99
Virus	<30	30 – 40	76	50 – 80
Amoebic cysts	50	Not affected	83 – 99	100
E. histolytica	Nil	0 – 99	10 – 99.9	-
Helminths ova	90	-	18 – 26	100
Ascaris ova.	66	93 – 99.2	77 – 99.8	100

(Compiled from Shahalam, 1989; Mara *et al.*, 1992; Rowe and Abdel-Magid, 1995).

5.1.1 Removal of pathogens by constructed wetlands

Initial studies on the application of constructed wetlands in wastewater treatment were focused on the bulk pollutants (BOD, COD, TSS). However, earlier investigations by Gersberg *et al* (1987) showed that constructed wetlands were effective in removing pathogenic organisms as well. Subsequent studies have further demonstrated that vegetated wetlands operated either as a pilot facility or full-scale, are effective in removal of faecal coliform bacteria. Gersberg *et al* (1989) and Haberl and Perfler (1990) reported faecal coliform removals of 98 %, while Schreijer *et al* (1997) report reductions of up to 90–95 %. Ottova *et al* (1997) obtained removals of 99 % in all but one of their wetlands investigated and Terry (1993) and Choate *et al* (1993) reported much higher reduction high levels (99.9%). Summary data on removal of indicator organisms derived from 21 North-American treatment wetland systems (Kadlec and Knight, 1996) shows performance ranging from minus 182 % to 99.9%. The negative removal range illustrates input of faecal coliform from other sources within the wetland other than the influent wastewater.

The processes responsible for the reduction of pathogen populations in wetland treatment systems are known to be controlled by a combination of physical, biological and chemical factors (Brix, 1993; Vincent *et al.*, 1994; Cooper *et al.*, 1996). The key physical processes are sedimentation and attachment to the rooting biomass and other solids such as broken stems, umbels etc that may be present. Both processes tend to increase the retention time of the faecal bacteria in the wetland and thereby enhance die-off by other routes. The nature and density of the rooting biomass can greatly influence the extent of faecal bacteria removal via these processes. This influence was demonstrated in the studies of Kansiime and Nalubega (1999) in a natural wetland where faecal coliform counts were consistently higher in zones dominated by *Miscanthidium violaceum*, than in zones dominated by papyrus. The rooting mat of the former was tight and compact and thus had a reduced total surface area. In contrast, papyrus mat is hollow and interwoven giving it a larger surface area for entrapment and attachment of faecal coliforms.

The faecal coliform bacteria and other pathogens, which get attached to the rooting media or are settled at the bottom, are further subjected to chemical and biological degradation. Some of the suggested routes include chemical oxidation, photolysis, predation by higher organisms and exposure to biocides released by plant roots and natural die off (Seidel, 1976; Lijklema *et al.*, 1987; Scheuerman *et al.*, 1989; Cooper *et al.*, 1996). The contribution of each of the above routes is suggested to be a function of wastewater flow rates, nature of the plants and type of the wetland (Williams *et al.*, 1994).

The decay of pathogens in surface flow wetlands (SF) is described by many authors to be a first order process. The exponential decline in bacterial populations on wetland may be expressed by equations 5.1 or 5.2 (Kadlec and Knight, 1996).

$$\frac{C}{C_i} = \exp\left(-k_v.t\right) \tag{5.1}$$

Or

$$\frac{C}{C_i} = \exp\left(-\frac{k_T}{q}\right) \tag{5.2}$$

Where:

- C = bacterial population, No/ 100 ml at time , t
- C_i = influent bacterial population, No/100ml
- k_v = first order volumetric rate constant (day $^{-1}$)
- k_T = first order areal rate constant (m/day)
- q = hydraulic loading rate (m/day)
- t = time (days).

First order rate constants, k_v, ranging from 0.29 to 0.86 day^{-1} have been obtained in different wetland systems (Gersberg *et al.*, 1987; Bavor *et al.*, 1987; Gearheart *et al.*, 1989).

5.1.2 Scope of this study

The application of the wetland technology for wastewater treatment in Africa and in East Africa in particular, will to a large extent be dependent on the ability of these systems to produce effluent free or with low concentrations of faecal bacteria. Presently, there is no existing data or study that has been undertaken to investigate the performance of treatment wetlands systems with respect to faecal matter removal in the region. Given the unique conditions in the tropics, this study was formulated with overall aim of generating performance data under such environments.

The specific objectives of the study included:

(i) Determination of faecal coliform bacteria removal efficiency during all the three phases and their distribution within the wetland,

(ii) Determination of the optimal hydraulic retention times and the faecal coliform decay rates.

5.2 Materials and Methods

The design and the layout of the study site and the vegetation type at the Kirinya pilot constructed wetlands, is described in detail in Chapter two. The configuration set up of the ponds in the three operating phases is also described therein.

(i) *General approach*

The experimental work on performance of the wetland units and distribution of pathogens within the wetland was based on the use of indicator organisms and specifically faecal coliforms as described in standard methods (APHA, 1992). The use of indicator organisms as a tool for assessing the levels of pathogens in a water body is universally accepted since they are easy to monitor and they correlate with populations of pathogenic organisms (Laws, 1981 and Snell *et al.*, 1991). Identification and measurement of all pathogens present in wastewater is very expensive, time consuming and is not justifiable in the circumstances and the environment of this study.

(ii) *Sampling*

Influent samples used in the investigation were all composites taken as the wastewater was loaded into the wetland units. In the second phase, additional samples were drawn from within the open sections of the partially cleared ponds while in the third phase samples were taken from the effluent positions in each of the wetland units in the series. Sterilised glass bottles were used for sample collection throughout the study period.

5.2.1 Vertical profiles of faecal coliforms

The distribution patterns of faecal coliform within the different pond systems and the attendant influence of the systems was determined only in the vegetated wetland units during phase one. It involved taking wastewater samples at the mid-position of each of the vegetated wetland units at different depths measured from the water surface. In the papyrus wetland units (P 1, P 2, P 5 and P 6), samples were taken at the depths of: 10 cm (within the root – rhizome mat zone). At 30 cm (within the water column, below the rooting mat) and at 40 cm (sediment layer, 2–5 cm from the pond bottom). In the *Phragmites* units (P 7 and P 8 with a gravel substratum), water samples were drawn at the depths of 5-10 cm (stems zone) and at 20 – 25 cm (root – substratum zone). Samples were drawn at these various depths by use of a vacuum pump that was connected to a PVC tubing. The tubing was bound to a calibrated metre rule in such a way that it could be adjusted to the required depth.

5.2.2 Physico-chemical measurements

On a given sampling day, simultaneous measurements of electrical conductivity (EC) and pH were carried out at the same depths at which samples for bacterial examination were taken. Electrical conductivity was measured using an EC meter (WTW microprocessor type) which was secured to a calibrated steel rod. The rod could be dipped to the required depths and the EC read off.

5.2.3 Measurement of faecal coliform numbers

A membrane filtration method was used in the determinations. In the laboratory, samples were diluted based on the field data of Kirinya sewage characteristics available at the National Water and Sewerage Corporation (NWSC) central laboratory. Influent samples were diluted about 10,000 times while other samples were diluted between 0 and 100 times using a dilution mixture composed of $MgCl_2$ (190 mg/l) and KH_2PO_4 (42.5 mg/l) buffer (pH 7) in order to prevent cell rupture due to osmotic pressure. The samples were filtered through a cellulose nitrate membrane filter of a pore size of 0.45μm after which the membrane was placed on an absorption pad soaked in 2 ml of liquid lauryl sulphate lactose broth. Samples were incubated in a temperature-controlled incubator, at 44 ± 1 °C for 24 hrs and thereafter, all the characteristically yellow colonies were counted as faecal coliforms. The results were expressed as the number of organisms in 100 ml of the sample examined and by using appropriate dilution ratios, the numbers in the original sample were computed.

5.2.4 Influence of hydraulic retention time

The effect of hydraulic retention time (HRT) on faecal coliform removal was evaluated using the data generated from the different wetland units since they were operated at different retention times. In Phase 1, papyrus systems were run at three retention times: 4, 7 and 12 days with an extension to 14 days in the second phase while retention times of 7, 12 and 14 days were applicable in *Phragmites* systems (Chapter 2).

5.3 Results

5.3.1 Phase 1

Over this operational period, systems were loaded batch wise on a weekly basis and with no vegetation removed from planted ponds.

5.3.1.1 *Faecal coliform population profiles*

The faecal coliform distribution in the different wetland compartments generated from the measurements at the mid-pond positions is illustrated in Fig. 5.1 for papyrus systems. The population distributions measured depicted clear differences which, were based on the defining environmental characteristics within the compartments. The faecal coliform populations in the sediment layers (near the pond bottom) ranged from 2,100 to 11,500 /100ml while in the free water column, i.e. below the root – rhizome mat, the numbers ranged from 5,600 to 30,500 /100 ml. At the rhizome–root mat interface, faecal counts numbered between 3,500 to 19,400 /100 ml. In the *Phragmites* systems only two depths were considered because of the reduced water levels. The faecal coliform population in the root - substratum zone ranged from 300 to 600 /100 ml as compared to 200 to 2,600 /100 ml in the stem zone (Fig.5.2).

The electrical conductivity measured over the same zones were as follows, papyrus: the root–rhizome mat zone, 960–1200 μS/cm, in the water column and near the pond bottom (sediment layer), the ranges were between 1160 – 1198 μS/cm and 1114 – 1134 μS/cm, respectively. In *Phragmites,* at both depths, the electrical conductivity was in the range of 780 ± 18 μS/cm. The pH decreased from an average of 7.2 at the root – rhizome mat to 6.98 near the sediment layers in the two systems. Dissolved oxygen concentrations at all depths were less than 1 mg/l throughout the time of these measurements (Chapter 3).

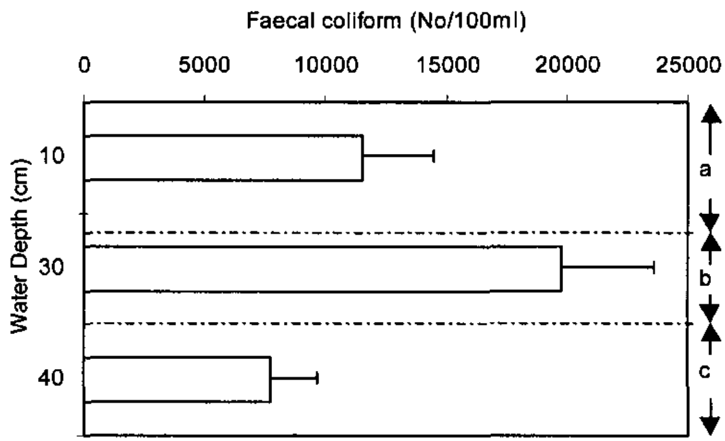

Fig 5.1 Faecal coliform distribution in *Cyperus papyrus* systems (ponds 1 & 2, HRT = 7) at the mid-pond position; horizontal bars indicate standard error of mean (n = 8), a = rhizome–mat zone, b = free water column and c = sediment layer (near pond bottom).

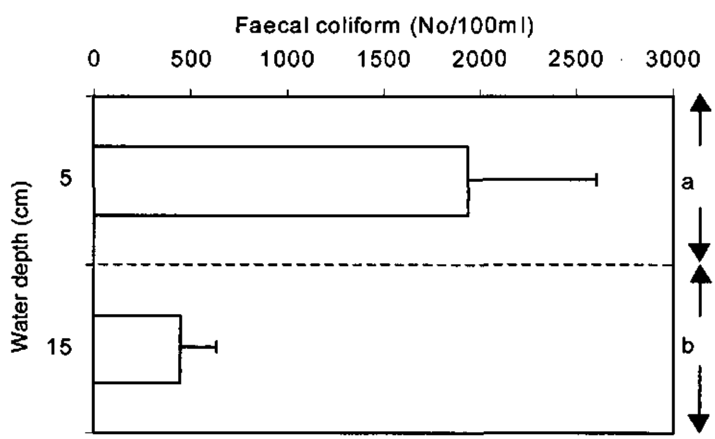

Fig. 5.2 Faecal coliform bacteria distribution in the *Phragmites mauritianus* systems at the mid-pond position (HRT = 7 days), bars represent standard error of mean (n = 6), a = the stem zone, b = root/substratum zone.

5.3.1.2 *Faecal coliform reduction*

The extent of faecal coliform reduction attained in the different pond units varied and significantly so with open pond units. The least (average) effluent faecal counts were recorded in the control open ponds (< 300 FCU/ 100 ml) and the highest in papyrus pond 5 (11,500 FCU/100 ml). The faecal removal in all the units was greater than three log units (i.e. > 99.9 %). However, faecal bacteria removals of above 4 log units (i.e. > 99.99 %) were only obtained in the systems run at low hydraulic loading rates of less than 5 cm/day. (Control ponds, *Phragmites* ponds and papyrus pond 6). A summary of the mean values characterising the different units is given in Table 5.2.

The Volumetric rate constant (k_v) was computed using equation 5.1 and considering influent and effluent faecal counts for each batch.

Table 5.2. Mean values and percent removals of faecal coliform from different ponds measured in Phase 1; mean faecal coliform bacteria population in the influent = 2.92×10^7 / 100 ml and n = 20.

System	Nominal retention time (days)	DO (mg/l)	Ph	Faecal coli. Effluent (No./ 100ml)	Faecal removal (%)	Rate constant, k_v (day^{-1})
C. papyrus P1	7	0.6	6.8	5880	99.95	1.18
C. papyrus P 2	7	0.9	6.9	3630	99.98	1.35
Open (no gravel) P 3	12	11.5	9.8	226	99.99	1.09
Open (gravel) P 4	12	12.3	10.1	197	99.99	1.02
C. papyrus P 5	4	1.1	7.2	11500	99.91	1.91
C. papyrus P 6	12	1.4	7.1	960	99.99	0.89
Phragmites P 7	7	2.3	7.6	1250	99.99	1.57
Phragmites P 8	12	1.5	7.8	830	99.99	0.83

The faecal reduction depicted in Table 5.2 shows a clear linkage between the amount of removed with retention time, pH and dissolved oxygen status of the pond.

5.3.2 Phase 2

In this phase, open zones with a surface area of 10 m^2 were created by complete removal of rooting and standing biomass near the inlet positions of papyrus ponds 1 and 5 and *Phragmites* pond 7. Weekly and daily batch loading was applied in the 1st and 2nd halves of the phase, respectively (Chapter 2).

The faecal coliform removal in this phase was evaluated based on the two loading formats applied in the two halves. Fig.5.3 shows the amounts faecal coliform removed in these two halves in the different ponds. The faecal coliform removed from all vegetated wetland units was greater under the weekly load format in the first half (over 3 log units) than in the daily format (2nd half). This observation is directly linked to the impact of the reduced retention time in the daily feeding regime.

The impact of the open zones created (ponds 1, 5 and 7) in the faecal coliform reduction is also demonstrated in this graph by the consistent elevated (though not statistically significant) removal values than in the corresponding fully vegetated units (ponds 2, 6 and 8). The residual faecal counts in the effluent from all units were less than 2000/100 ml in both halves although faecal counts less than 1000/100 ml, were characteristic of the longer retention times applied in the first half.

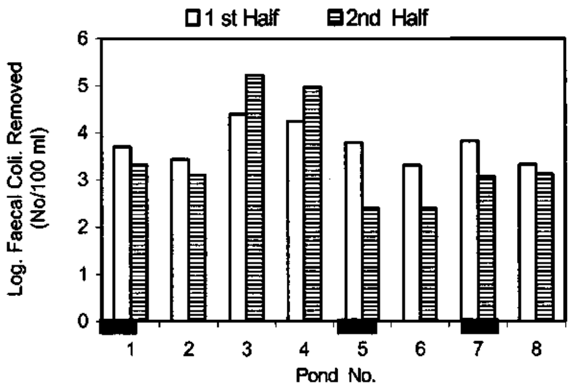

Fig. 5.3 Faecal coliform removed (Log. units) from the wetland units during the two operational halves of phase 2. Retention time of 14 days was applied to papyrus ponds 1 and 2 in 1st half and 9 days in 2nd half. Papyrus ponds 5 and 6 and *Phragmites* ponds had HRT of 7 and 6 days in two halves, respectively. Black boxes show ponds with part of biomass removed.

5.3.3 Phase 3

The total pond length was increased during this phase by combining different wetland units and modifying the configurations to form series S 1-2, S 8-4-7 and S 6-3-5 (Chapter 2).

Faecal coliform bacteria removal derived from the various series during this phase manifested the characteristics of the individual ponds. The faecal coliform counts measured in samples taken at different distances along the three series shows this effect (Figs. 5.4 and 5.5).

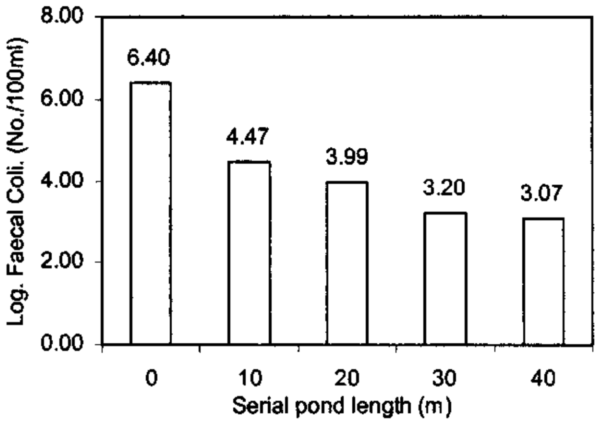

Fig.5.4 Logarithmic faecal coliform population counts along papyrus series, S1-2 comprising of small alternating vegetated and non-vegetated zones.

These graphs show that in all the three series, over 99 % faecal coliform reduction occurred in the first pond in the series. In the pond series with small alternating open and planted zones (S 1-2), a sustained exponential decrease in faecal coliform counts along the series was observed (Fig. 5.4). Meanwhile in the series with the large open pond in the middle (S 6-3-5 & S 8-4-7), there was in increase in the faecal coliform counts in the last vegetated wetland unit (Fig. 5.5).

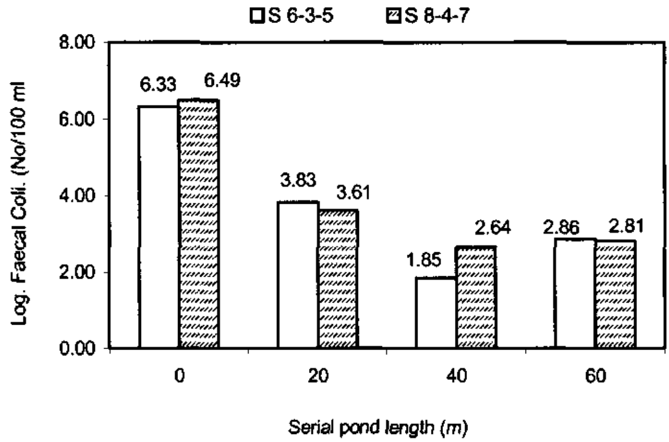

Fig.5.5 Logarithmic faecal coliform population counts in the influent load to the series and effluent at 20 m, 40 m and 60 m serial pond length in each of the three wetland units joined in series S 6-3-5 and S 8-4-7.

Throughout the time of the experiments, the faecal coliform counts in the final effluent from pond in series S 6-3-5 ranged from 380 to 1300/100ml and 100 to 1200/100 ml in series S 8-4-7. The range in series S 1-2 was between 300 to 1900 /100ml. In each of the three series, the average percent faecal coliform removal was above 99.9 %.

5.4 Discussion

The variance in the faecal coliform population distribution within the wetlands as illustrated in Figures. 5.1 and 5.2 demonstrated the role played by the vegetation in their removal. Higher faecal counts were present in the free water zone i.e. below the rhizome–root mat of papyrus as compared to the populations within the rooting mat and near the pond bottom. These observations may be explained by the strong interaction that existed between the flowing wastewater and the rooting mat. This promoted the physical processes, namely entrapment and attachment through which the faecal coliforms are removed. These processes, also identifiable with the removal of suspended solids, together with other subsequent chemical and biological degradation processes are responsible for the reduced faecal coliform numbers within the mat and in the sediment layer. These interactions are either non-existent or minimal in the free water column. Consequently, the possible pathways for pathogen die-off and /or removal were diminished, hence the high faecal populations present or suspended in the water column. A similar explanation holds for the observations made in the faecal coliform population distribution in *Phragmites* systems. From these distribution patterns it can be concluded that in order to obtain increased reductions in faecal bacteria in treatment wetlands, maximum interaction between the rooting medium and wastewater is necessary.

Channelling and free flow below the substrata or roots – rhizome will significantly reduce the performance of the systems.

The results from phase 3 (Figs. 5.4 & 5.5) showed that over 90% of faecal coliform reduction occurred near the inlet positions. These observations correlate with trends that were obtained in BOD and total suspended solids reduction (Chapter 3, Fig 3.5) and findings from temperate wetlands (Brix, 1993; Cooper *et al.*, 1996). The trend obtained suggests that faecal coliform partitioned more in the suspended solids matrix therefore sedimentation, the principle route that accounted for suspended solids removal was also a significant removal pathway for the faecal coliform bacteria. Sedimentation and entrapment are enhanced by a longer retention time and consequently as illustrated in Table 5.2, pond operated at a longer HRT resulted into a high faecal removal. The influence of type and structure of the rooting medium on settlement achievable and hence the faecal bacterial populations that get removed was demonstrated in this study. Faecal coliform removal in the floating papyrus units was similar to that in *Phragmites* units and yet the latter had a substratum base. Therefore the contribution of the massive root mat in the faecal removal in papyrus was significant.

The lethal effect of sunlight on the indicator coliform organisms is mentioned in previous studies (Gersberg *et al.*, 1987). Secondary effects driven by solar energy such as high pH > 9, is reported to be lethal to even viruses (Faechem *et al.*, 1983). This photolytic induced faecal coliform removal was considered significant at the Kirinya pilot constructed wetlands given the location at the equator and with a daily average sunlight duration of 11 hours. The higher volumetric rate constant values computed (Table 5.2) as compared to rates obtained in temperate zones (Bavor *et al.*, 1987 and Gearheart *et al.*, 1989) supports this fact. Faecal coliform reduction was particularly high in control ponds and in partially vegetated ponds that were exposed to direct sunlight (Fig. 5.3). The maximum pH recorded in the open ponds and zones was above 10 with high dissolved oxygen concentrations (Figs. 3.1 and 3.2, Chapter 3). A combination of these two environmental parameters created conditions that do not favour faecal coliform survival and consequently, contributed to the observed reductions in these ponds. A conclusion derived from these results is that including open zones in the wetland increases the efficiencies of faecal coliform removal.

In phase 1, significant reduction of faecal coliforms was attained in papyrus and *Phragmites* units despite the anoxic conditions and rather neutral pH which has minimal toxicity to bacteria. Apart from sedimentation and entrapment mentioned earlier on, it is suggested that other documented faecal bacterial reduction routes such as predation by higher organisms in the food chain and biocidal effects derived from roots/rhizome may have also contributed to their removal. Visual inspection of the water in the ponds showed very diverse communities present. However, the extent and actual contribution by each of the pathways was not quantified in this study.

The significant influence of retention time on faecal coliform removal, which was shown in the studies of Williams *et al* (1994) and Schreijer *et al* (1997), was reflected in the results from this study as well. Highest faecal coliform removal greater than 4 log units were only attained at retention time (HRT) above 7 days in both *Phragmites* and papyrus systems (Table 5.2). The faecal coliform reduction obtained at the shorter HRTs' of 4 days were nevertheless still high (> 3 log units) and having effluent faecal bacteria counts of less than 10,000/100ml which is within the Uganda National Standards for wastewater discharge into water or land (NEMA, 1999). For the large-scale use of constructed wetlands, retention times above 7 days will imply large land areas and hence increased costs. Therefore retention times between 4 to 7 days are considered suitable for systems in the environments such the Kirinya one. Optimal reductions of the bulk pollutants; biochemical oxygen demand, BOD and suspended solids, TSS were obtained during these operational phases (Chapter 3) at these shorter retention times.

In the alternating open water and planted zones applied in operational phase 3, high faecal coliform removals (> 99.9%) were determined in all the three series. Such elevated faecal coliform removals in similar alternating arrangement are reported by Bavor *et al* (1987) in studies carried out in sub-tropical environments in Australia. In this study, the increase in faecal counts noted in series S 6-3-5 and S 8-4-7 after the open pond can be associated with the extra sources of faecal coliform bacteria within the wetland, which included birds and other wild life. The big monitor lizards and weaverbirds particularly frequented the papyrus wetland units may explain the bigger increases in faecal bacteria observed. The exponential faecal coliform population decreases in series S 1-2 suggests a minimal external input from within the wetland because of the small vegetated sections of wetland.

Overall, the faecal coliform removal efficiency obtained in the various phases and systems during this study (>99.9%) were only comparable to that reported in activated sludge systems and stabilisation pond systems (Mara, 1992; Rowe and Abdel-Magid, 1995). The wetland systems however, have the advantage of the low investment and operating cost as compared to the activated sludge systems. Waste stabilisation ponds on the other hand have an inherent problem of producing secondary TSS and BOD and oxidised forms of nitrogen, which are not desired products of the treatment. In the wetland configuration adopted in the combined series S 1-2, S 6-3-5 and S 8-4-7, simultaneous and consistent high removal of faecal coliform (>99.9), BOD (> 80%) and TSS (>74 %) as well as for ammonium N (> 85%) (Chapters 3 and 4) was achieved. It is therefore concluded that these types of wetland configurations are the most suitable for large-scale application in environments similar to that at Kirinya, Jinja.

The rate at which bacterial populations get reduced in wetlands is shown by the value of the decay rate constant. Area-based rate constant, k_T, ranging from 0.014 – 0.378 m/day with a mean value of 0.3 m/day are reported by Kadlec and Knight (1996) in some North American treatment wetlands. Volumetric rate constant, k_V, in the range of 0.29 - 0.86 day^{-1} is reported from separate studies in Australia (Bavor *et al*, 1987) and in the USA (Gersberg *et al.*, 1987). In this study, the faecal coliform volumetric rate constants (k_V) calculated using equation 5.1 for each individual pond (Table 5.2), were comparatively greater than reported elsewhere. This indicated that the Kirinya constructed wetlands provided a suitable environment for rapid faecal coliform removal. It is evident that the faecal decay rates were consistently greater in *Cyperus papyrus* systems than in *Phragmites* or even the open ponds which is a manifestation of the effect of the massive root–rhizome mat in the former systems. It is concluded that *Cyperus papyrus* systems have an advantage over the *Phragmites* ones in regard to faecal coliform removal.

5.5 Conclusions

A faecal coliform removal efficiency above 3 log units (99.9%) was obtained at the Kirinya pilot constructed wetlands during all the operational phases. The faecal counts in the effluent from the different wetland units and series was less than 10,000/100 ml which, is the Uganda National Standard for discharge of wastewater into water or land. It is concluded that the systems are effective and can be used for large-scale applications. *Cyperus papyrus* plant-based systems have an advantage because of the massive rooting plant biomass with a large surface area that is an essential requirement for elevated faecal coliform removal. The retention time of between 4- 6 days was found to be optimal for faecal coliform removal in all pond configurations. Faecal coliform counts in the effluent were least under the alternating vegetated and non-vegetated serial configuration. Since other pollutants evaluated (BOD, TSS and nutrients) were optimally removed as well, it is recommended as the best option for use in such environments.

Finally, since this study was based on only faecal coliforms, additional studies aimed at quantifying other pathogens is strongly recommended for further justification of the constructed wetland technology application in the warmer regions.

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Chapter 6

Performance Evaluation of a Single Household Constructed Wetland

Abstract 127

6.1 Introduction 127

6.1.1 Small household constructed wetlands 128
6.1.2 Scope of study 129

6. 2 Study Area 129

6.2.1 Site Location 130
6.2.2 Design Criteria and lay out 130
6.2.3 Start up and operation 131

6.3 Construction and Operating Costs 131

6.4 Materials and Methods 131
6.4.1 Measurements 131
6.4.2 Systems management 133
6.4.3 Structured interviews 133

6.5 Results 133

6.5.1 Physical parameters 133
6.5.2 Removal of pollutants 134
6.5.3 Plant growth and nutrient distribution 137
6.5.4 Interview results 137

6.6 Discussion 138

6.7 Conclusions 141

References 141

Abstract

The viability of using a small constructed wetland to treat wastewater from a household in a tropical environment was evaluated in this study, using two parallel wetland systems located within a residential area in Jinja Municipality, Uganda. Biochemical oxygen demand (BOD), chemical oxygen demand (COD) and total suspended solids (TSS) were effectively removed by over 65 % and to concentration levels that meet the Uganda wastewater discharge regulations. 24% of phosphorus and 38% ammonium were removed by the systems but with effluent concentrations above the regulated values. Faecal coliform reduction of only of 1.13 log units was achieved. Nuisance and public health risks from mosquitoes breeding in the wetland proved to be the critical issue feared by the users of the facility.

6.1 Introduction

The treatment and disposal of wastewater in any convenient way is one of the essential requirements for a healthy and productive life at both individual and community levels. In the industrialised countries conventional sewage systems are now the standard and the most convenient sanitary way for disposal and treatment of wastewater. In many developing nations, limited financial resources make this option not available for the majority of the people. Despite this limitation conventional sewage treatment is still the most sought option by the planners and decision-makers in these countries. The convenience of the technology is probably the main reason for this bias and the general lack of interest in alternative sanitation systems which in fact may offer health solutions to the majority of the people given the financial, cultural practices and infrastructure available. During the International Drinking Water Supply and Sanitation Decade (1980-1990) (Feachem *et al.*, 1983), efforts were made to popularise the use of other alternative wastewater management technologies at household and community levels. Constructed wetland technology was still new and was then not mentioned as one of the alternatives.

The common types of household sanitation systems applied to differing extents in different countries include pit latrines, pour-flush toilets, composting toilets, aquaprivies and septic tanks (Kalbermatten *et al.*, 1982). These systems are particularly attractive because of the low cost in setting them up within the homesteads. However, their application is limited to rural and low-density urban areas. All of these systems are potential non-point pollution sources for groundwater and surface water. This is particularly the case when they are built in areas with poor soil percolation rates, high water table or karst topography or in heavy rainfall regions (Steiner *et al.*, 1993). Poor designs and construction may worsen the situation. Even when these systems meet the required standards, seepage from them can still reach the groundwater aquifers and pose a problem to public health and other legitimate water uses (Lewis, *et al.*, 1982). Poorly built pit latrines were highlighted by Kiyonga (1998) as some of the major contributory factors to the malaria and cholera epidemic of 1998 in Uganda.

Cheaper, easier to operate and maintain alternatives are required to handle wastewater at household level. Use of small household-scale constructed wetlands to treat domestic sewage is one such option investigated in the United States (Steiner and Combs, 1993) and the United Kingdom (Green and Upton, 1993). The basis for applying small wetlands stems from research on the use of constructed wetlands in wastewater treatment (Reddy and Smith, 1987; Hammer, 1989; Cooper and Findlater, 1990; Moshiri, 1993; Haberl *et al.*, 1996; Vymazal *et*

al., 1998). Constructed wetlands can attain a high degree of pollutant removal that meet the regulatory standards.
During this study at Kirinya in Jinja, the capacity of a tropical constructed wetland to remove organic and suspended matter, nutrients and pathogens from wastewater was demonstrated (Chapters 2 to 5). The removal efficiency of biochemical oxygen demand (BOD) and total suspended solids (TSS) was greater than 70 % with residual effluent concentrations below the regulatory 50 mg/l. Nutrients (nitrogen and phosphorus) removal was variable. Ammonium reduction varied from 25 % to 90 % under different system configurations while phosphorus reduction remained below 50 %. Faecal coliforms (FC) reduction was high with removals of over 99.9 % during all the different operational phases.

6.1.1 Small household constructed wetlands

The initial performance data of household-scale constructed wetlands was generated from the research and demonstration work by the Tennessee Valley Authority in the United States (Choate *et al.*, 1993; Steiner *et al.*, 1993; Steiner and Combs, 1993; Terry, 1993). Designs incorporated two units connected in series with the second cell at a lower elevation than the first one. Parallel unit arrangement has also been applied and these have the advantage that they can be operated with more flexibility. Subsurface flow wetlands are the common type of wetlands that have been built. Results obtained show a marked improvement in the quality of effluent from septic tanks after flowing through the wetland. A summary of the performance from different small wetland systems operational in the Tennessee Valley is outlined in Table 6.1.

The cost of household wetland units built in the USA was variable and ranged from US $ 2,000 to US $ 9,000 (Steiner and Combs, 1993). The cost was dependent on the hydraulic loading, type of wetland system (subsurface wetlands were more expensive than surface flow types), the location and the level of participation of the user. Typical costs for the three systems reviewed are also given in Table 6.1

Constraints experienced in the application of this technology in the United States and the United Kingdom emanate from both the engineering and social aspects. Land requirement coupled with design and construction errors arising from over or underestimation have been mentioned and these are reflected in the costs. Often builders have no prior experience with the size of the systems required. Other derived problems including odour, breeding by mosquitoes and other nuisance vermin, have been cited. These secondary effects are of more direct concern to the users in view of the fact that most of these systems are built in the vicinity of homes. An information gap between the potential users and designers was identified as a constraint since the advantages of the systems were still not well understood. Lack of incentives such as relief on discharge costs was also listed as a possible de-motivator for those willing to use the technology.

Table 6.1. Typical performance obtained from small household constructed wetlands in the USA. (Compiled from Steiner and Combs, 1993)

Location	Hydraulic load (m3/day)	Cost (US $)	Parameter	Influent (Septic Tank effluent)	Effluent	Percent removal
Single Mountain	2.27	2,000	BOD	247	27	89
			TSS	60	13	78
			Faecal coli.	280,000	61,000	78
Chattanooga	4.95	8,000	BOD	44	12	73
			TSS	130	6	95
			Faecal coli.	725,000	5800	99
Kentucky	1.36	3,300	BOD	172	31	82
			TSS	59	16	73
			Faecal coli.			99.9

Key: BOD and TSS units = (mg/l), faecal coliform (FC) = No./100ml.

6.1.2 Scope of study

The potential of small constructed wetlands in reducing non-point source pollution in homes especially in warmer areas seems to exist regardless of the present shortcomings experienced in the temperate regions. Data on their performance capacity and social consideration is essential before they can be popularised in these regions. At the moment, there is no evidence from the literature of any research work or investigation that has been carried out using this type of systems in any developing country and more so in sub-Saharan Africa. This study was aimed at demonstrating the technology in a readily accessible area and assessing the viability of using the systems in a typical Ugandan household. The promising findings from the parallel studies on the relatively bigger Kirinya pilot wetlands mentioned earlier provided the impetus for the investigation.

The specific objectives of this study were:

(i) To determine the potential of a household-scale constructed wetlands in treating domestic wastewater in tropical environments;

(ii) To evaluate the advantages and disadvantages of the system;

(iii) To assess ways of improving the design and operation criteria of the system;

(iv) To raise awareness of the technology among the local population and administrators.

Cyperus papyrus, one of the plants used at the Kirinya pilot wetlands was selected as a candidate plant mainly because of its rapid growth rate, massive rooting media and the relative better performance it had over *Phragmites*. Besides, the provisional requirement of a solid support medium like gravel for the *Phragmites* plants made them a more costly option in the circumstances.

6. 2 Study Area

The establishment of this small wetland at the selected site was made possible through the initial contacts with the Rotary Club of Jinja. Mr. Sonko Kiwanuka who was the then Officer In-Charge of National Water and Sewerage Corporation, Jinja Area and a Rotarian, made the initial request. RIZA (The Institute for Inland Water Management and Wastewater Treatment) of The Netherlands provided the funding for construction.

6.2.1 Site Location

The small constructed wetland is located about 80 km East of Kampala at $00^0 25'$N; $33^0 12'$E, the residential house of Dr. Fred Mukwenda, plot No 196, Nile Avenue in Jinja municipality, Uganda. The house overlooks the source of the River Nile at Ripon falls. It is nearly 2 km to the west of the Kirinya pilot constructed site. The plot size is 0.75 hectares and the wetland is sited 25 m from the main house and 6 m from the adjacent northern boundary fence (Plate 1).

Plate 1. The site of the household constructed wetland at Jinja Municipality

6.2.2 Design Criteria and lay out

The wetland was designed with the following assumptions and conditions:
1. The main house and the servants quarters are occupied by a maximum of 20 persons
2. Wastewater is pre-treated by a septic tank up to a 40 % BOD reduction
3. Hydraulic load system equivalent to 2 m^3/day (each unit 1 m^3/day)
4. Maximum hydraulic loading rate of 10 cm/day
5. Maximum influent BOD concentration of 180 mg/l
6. Effluent BOD concentration of 30 mg/l.

The wetland built comprised two equal parallel units each 10m long, 1 m wide and 1 m deep with a slope of 1%. A distributor chamber was in-built at the inlet such that equal wastewater volumes flowed into the two channels. The outflow pipe was fitted such that a constant water level of 0.8 m was maintained in each of the systems. The walls of the wetland were made from concrete blocks with floors of 150mm concrete slabs with sheeting. The effluent from the ponds flowed into a soak away pit of medium depth. Details are given in Figs. 6.1 and 6.2.

6.2.3 Start up and operation

The investigations in this study were carried out between October 1997 and March 1999.

Transplanting and equilibration criteria used in this small constructed wetlands was similar to those applied at Kirinya pilot constructed wetlands (Chapter 2). The wetland channels were filled with water drawn from a facultative pond at Kirinya Sewage Works to seed the systems. *Cyperus papyrus* propagules were obtained from the cultivated sections of the natural swamp. The first lots of plants were destroyed by the exceptionally strong winds after 6 months. New plants were re-established after removing broken shoots and dead rhizomes from the wetland units in September 1998.

Rainfall and other climatic variables were assumed to be similar to those measured at the Kirinya wetland site (chapter 2). Due to the plumbing problems in the main house and in the servants quarters, an external standpipe had to be fixed in the yard and most of outflow from it including the laundry was diverted to the septic tank.

6.3 Construction and Operating Costs

The household wetland was built at a cost of US $ 6,000. Materials for the concrete works, pipes, transportation, labour and the pre-treatment septic tank contributed to this total cost. Concrete works alone was worth 40% of this cost. Operational costs covered the water bills for the first year (this was the precondition given by the landlord for establishing the system in the premises), pruning, desludging and spraying with the public health insecticide. For the whole experimental period (2years), it amounted to US $ 1,000.

6.4 Materials and Methods

The methodology applied was similar to that described for the Kirinya systems (Chapter 2). Sampling sites were set up at 2 m intervals from the influent chamber in each of the channels. The analytical methods described in Standard Methods (APHA, 1992) and the HACH DR 2010 and 2000 spectrophotometers handbooks were followed in all cases. Brief descriptions of procedures for determination of the various parameters are outlined in Chapter 2.

6.4.1 Measurements

The experimental work was carried out in the two planting cycles that arose because of the wind effects. The first cycle covered the period October 1997 to April 1998 and the second one from October 1997 to March 1999. The following key pollution index variables were investigated at defined intervals. Dissolved oxygen, temperature, electrical conductivity and pH, COD, BOD, TSS, ammonium, total nitrogen, ortho and total phosphate, total and faecal coliforms.

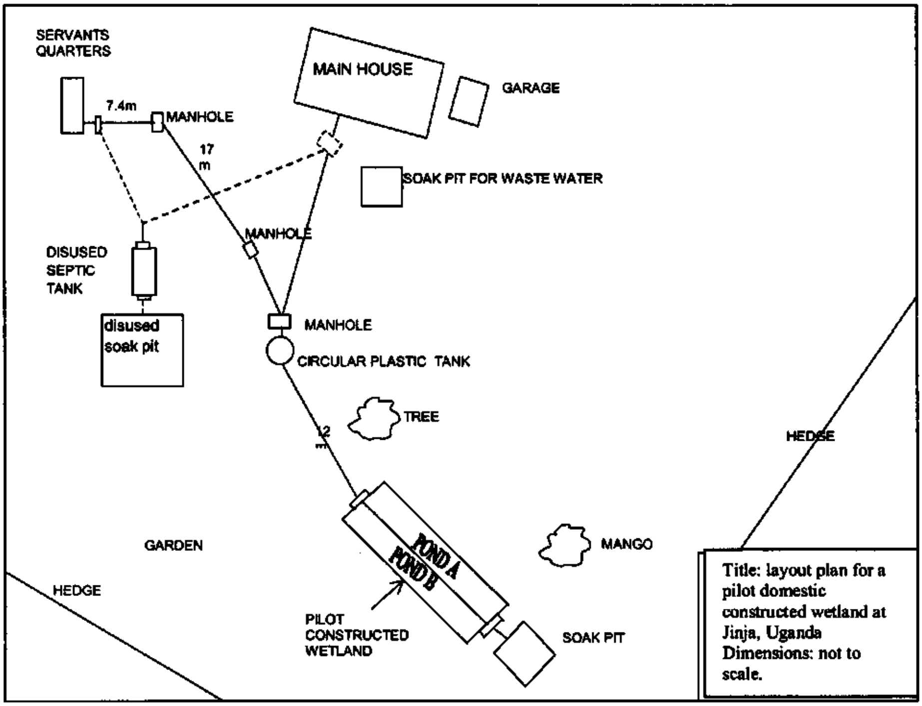


Fig. 6.1 Lay out of the residential house and the constructed wetland and the flow network at Nile Avenue, Plot. 196 Jinja Municipality.

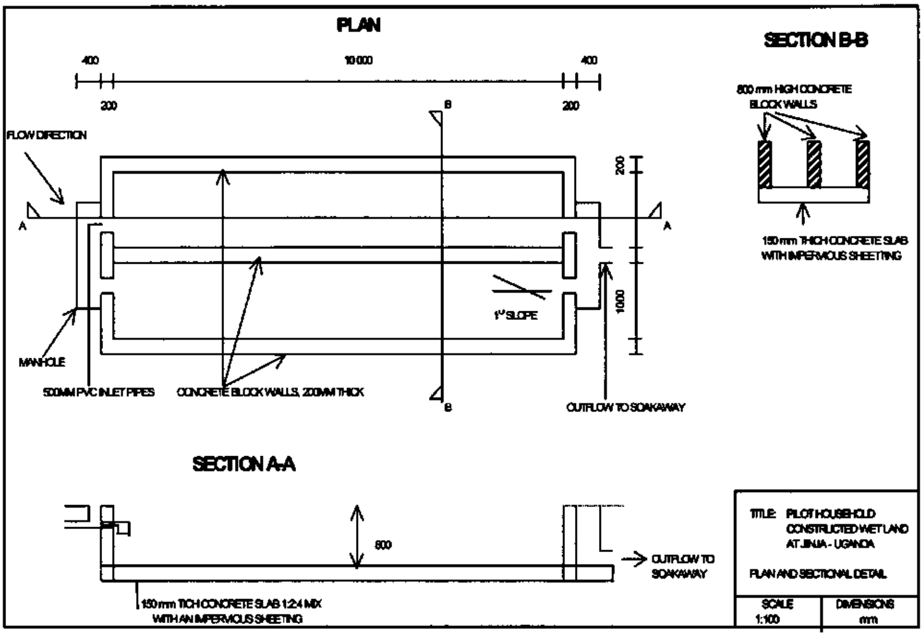


Fig. 6.2 Detailed drawing showing the plan and cross sectional details of the small household-scale constructed wetland.

Plant biomass changes and nutrient distribution in the different plant parts was measured as described in chapter 4. The water consumption in the household was read off from a water meter that was installed in the second planting cycle. It served both the main house and the servants quarters. Daily wastewater hydraulic loading into the wetlands was estimated as a percentage of the water received in the premises. Effluent volumes from the wetlands were estimated using a timed bucket collection of the outflow volumes from each of the units.

6.4.2 Systems management

A constant maintenance programme was put in place throughout the experimental period and included:

(i) Regular spraying of the units with anti mosquito repellent (FENDDONA, 6% with active ingredient of 60 g/l Aplhacypermethrin) to avert mosquito nuisance and public health risk to the residents and neighbours.
(ii) Desludging the septic tank once over the time of the experiment.
(iii) Fencing the systems to minimise the risk to children and to bar trespassers.

6.4.3 Structured interviews

A survey by an oral interview and a questionnaire method were used to gauge the opinion of the users (from the main house and the servant quarters) and a neighbour living in an adjacent house, about the wetland technology. The persons interviewed (5 adults) were an informed group who had witnessed the construction and operation of the small wetlands. In addition, prior to setting up of the wetland unit, the respondents were briefed on basics on the wetland technology in wastewater treatment. The owner of the premise was keen on the bacteriological quality of the systems, he was provided with a regular update of the faecal coliform results determined. The oral interview was carried towards the end of the investigations. The questions asked sought the following:

(i) previous knowledge about the use of wetlands to treat wastewater
(ii) the observable advantages and disadvantages of the system.
(iii) the possibility of recommending the use of the technology to others
(iv) any improvements viewed as essential for the performance of the systems
(v) general comments.

6.5 Results

The septic tank effluent (wetland influent) volumes entering the wetlands were very variable in time and quantity and depended on the day of the week. Higher flows were received usually during weekends. Computed values from the water meter readings taken in the 2nd planting cycle ranged from 0.184 to 0.9 m^3/day while effluent volumes varied between 0.083 to 0.3 m^3/day.

6.5.1 Physical parameters

Throughout the time of experimentation, channel water temperatures averaged to 22 ± 0.5 °C while dissolved oxygen concentrations never exceeded 0.5 mg/l in both units. The pH ranged between 6.90 and 7.85 and the electrical conductivity decreased from 1.4 ± 0.1 mS/cm in the influent to 1.29 ± 0.09 mS/cm in the effluents.

6.5.2 Removal of pollutants

The results of systems performance presented cover all the measurements derived from the two planting cycles. Transect and the weekly results represent measurements only in the second cycle.

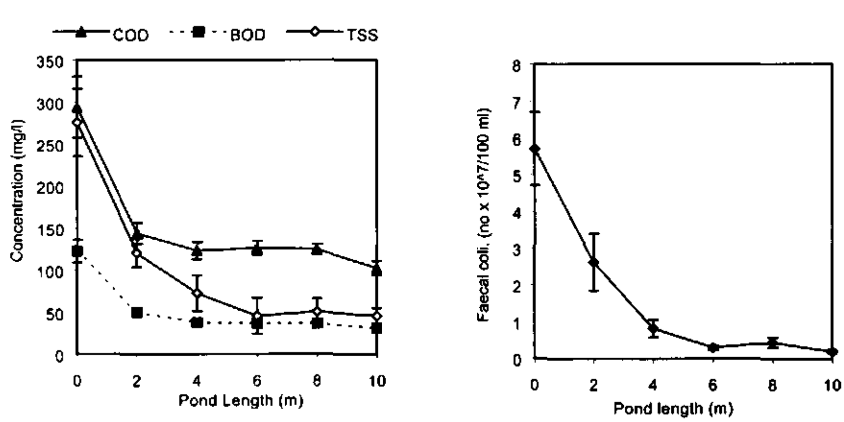
(i) *Organic and suspended matter, and coliform bacteria*

The average concentrations of COD, BOD, TSS, faecal and total coliforms in the septic tank effluent (influent to the wetland units) and the effluent from the two demonstration wetland units are given in table 6.2. The percent reduction calculated on the basis of concentration for each parameter is also indicated.

Table 6.2. Mean Influent and effluent data of bulk pollutants at the demonstration household-scale wetland, ± standard error of the mean, n = 48.

Parameter	Septic tank Effluent	Wetland Effluent	Removal efficiency (%)
TSS (mg/l)	275.5 ± 40.4	45.8 ± 9.2	83.4
BOD "	122.7 ± 13.7	31.2 ± 2.5	74.6
COD "	293.8 ± 36.4	102.4 ± 8.7	65.2
Total coli. Log(No./100ml)	8.573	7.238	96.6
Faecal coli. (")	7.756	6.283	94.9

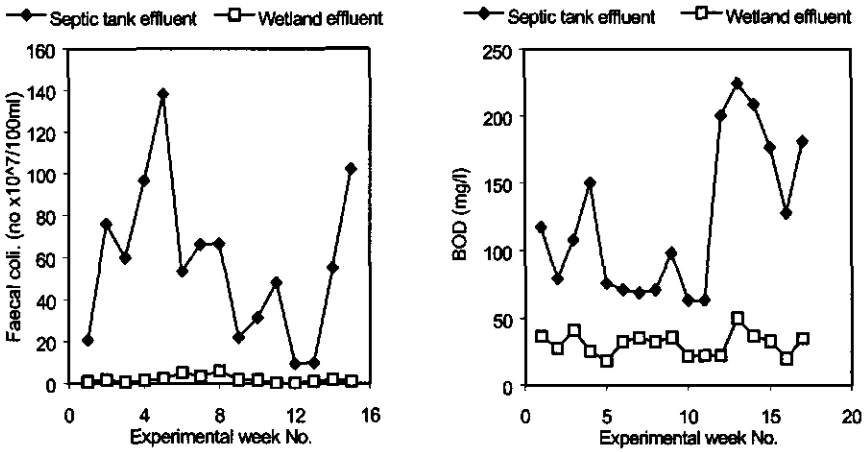
The concentration profiles of TSS, BOD, total coliforms and faecal coliforms along the length of the pond showed that much of their removal took place within the first 10 – 20 % of channel length, i.e. near the inlet (Fig. 6. 3). No significant reductions occurred thereafter. This is a typical trend observed at Kirinya pilot constructed wetlands and other wetland systems referenced in literature.



(a) (b)

Fig. 6.3 The progression of (a) COD, BOD and TSS concentration and (b) faecal coliform population along the longitudinal length of the household constructed wetland (Oct. 98 to Mar. 99).

The concentration of TSS, COD, BOD and faecal coliform in the septic tank effluent was variable throughout the time of the study and probably reflected the activities taking place in the houses. However, the concentrations of these parameters remained steadily lower than influent regardless of the fluctuations in the septic tank effluent. These trends are shown in Fig. 6.4 for BOD and faecal coliforms.



(a) (b)

Fig. 6.4 Concentration of (a) faecal coliforms and (b) BOD in the influent (septic tank effluent) and effluent from the household wetland (Nov. 98 to Mar. 99).

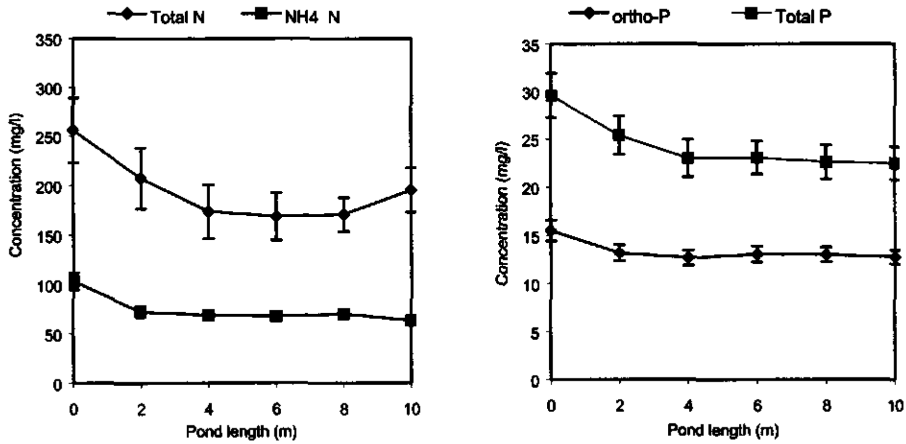
(ii) *Nitrogen and phosphorus*

Summary data of concentrations of ammonium nitrogen, total nitrogen, ortho- and total phosphate in the septic tank effluent and wetland effluents measured during the study is given in Table 6.3. All the wetland effluent values exceeded those obtained from Kirinya pilot wetland.

Table 6.3. Nitrogen and phosphorous compounds concentration in the septic tank effluent and wetland effluent (mean, n = 40).

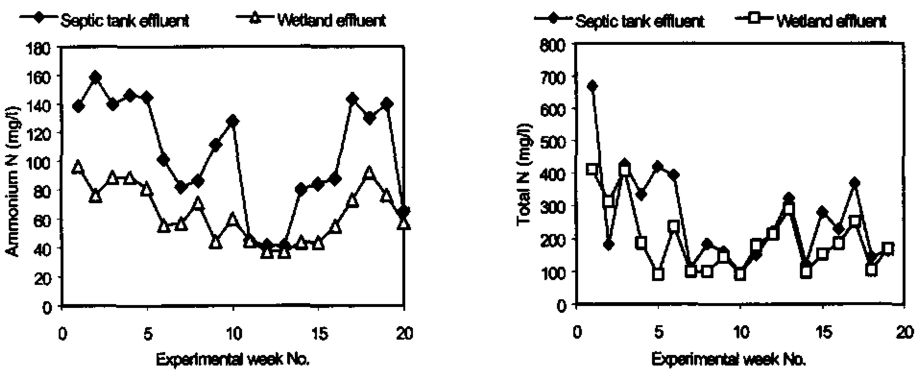
Parameter	Septic tank effluent (mg/l)	Wetland effluent (mg/l)	Removal efficiency (%)
Ammonium N	103.2 ± 8.5	63.4 ± 4.2	38.6
Total N	256 ± 33.2	195.2 ± 22.5	24.1
Ortho-phosphate	15.5 ± 1.1	12.7 ± 0.7	21.7
Total phosphate	29.6 ± 2.3	22.4 ± 1.7	23.8

The longitudinal concentration gradients for the same parameters are illustrated in Fig. 6.5 where it is noted that in all cases, the small reduction observed took place within the first half of the wetland only. The wetland effluent concentrations of the nutrients measured were responsive to the influent concentration fluctuations unlike the bulk pollutants (Figs. 6.6 and 6.7). The concentrations of phosphorus (total and ortho-phosphate) and ammonium in the septic tank effluent were similar to that obtained from raw sewage which was indicating the minimal performance of the septic tank. This trend indicated the low ability and limitation of the small wetland to reduce these nutrient pollutants.



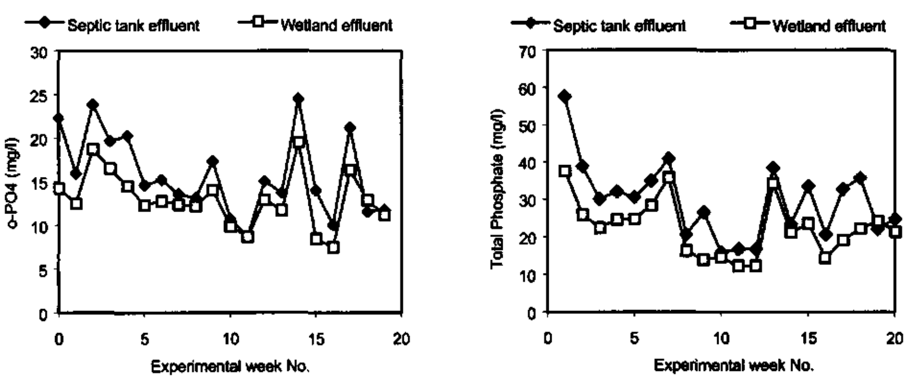
(a) (b)

Fig. 6.5 Longitudinal concentration profiles of (a) nitrogen and (b) phosphorus compounds in the wetland units.



(a) (b)

Fig. 6.6 Concentration of (a) ammonium and (b) total nitrogen in the influent (septic tank effluent) and effluent from the household wetland (Nov. 98 to Mar. 99).



(a) (b)

Fig. 6.7 Concentration of (a) ortho-phosphate and (b) total phosphate in the influent (septic tank effluent) and effluent from a household wetland (Nov 98 to Mar 99).

6.5.3 Plant growth and nutrient distribution

The growth and the developments of the *Cyperus papyrus* plants in this small wetland were followed over two cycles during the study period. Strong winds destroyed the first lot 6 months after planting.

(i) *Biomass*

In both planting cycles, plants were uniformly distributed in the wetland units like in Kirinya pilot wetlands. Maximum shoot heights of 2.52 m were recorded in the first cycle and 2.25m in the second cycle. An overall pond average of 0.55 m was obtained for the entire growth phase of the experiment. Similarly, maximum shoot density in the first and second cycles were 70 and 56 shoots /m^2 respectively. Average standing biomass yield (dry weight, DW) of 2.35 kg DW/m^2 was determined during the second cycle with much of the contribution generated by the young shoots that were continuously sprouting. Growth rates measured over two month intervals during the exponential growth stage were 302 kg DW ha^{-1} day^{-1}. Rooting biomass was not quantified but physical inspection showed that the floating rooting mat was not yet completely established. This was probably the reason why the plants were continuously susceptible to wind destruction and did not grow up to the shoot heights measured in Kirinya pilot wetlands.

(ii) *Nutrient uptake*

The content of nitrogen and phosphorus in the standing and rooting biomass measured in the second planting cycle is given in Table 6.4. The values are comparable to those obtained in the bigger Kirinya pilot wetlands

Table 6.4. Nutrient distribution in the different parts of the *C. papyrus* plants in the small constructed wetland.

Plant Part	Total Nitrogen, TN (mg/g)	Total Phosphate, TP (mg/g)
Umbel	21.91 ± 1.51	5.95 ± 0.43
Culm	14.95 ± 2.21	5.61 ± 0.59
Rhizome	19.96 ± 2.60	7.90 ± 0.83
Roots	22.16 ± 1.39	10.05 ± 0.85

6.5.4 Interview results

Although the sample size of the survey carried out was small, their response provided very useful and original insight to the way the wetland technology is understood and perceived at the time in Uganda. The specific responses to the questions posed are outlined in the next paragraphs.

(i) *Wastewater treatment by wetlands*

All the respondents indicated that up to the time of this study, they had no prior knowledge on the possibilities of using wetlands to treat wastewater directly let alone the use of household scale wetlands as an alternative on-site sanitation method. 80% of the respondents expressed scepticism about the viability and social acceptability of household wetlands in Uganda.

(ii) *Perceived advantages and disadvantages*

Although all the respondents agreed on the sanitation implication of the technology, they expressed more concerns on the immediate health risks and running costs of the small systems compared to other on-site sanitation systems familiar to them. Among the disadvantages mentioned, four were common to all and included:

(i) Mosquito breeding and risk of malaria infection.
(ii) A high risk of exposure to pathogenic bacteria especially for children playing within the compound.
(iii) Extra labour cost of maintaining it as compared to the stand-alone septic tanks. (observed from the harvesting of broken shoots)
(iv) The rather big land area it occupied and the life span of the plants.

Of all these, the risk of malaria infection was identified by all the respondents as the limiting factor and suggested that to be a factor to be considered in any future works. However, all the respondents acknowledged the aesthetic value the plants gave to the home and they even suggested use of more attractive flowering plants.

(iii) *General comments*

The general comments raised by all were related to the following concerns:

(i) the suitability of the water flowing out for use by the owner of the house for vegetable growing especially in the dry seasons
(ii) the possibility of using other plants that can resist wind unlike the papyrus that readily broke when winds are strong
(iii) the safety of eating the mangoes from the tree adjacent to the wetland
(iv) how we (the researchers) felt working with such potentially high risk systems
(v) how local community members could be trained and participate in the work
(vi) how much it would cost without the concrete work
(vii) what was wrong with septic tanks and other on-site sanitation systems.

6.6 Discussion

The treatment performance of this small household wetland was greatly influenced by the variable concentrations of the septic tank effluent loaded. Although there was no comparative data on septic tank performance in Uganda, available data from the household wetland studies in the USA indicated that the effluent from Jinja had very high amounts of faecal and total coliforms, nitrogen and phosphorus which were in the same order as raw sewage. TSS was more by over 100 % and as a consequence, accumulation of suspended solids near the inlet was more rapid than was observed at the Kirinya pilot wetlands. The sludge at this point had to be removed after only one years' operation, which was not a desired activity.

The oxygen concentrations, pH and trends for TSS, COD, BOD and faecal and total coliform (Fig. 6.3) were characteristically similar to what was obtained at the Kirinya pilot wetlands. This suggests that the main controlling mechanisms of sedimentation, aerobic and anaerobic degradation processes as outlined in chapter 3 were responsible for the reduction derived in this small system as well. The overall removal of TSS, BOD and COD in this wetland unit was very high despite the shorter pond length and the high influent concentrations as compared to the Kirinya systems. The treatment performance was also comparable to what was obtained in some household wetlands in the USA (Table 6.1). The effluent concentrations were within the range of Uganda National Standards (NEMA, 1999) regulations of 50 mg/l for BOD and 100 mg/l for COD and TSS.

A modest reduction of faecal populations (by 1.13 log units) was realised leaving very high faecal numbers in the effluent (6.2 log units). The reduced performance in faecal removal was linked to the high faecal counts in the septic tank effluent. Apparently an insignificant removal was realised in the septic tanks (effluent = 7.8 log units, typical raw range). Influent concentration of pathogens is one factor that determines the degree of their removal in the wetland. In addition, this dismal performance was probably also due to the decrease of retention time caused by water flow short-circuiting which could be seen when soap washings entered the wetland units. The residual faecal populations in the effluent from the wetland units represented a public health risk. Re-use of this effluent for garden irrigation as initially envisaged could not be recommended for this reason. Further re-evaluation of this wetland capacity in reducing faecal populations will only be useful after the septic tank is stabilised or replaced.

The extent of ammonium and total nitrogen removal by the system was low and even then most of it was registered near the inlet (Fig. 6.5a) which may suggest that adsorption to solids that settled out rapidly in these zones influenced the reduction. This reduced impact of the wetland on nitrogen removal was characteristic for the whole experimental period (Figs. 6.6a & b). The influent ammonium concentrations (~ 100 mg/l) even neglecting the organic nitrogen which, from previous studies, is known to be high in septic tank effluents (Reed *et al.*, 1995), imposed an oxygen demand of over 400 mg/l. Based on the observed residual ammonium N and oxygen concentrations and BOD removed, it is evident that the oxygen that drives the nitrification–denitirification pathways for ammonium removal was inadequate and therefore reducing the removal capacity. Further, from the tables of equilibrium concentration relationships of aqueous ammonia/ammonium N at different pH and temperatures (Emerson *et al.*, 1975), the percent of ammonia (NH_3) in aqueous medium based on the prevailing pH (7.5) and temperature (22 ^{0}C) was only 1.43 %. This suggests that reduction of ammonium nitrogen by volatilisation route was negligible.

The limitation of ammonium removal in this household wetland was similar to the findings at Kirinya pilot wetlands during the first operational phase (chapter 4). Choate *et al* (1993) and Green and Upton (1993) also found reduced ammonium removal in their studies of similar small household systems. It is concluded that this system in the present design configuration and loading may not produce effluent with ammonium concentration meeting the discharge standards. Although the nitrogen enriched could be of value if it were to be reused for gardening, the bacterial status eradicates this advantage.

The small reduction of phosphorus (by an average of 24% of TP) was lower than that obtained from the Kirinya pilot wetlands (Chapter 4). But the influent concentrations into the Kirinya systems were very low (1.4 – 6.5 mg/l) compared to those loaded into this small wetland unit (15.5 mg/l). Large quantities of phosphorus were entering the wetland from the daily washing usually with bar soap containing 5 g P / kg. It was estimated that about 0.2 m^3

of laundry washings were entering the wetland daily, particles of soap were sometimes observable in the unit. The present size and design of this household constructed wetland with no obvious mechanisms for P removal such as adsorption and precipitation by aluminium, iron and calcium ions limits its ability to remove phosphorus from the wastewater. Household wetland systems reported by Steiner and Combs (1993) with elevated phosphorus removal had gravel or soil substrate.

The full potential plant biomass yields from this small system could not be ascertained due to the persistent interference by wind. Nevertheless, the yields obtained in the time of measurement were within the range reported by Kansiime and Nalubega (1999) and Balirwa (1998) in the natural wetlands within the L. Victoria catchment. The papyrus yields obtained at Kirinya pilot wetlands were however, twice as high (5.76 kg m^{-2}) as that obtained from this small wetland (2.35 kg m^{-2}). The difference is linked to the wind effects, which were more severely felt in the latter systems. The nutrient content in the various plant parts (Table 6.4) were similar to that measured for the plants at Kirinya (Table 4.1) although phosphorus content in the small wetland tended to be higher although not significantly. The high P loading into the small wetlands may have influenced the P uptake capacity. The contribution of plant uptake to removal of the nutrients in the small wetland was not quantified in this system. From the similarity in the nutrient content in the different plant parts with that of the Kirinya systems, it can be concluded that nutrient uptake had a significant role when the plant were young.

The nuisance and the risk of malaria infection from the mosquitoes breeding in the wetland units was by far the most serious concern of the users of the facility rather than the treatment that was taking place. Although the types of mosquitoes breeding in the wetland were not identified, their mere presence in large numbers was sufficient to raise the fears. The small wetland was a free surface water flow type and it was assumed, like in Kirinya, that the thick papyrus-rooting mat would be formed rapidly to cover all the exposed water surfaces which are the potential breeding areas. However due to the strong winds experienced at the site, mat establishment was very slow leaving behind open water zones. Bacterial insecticides and mosquito eating fishes, *Gambusia affinis* have been applied elsewhere to control mosquito larvae populations in treatment wetlands (Reed *at al.*, 1995; Wieder *et al.*, 1989). Attempts to get this fish species from the nearby Jinja Fisheries Research Institute (FIRI) were not successful. The only control applied was the use of the public health insecticide which, was an expensive and non- sustainable option.

Subsurface flow systems are more attractive in the location of the household-scale wetlands since water exposed zones are definitely minimised regardless of the plant type used. In view of the public health concerns, it is concluded that the design (surface flow) applied in this study was not the appropriate choice for the area. Secondly, papyrus plants are prone to wind destruction, the regular pruning activity performed to ensure its continued functioning is a labour input that can discourage potential users. Plants resilient to wind effect such as *Phragmites* spp. or *Typha* are possible alternatives and are suitable for subsurface flow systems. From the comments of the users and members of the general public and from the researchers' view, it is evident that the wetland technology for wastewater treatment is a new concept and is not yet understood by a wider public. Although great interest was expressed, more public awareness and demonstration is essential not only for the small systems but also for the bigger ones such as Kirinya.

Despite the apparent shortcomings in the design and operation of this household wetland system, the owner of the house (Dr. Mukwenda) was pleased to be associated with innovative research into water pollution problems in the country. Continued co-operation in any future modifications in this demonstration wetland has been assured.

6.7 Conclusions

This study has demonstrated that small constructed wetlands at household level in the tropical environments have the capacity to remove organic pollutants as well as suspended solids to acceptable discharge levels. Under the present design and operating conditions, nutrients and faecal coliform removal capacity of this small wetland system was very low. The septic tank was mal-functioning and it affected to a large extent the overall performance of the small wetland. Good pre-treatment is essential for this type of systems to work and hence any future application should be against a functioning septic tank or any pre-treatment form with a longer detention time. Mosquito breeding must be minimised by use of subsurface flow systems. Surface flow systems although cheaper to construct are definitely not a good choice for use in home locations especially in the tropical environments.

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Chapter 7

Economic Valuation of Constructed Wetlands' Potential in Uganda

7.1 Introduction 145

7.1.1 Ecological valuation 145
7.1.2 Economic valuation 145

7.2 Constructed wetlands 146

7.2.1 Rationale 146
7.2.2 Construction costs 147
7.2.3 Operation and maintenance costs 147

7.3 Economic appraisal of a constructed wetland 148

7.3.1 Investment and recurrent cost aspect. 148
7.3.2 Non-monetary considerations 150

7.4 Conclusions 151

References 151

Appendix 153

Appendix 1: Design criteria and process design of a constructed wetland for a population equivalent of 4000 155
A. Criteria used 155
B. Process design 155

Appendix 2: Design criteria and process design of a waste stabilisation for a population equivalent of 4000 $^{(++)}$ 157

A. Design criteria 157
B. Process design 157

Appendix 3: Investment and Recurrent costs of the constructed wetland 159

A. Investment costs 159
B. Annual recurrent costs 159

Appendix 4: Investment and Recurrent costs of the waste stabilisation pond 159

A. Investment costs 159
B. Annual recurrent costs 159

Appendix 5: Present value cost calculation for the Constructed Wetland 160

Appendix 6: Present Value Costs calculation for the Waste Stabilisation ponds 160

7.1 Introduction

Wetlands are dynamic and among the most productive ecosystems with immense significance to mankind. The biotic and non-biotic values of wetlands from a historical perspective, can be linked to early civilisations along the major rivers such as the Tigris, Euphrates and the Nile where the wetlands provided food, drinking water, fish, etc to the people settled in these regions. The hydrological and chemical cycling functions and attributes of wetlands are well recognised and it is the basis for the description of wetlands as the kidneys of the landscape (Denny, 1996) and as biological supermarkets because of the extensive food webs and rich biodiversity they support (Mitsch and Gosselink, 1993).

However, despite these perceived ecological values of wetlands, loss of natural wetlands world wide in the recent times is very rapid. In the different countries, the administrators and politicians commonly view wetlands as wastelands that impede development. Barbier *et al* (1997) suggest that the failure to quantify and account for the non-market ecological values of wetlands is the main cause of this misconception and is a driving force for the depletion of wetlands. Many decision-makers easily understand the values of a resource if there is a monetary measure attached to it.
A combination of ecological and economic valuation methodologies have been applied to quantify these perceived free services and benefits derived from natural wetlands.

7.1.1 Ecological valuation

The ecological valuation approach as described by Odum E. T (1979) is a numerical ranking method on an arbitrary scale, of identified values of a specific wetland that can be applied to assess its habitat value. This valuation approach is site specific, it is subjective and cannot be duplicated in other wetlands. It also tends to appeal more to the scientists than the decision-makers. Its application combined with economic analysis could be of significance in constructed wetland technology option considerations.

7.1.2 Economic valuation

Economic valuation of wetlands attempts to put a monetary denominator to all identified ecological values of wetlands. The concept of total economic valuation of wetlands which is a framework for distinguishing and grouping wetland values has been suggested by Folke (1991), Mitsch and Gosselink (1993) and Barbier *et al* (1997) as the most appropriate. The method distinguishes the Use Values and Non-use Values of a wetland. The wetland use values that are relevant to treatment wetlands are subdivided into two categories: direct and indirect use values. The direct use values may be related to the identifiable direct benefits to the individual or the owners of the facility, while the indirect use value are linked to the benefits that can be applied to a community at large. The existence value is a non-use value and it is a benefit that does not involve human interaction. A summary of the direct and indirect user values that can be linked to treatment wetlands is given in Table 7.1.

The economic importance of each of the values listed depends on the location of the wetland (tropical or temperate regions), and the abundance and diversity of the species that control that specific value. For constructed wetlands, the direct use value of wastewater treatment can be appreciated when related to the cost of treating wastewater using conventional systems. The costs of setting up a conventional wastewater treatment system particularly in the poor countries are often prohibitive.

Table 7.1 Classification of economic values for treatment wetlands based on identified use and existence values.

Direct Use Value	Indirect Use value	Existence value
• Fish • Fuel wood • Recreation • Wildlife harvesting • Energy / peat • Wastewater treatment	• Ground water recharge • External ecosystem support • Micro-climate stabilisation • Scientific study	• Bio-diversity • Cultural

Source: Barbier *et al* (1997).

Depending on the location, size and type of macrophytes used in the wetland, other direct use values such as fuel wood, matting and recreation can be of significant monetary value. Kansiime and Nalubega (1999) estimated an income of nearly US $ 150 per month per harvest in the dry season from papyrus harvesting in the Nakivubo wetland. In constructed wetlands, the income from this source can still be useful to the operators. A non-use value such as bio-diversity will be a direct consequence of establishing the constructed wetland systems in a given locality.

Treatment wetlands like natural wetlands have the potential negative effect of attracting nuisances to both wildlife in the habitat and humans e.g. mosquitoes and other biting insects. It should be considered when valuing treatment wetlands.

7.2 Constructed wetlands

7.2.1 Rationale

The main attraction for using constructed wetlands in pollution control has not only been due to their functional values but also because of the economic cost involved in establishing them. Several authors have indicated that the capital and operational costs of constructed wetlands are low when compared to the conventional systems (Brix and Schierup, 1989; Cooper and Findlater, 1990; Denny, 1997). The size of land required for siting constructed wetlands is within the same range as for waste stabilisation ponds and this is often twice the land areas used for conventional treatment systems established in similar environments (Mara and Caincross, 1989). Hence, one of the biggest investment costs in setting up constructed wetlands may be acquisition of land. However, Kadlec and Knight (1996) urge that the price of land becomes insignificant if the life expectancy of constructed wetlands and the salvage value are considered together. The life expectancy of conventional treatment systems is usually 20 years with no salvage value given to the mechanical parts etc. In contrast, life expectancy of a treatment wetland is longer and by the end of 20 years, the land associated with the wetland will have more or equal value than at the time of acquisition.

7.2.2 Construction costs

The construction costs of wetlands are variable due to several influencing factors that are specific for a given site and depends on what is already at the site (Mitsch and Gosselink, 1993; Kadlec and Knight, 1996). The main factors that affect the cost include among others the following:

(i) location and topography;
(ii) accessibility;
(iii) wetland design and type (surface or sub surface flow);
(iv) site improvements;
(v) compliance monitoring infrastructure;
(vi) availability of field equipment.

Typical construction cost derived from some of the constructed wetlands obtained from the North American database are shown in Table 7.2 (Knight *et al.*, 1993; Kadlec and Knight, 1996). The values show subsurface flow wetlands to be about eight times more costly than surface flow ones. This is mainly linked to the costs associated with the procurement of the substratum, transportation and filling of the wetland.

Table 7.2 Capital costs for surface and subsurface flow treatment wetlands in the United States, (1993). (Source: Kadlec and Knight, 1996).

System	Wetland type	Area (ha)	Flow (m^3/d)	Cost (US$/ha)
Eureka, SD	FWS	16.34	1,045	32,359
Arcata, CA	FWS	33	8,781	44,622
Show Lor, AR	FWS	52.2	5,299	4,508
Ironbridge, FL	FWS	494	75,720	53,840
Everglades ENR, FL	FWS	1,406	636,208	9,957
Phillips High School, AL	SSF	0.20	76	262,496
Sibley, LA	SSF	0.21	492	310,969
Benton, LA	SSF	0.48	1,173	866,169
Carlisle, AR	SSF	4.35	3,255	77,199
Denham Springs, LA	SSF	6.15	11,355	358,373

Key: FWS = Free water surface flow; SSF = Sub- surface flow.

7.2.3 Operation and maintenance costs

The operation and maintenance (O & M) costs reported correlate with the type and size of the constructed wetland system. The general factors that contribute to O & M include but are not limited to the following:

(i) site maintenance, e.g. access roads, lawn ;
(ii) pumping energy;
(iii) equipment maintenance;
(iv) compliance monitoring;
(v) control of nuisance, e.g. mosquitoes, burrowing rodents;
(vi) personnel costs, highly trained man power is not required.

The variability in the capital costs highlighted in Table 7.2 are also reflected in the different European constructed wetland systems reviewed. In the Netherlands, the total investment cost including land cost, for typical surface flow wetlands with a design flow of 2800 m^3/day is estimated to be in the range of US $ 200,000 to 250,000 per ha. (Veenstra, 1998). The investment costs (including land) in the United Kingdom of a recent subsurface wetland designed for 25,000-population equivalent (PE), is cited by Cooper and Green (1998) to be US $ 680,000 per ha. In Poland, the average capital cost ranges between US $ 90,000 to 1,5 million per ha for subsurface systems and between US $ 40,000 to 400,000 per ha for free water surface flow systems (Kowalik and Obarska-Pempkowiak, 1998). The estimated cost in Thailand based on data of Koottatep (1999) is US $ 130,000 per ha (excluding cost of land) for a subsurface wetland receiving 600 m^3/day.

There is no earlier comparative data in sub-Saharan Africa. The cost for the construction of the Kirinya pilot wetlands (320 m^2) that were designed for a maximum flow of 40 m^3/day was US$ 33,000 (excluding land costs). The Kirinya wetland units were established for carrying out research and optimisation, the cost does not reflect the case when the systems are built for routine treatment. In order to understand the viability of the constructed wetland technology in Uganda, a comparative analysis with the common technology option of waste stabilisation ponds is essential. The principal objective of the study was therefore to determine the cost implications of selecting a constructed wetland or a waste stabilisation pond for treating domestic wastewater for a defined size population.

7.3 Economic appraisal of a constructed wetland

7.3.1 Investment and recurrent cost aspect.

Constructed wetlands and waste stabilisation ponds have a potential to be applied in many parts of Uganda. The wastewater treatment processes in both systems are greatly influenced by the prevailing tropical weather conditions. In the present economic circumstances in the country, they could be the first wastewater treatment options that are considered for use in small towns and institutions with a centralised sewage collection system. Currently, waste stabilisation ponds are the only systems considered for application in such environments.

In order to assess the full potential for using the constructed wetland technology in Uganda, a comparative costing with waste stabilisation pond systems was done. This was conducted through an economic appraisal of a constructed wetland and a stabilisation pond system designed for a population equivalent of 4,000. It involved determining the Total Annual Costs (TAC) of both systems based on the calculated total Present Value Costs (PVC). The capital, operation and maintenance costs were based on the existing pricing used by the National Water and Sewerage Corporation (NWSC, 1999) and those derived from the Kirinya pilot wetlands. The price of the land was obtained from the Uganda Land Commission. A 10% interest rate was applied (World Bank rates in Uganda).

The detailed design criteria and process design computations for the constructed wetland and waste stabilisation ponds are given in appendices 1 and 2, respectively. The investment and recurrent costs based on the design are summarised in appendices 3 (constructed wetland) and 4 (waste stabilisation pond).

The total present value costs (PVC) was determined by summing up all the annual incremental capital, operating and maintenance costs over the 20 year design period. The total annual costs (TAC) for each treatment system was calculated by multiplying the present value costs (PVC) (Franceys, 1999; Kootattep, 1999) with the capital recovery factor (CRF) as shown in the equations 7.1 and 7.2. Details of the calculation of PVC for each treatment system is given in appendices 5 and 6.

$$TAC = CRF \times \text{Total present value cost.} \tag{7.1}$$

Where: TAC = total annual costs
CRF = capital recovery factor

$$= \left[\frac{i(1+i)^n}{(1+i)^n - 1} \right] \tag{7.2}$$

i = interest rate, % (10 %)
n = design period (20 years).

Using equation 7.2, a CRF of 0.117 was calculated.

A summary of the appraisal comparative data for the two systems is presented in Table 7.3.

Table 7.3 Comparative costing for the constructed wetland and waste stabilisation pond wastewater treatment systems (for a PE of 4000) and based on the costs in Uganda (as of August 1999)

Costs (US $)	Constructed wetland systems	Waste stabilisation systems
Investment	70,667	101,527
Total Present Value, PVC		
Capital	61,323	88,102
Recurrent	35,802	35,203
Total Annual, TAC		
Capital	7,202	10,348
Recurrent	4,205	4,135
Land area (ha)	0.72	0.84

The data from table 7.3 show that the annuitized investment cost which includes the cost of land for waste stabilisation ponds was 30% more than that for constructed wetlands. However, the operating and maintenance costs were similar for both types of systems, which implies that the amount of money spent annually for both technologies should be the same. The total annual cost (TAC) for a conventional treatment system under similar conditions (derived from lumped investment costs of US $ 1000/m^3 and recurrent cost of US $ 0.3/m^3) was US $ 40,700 (capital) and US $ 34,870 (recurrent). This comparison indicates that conventional systems are a more expensive alternative to use in this environment than either waste stabilisation or constructed wetland systems. Based on this economic appraisal, it is concluded that constructed wetlands are economically viable alternative for wastewater treatment in Uganda. They can be built and maintained at competitive prices as compared with stabilisation ponds.

7.3.2 Non-monetary considerations

The analysis given in 7.3.1 was based only on investment and recurrent cost. However, other considerations that cannot easily be transformed into direct monetary terms are equally important in selecting the treatment options. These are considered in this section.

(i) *Treatment performance*

Stabilisation ponds generate secondary BOD and TSS from the algal biomass. This can influence the quality of the water in the receiving water body especially if it is small or shallow. Stabilisation ponds reduce organic nitrogen to a large extent but it gets converted into inorganic nitrogen. All these nitrogen forms have a potential negative effect on the receiving water in the natural environment. From the Kirinya studies, organic and suspended matter was reduced in the vegetated wetlands to levels that meet the Uganda National Standard (NEMA, 1999). The removal of the ionised forms of nitrogen was also enhanced by deliberate configuration modifications of the vegetated wetland. Wetland systems provide this advantage. A monetary value may be assigned to this advantage of constructed wetlands over the waste stabilisation ponds basing it on the penalties that are levied for non-compliance.

(ii) *Value costs*

The use and non-use values (Table 7.2) of wetlands when included in the comparison can be assigned a ranking that can be related to a monetary cost. The size of the Kirinya wetland size and the duration of the experiments were not sufficient for quantification of products linked to these use values. However, it is envisaged that the system proposed can give the additional benefits. The ranking is considered in Table 7.4.

Table 7.4. Ranking of the use value functions identified for treatment wetlands in comparison with the waste stabilisation ponds

Use Value	Constructed wetlands	Waste stabilisation ponds
Fish production	+	+
Fuel wood	+	-
Recreation	+	-
Wildlife	++	+
Matting	+	-
Scientific study	++	+
Bio-diversity	++	+

Key: (+) indicates positive contribution and (-) no or insignificant contribution for that functional value. Ranking is based on the observations derived from Kirinya pilot wetlands and Kirinya waste stabilisation ponds.

Fish is present at the maturation ponds of the NWSC Kirinya waste stabilisation ponds at Jinja but it is hardly harvested and marketed because of the prejudice held by many consumers against direct sewage products. Presently, consumers readily accept fish caught in the adjacent natural wetland that receives effluent from the treatment works. It is envisaged that fish obtained from a constructed wetland will also be more socially acceptable to the consumers. Given the high productivity of the tropical plants (> 100 kg ha^{-1}, Chapter 4) biomass harvest products are important for various applications that can generate financial benefits to the persons involved. The wildlife, scientific study and bio-diversity use value costs derived from constructed wetlands can be associated to both water and vegetation while in waste stabilisation ponds it is only from water.

From the above evaluation, it is evident that constructed wetlands treatment systems have an added advantage over the waste stabilisation ponds.

(iii) *Nuisance considerations*

Constructed wetlands as well as stabilisation ponds can be associated with some values that may be considered hazardous. For example, the high organic loading and dense vegetation growth in wetlands makes them have a potential for production of mosquitoes and other biting insects. Poisonous snakes also tend to stay in the cool environment provided by the wetlands. This was the case in Kirinya. The stabilisation ponds on the other hand have a reduced impact of these insects especially if well designed (Kalbermatten *et al.*, 1982). Although the monetary impact of these nuisances cannot be quantified based on the work at Kirinya, it is still essential that they be considered when selecting a technology to be applied in a given location.

In conclusion, it may be assumed that as the wetland technology use increases in the region and the market value for its other ecological values is well quantified, the investment and recurrent costs will get reduced further.

7.4 Conclusions

From the economic appraisal and ecological analysis carried out on the two treatment systems, the following deductions were arrived at:

1) The total annual costs (TAC) derived show that constructed wetlands can be established at lower investment costs than stabilisation pond systems. Both systems are more cost effective than conventional treatment systems.
2) The use of only the investment and recurrent costs in appraising the potential applicability of the wetland technology tends to ignore other aspects that may be of significance
3) There are added functional advantages that can be gained by using the wetlands instead of the stabilisation ponds.
4) The use of constructed wetlands can be considered both technically and economically as a viable option in the tropical environments.

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APPENDIX

Economic Appraisal of constructed wetlands and waste stabilisation ponds

Appendix 1: Design criteria and process design of a constructed wetland for a population equivalent of 4000

A. Criteria used

Volumetric Loading:	400 m^3/day
BOD of raw sewage	350 mg/l
BOD of influent wastewater	150 mg/l
Desired effluent BOD	30 mg/l
Temperature of water in the wetland	23 °C

B. Process design

(i) *Sedimentation pond*

Details as for anaerobic pond in Waste stabilisation pond design
Area, A_{sed} = 156 m^2 (12 m x 13 m) and depth of 3 m.

(ii) *Constructed wetland*

1. Wetland type:
   Subsurface flow system with *Phragmites* or papyrus as the macrophyte;
   The rooting depth, d = 0.3 m (based on the Kirinya data).

2. Bed slope, S = 2 % (0.02) (for ease of construction).

3. Media coarse sand or gravel, the porosity , n = 0.35 ,
   Hydraulic conductivity, k_s = 500 m^3/m^2-d

4. The temperature dependant rate constant, K_T:
   $K_T = K_{20}\,(1.1)^{(T-20)}$; (T = 23 °C)
   $= 1.14\ d^{-1}$

5. Cross section area, A_x, of the wetland bed:
   $A_x = Q/k_s\,.S$
   $= 400/(500 \times 0.02) = 40\ m^2$

6. Width of the wetland determined using equation:
   $W = A_x / d$
   $= 40/0.3 = 133$ m

7. Surface area of the wetland determined using equation:
   $A_s = (Q\,(\ln C_i - \ln C_e))/(K_T.d.\,n)$
   $= 400(\ln 150 - \ln 30)/(1.14 \times 0.3 \times 0.35)$
   $= 5356\ m^2$

8. The length of the wetland determined using equation:
   $L = A_s/ W$

= 5356/ 133
= 40 m

9. Total surface area for the wetland and the sedimentation pond:
   $A_T = A\ s + A_{sed}$
   $= 156 + 5356$
   $= 5512\ m^2$

10. The wetland is divided into individual cells each, 13 m for an improved hydraulic control at the inlet. The total number of cells to be constructed is 10, each 13 m x 40 m.

11. To increase nitrogen removal via the nitrification-dentrification pathway, 2 open zones of area 10 m^2 in alternation with the planted segment will be created and maintained in each of this cells (Justification, Chapter 4).

12. The adjacent free land area required (30%) of the built area: 1655 m^2.

13. Total land area requirement for the treatment system: = 7200 m^2.

Appendix 2: Design criteria and process design of a waste stabilisation for a population equivalent of 4000 (++)

A. Design criteria

Volumetric loading, Q (m^3/day)	400
Influent BOD concentration, C_i (mg/l)	350
Faecal coliform counts in the influent, N_i (No./100 ml)	1E+08
Faecal coliform counts in the effluent, N_f (No./100 ml)	1000
Water temperature, T (° C)	23
Evaporation rate, ET (mm/d)	6

B. Process design

(i) *Anaerobic pond*

Volumetric BOD loading, λ_v ($gm^{-3}.d^{-1}$)	300
Pond volume (m^3), ($V_a = C_i . Q/ \lambda_v$)	467
Retention time (days), ($\Theta_a = V_a/Q$)	1.17
BOD removal (%)	60%

(ii) *Facultative Pond*

Depth, D (m)	1.5
Surface BOD loading, ($kgha^{-1}d^{-1}$), ($\lambda_s = 350(1.107\text{-}0.002 .T)^{(T-25)}$)	311
Surface area (m^2), ($A_f = 10\ C_i.Q / \lambda_S$)	4503
Retention time (days), ($\Theta_f = 2\ A_f.\ D / (2.Q - 0.001\ A_f.\ ET)$)	17
Effluent flow (m3), ($Q_e = Q - 0.001\ A_f.\ ET$)	373

(iii) *Maturation Pond*

Depth, D_2 (m)	1.5
First order rate constant for faecal decay, $K_T = 2.6\ (1.19)^{(T-20)}$	4.38

Retention time, ($\Theta_m = \{(Ni/Ne(1+K_T.\Theta_a)(1+K_T.\ \Theta_f))^{1/n} - 1\}/ K_T$)

n	Θ_m
1	*108202 13*
2	*1571.3*
3	*82.4*
4	18.7
5	7.6
6	4.1
7	2.6

When n = 1,2,3,4; is rejected since $\Theta_m > \Theta_f$
n = 7 rejected because Θ_m is less than the minimum 3 days
Comparison between n = 45,6;
Θ_m * n product is least when n = 6 and it is selected

(Checking loading, assume 70 % BOD removed in preceding ponds)

Surface BOD loading ($kgha^{-1}d^{-1}$), ($\lambda_m = 10(0.3\ .\ C_i).\ D_2/\Theta_m$)	382.9
Value is more than 75 % of load to facultative pond (0.75 x 311)	233.18
Surface loading considered is 233.18	
Retention time, Θ_m is therefore = ($10.C_i.D_2.\ 0.3/\lambda_m$)	6.8

Pond surface area (m^2), $(2.Q.\theta_m/(2.D_2 + 0.001.ET.\theta_m)$ 1777

(iv) *BOD removal for whole system*

Assumption: 90 % removal in the anaerobic and facultative ponds and 25 % in maturation pond:

Final effluent concentration:(mg/l), (350 x 0.1 x 0.75) 26.25

(v) *Summary:*

Anaerobic pond	volume	m^3	467
	retention time	d	1.17
Facultative pond	area	m^2	4503
	retention time	d	17
	depth	m	1.5
Maturation pond	area	m^2	1777
	retention time	d	6.8
	depth	m	1.5
Total retention time		d	25.4
Total surface area (156 m^2 for anaerobic ponds)		m^2	6436
Total Land requirement for pond systems		m^2	8400

(++): Criteria and method adopted from: Mara D.D; Alabaster, G. P. and Mills, S. W. 1992. Waste stabilisation ponds. A design manual for Eastern Africa, *Lagoon Technology International*, Leeds, England.

Appendix 3: Investment and Recurrent costs of the constructed wetland

A. Investment costs	Unit	Rate (US $)	Quantity	Amount (US $)
1. Excavation works	m^3	5	3145	15,725
2. Side lining –sedimentation pond	m^2	8.4	150	1,260
3. Lining bottom works -sedimentation pond	m^2	2	600	1,200
4. Leak-proof wetland bottom works	m^2	2	5356	10,712
5. Lining sides of wetland with LPDE	m^2	4	530	2,120
6. Refilling of wetland + planting	m^3	5	1610	8,050
7. Land costs	ha	30000	0.72	21,600
8. Other miscellaneous works	L/S			10,000
Total Capital (US $)				**70,667**
B. Annual recurrent costs				
9. Technician	MM	280	12	3,360
10. Maintenance of site	L/S			240
10. Other repair costs	L/S	60	12	720
Sub/Total 1				**4,320**
11.Desludging every after 5 years	L/S	2500	1	2,500
12.Plant harvesting, every 2 years (half area)	m^2	0.5	2678	1,339
Sub/Total 2				**3,839**

Appendix 4: Investment and Recurrent costs of the waste stabilisation pond

A. Investment costs	Unit	Rate (US $)	Quantity	Amount (US $)
1. Excavation works	m^3	5	9887	49,435
2. Side lining –sedimentation pond	m^2	8.4	150	1,260
3. Lining sides of facultative and maturation ponds	m^2	4	690	2,760
4. Leak-proof pond bottom works	m^2	2	6436	12,872
5. Land costs	ha	30000	0.84	25,200
6. Other miscellaneous works	L/S			10,000
Total Capital				**101,527**
B. Annual recurrent costs				
7. Technician	MM	280	12	3,360
8. Maintenance of site	L/S			240
9. Other repair costs	L/S	60	12	720
Sub/Total 1				**4,320**
10.Desludging every after 5 years	L/S	5000	1	6,000
Sub/Total 2				**6,000**

Appendix 5: Present value cost calculation for the Constructed Wetland (All costs are in US $)

Year	Capital Cost	Recurrent Cost	Desludging and plant harvest costs	Total recurrent cost	Discount Factor 10%	Present value costs - Capital	Present value costs - Recurrent
1	35333.5				0.909	32121	0
2	35333.5				0.826	29201	0
3		4320		4320	0.751	0	3246
4		4320	1340	5660	0.683	0	3866
5		4320		4320	0.621	0	2682
6		4320	1340	5660	0.564	0	3195
7		4320		4320	0.513	0	2217
8		4320	3840	8160	0.467	0	3807
9		4320		4320	0.424	0	1832
10		4320	1340	5660	0.386	0	2182
11		4320		4320	0.350	0	1514
12		4320	1340	5660	0.319	0	1803
13		4320		4320	0.290	0	1251
14		4320	3840	8160	0.263	0	2149
15		4320		4320	0.239	0	1034
16		4320	1340	5660	0.218	0	1232
17		4320		4320	0.198	0	855
18		4320	1340	5660	0.180	0	1018
19		4320		4320	0.164	0	706
20		4320	3840	8160	0.149	0	1213
		TOTAL PRESENT VALUE COSTS				**61,323**	**35,802**

Appendix 6: Present Value Costs calculation for the Waste Stabilisation ponds

	Capital Cost	Recurrent Cost	Desludging cost	Total recurrent cost	Discount Factor 10%	Present value costs - Capital	Present value costs - Recurrent
1	50763.5				0.909	46149	0
2	50763.5				0.826	41953	0
3		4416		4416	0.751	0	3318
4		4416		4416	0.683	0	3016
5		4416		4416	0.621	0	2742
6		4416		4416	0.564	0	2493
7		4416		4416	0.513	0	2266
8		4416	6000	10416	0.467	0	4859
9		4416		4416	0.424	0	1873
10		4416		4416	0.386	0	1703
11		4416		4416	0.350	0	1548
12		4416		4416	0.319	0	1407
13		4416		4416	0.290	0	1279
14		4416	6000	10416	0.263	0	2743
15		4416		4416	0.239	0	1057
16		4416		4416	0.218	0	961
17		4416		4416	0.198	0	874
18		4416		4416	0.180	0	794
19		4416		4416	0.164	0	722
20		4416	6000	10416	0.149	0	1548
		TOTAL PRESENT VALUE COSTS				**88,102**	**35,203**

Chapter 8

Summary and Conclusions

8.1 Introduction 163

8.2 Treatment performance evaluation and economic appraisal 163

8.3 Guidelines for design, operation and management 168

8.3.1 Wetland design and operation 168
8.3.2 Optimising nutrient removal and vegetation management 169
8.3.3 Wetland Construction 169

8.4 Limitations and Recommendations 169

8.1 Introduction

Wastewater treatment using the conventional sewage treatment systems approach is a convenient technology that is in use in many industrialised countries. Nevertheless, it is still an expensive venture that can not be delivered to the majority of the populations in the developing nations. High population growth rates and rapid urbanisation are a common feature in these countries, but sewage service delivery like many other sectors is most often out paced by these changes. There is need for utilising alternative treatment methods that are efficient, cost effective and can be applied to a wider community. Treatment wetlands, both natural and constructed, are such an alternative that have been researched and applied mainly in the temperate regions but to a negligible extent in the tropical regions and particularly, in sub-Saharan Africa. This study served as a basis for research and demonstration of the potential application of the constructed wetland technology within the East African sub-region. The existing warm equatorial and tropical climatic conditions in the region are expected to stimulate microbial activity and plant growth throughout the year and this will enhance continuous removal of pollutants from wastewater. The treatment efficiencies derived are expected to be of a different order from that of the temperate region due to this variance in climatic conditions.

The goal of this study was focussed on an improved understanding of the functioning of constructed wetlands as wastewater treatment systems in the tropical environment and optimisation of the influencing variables. Demonstration and raising awareness of the technology was a crucial factor of the study. The specific objectives of the study were:

(i) to determine the processes and treatment performance attainable in constructed wetlands of *Cyperus papyrus* and *Phragmites mauritianus* plant species with respect to COD, BOD, TSS, nutrients (N & P) and pathogens under different operating conditions.

(ii) to determine the functional role of the macrophytes in the uptake of nutrients and their storage capacities in the standing and rooting biomass

(iii) to evaluate the design and performance of a household scale constructed wetland

(iv) to propose guidelines for design, operation and management of the wetlands based on the acquired knowledge and the costs involved.

8.2 Treatment performance evaluation and economic appraisal

The studies were carried out at Kirinya pilot constructed wetland located within the Jinja municipality in Uganda. The wetland was divided into eight units each of 2 m x 20 m x 1 m and was built such that leakage or ground water infiltration was minimised. Pre-settled wastewater, diverted from the outflow of the anaerobic lagoons of the Kirinya Sewage Works was used as the influent to the wetlands. *Phragmites mauritianus* were planted in two of the units containing a gravel base and *Cyperus papyrus* was planted in four units floating without a substratum base. Two units were used as controls. The percentage plant cover and pond length defined the wetland unit configurations in three different phases applied. In the first phase (April 96 to March 97), the plants covered all the surface areas of the vegetated units. In the second phase (September 1997 to April 1998), the total biomass both standing and rooting, was removed from an area of 10 m^2 near the influent position in two papyrus and one *Phragmites* wetland unit. In the last phase (August 1998 to March 1999), the pond lengths were extended by two modes; firstly by combining two planted wetland units in series with a non-vegetated one, the latter being in the middle of the series. Secondly, by combining two papyrus' wetland units with small areas of alternating planted and unplanted sections, each 10 m^2. Wastewater was loaded into the units on a weekly batch-wise format in the first phase and the first half of the second phase before adopting a daily load format up to the third phase. Hydraulic loading rates were varied independently in each wetland unit within the range of 1.3 to12 cm/day. The optimal hydraulic loading rate in the range 2 - 5 cm day^{-1} deduced from the first two phases was applied in the series configurations in phase 3. Continuous monitoring of the climatic variables, namely wetland water temperatures, rainfall and evapotranspiration was done.

The average temperatures of the water in vegetated and non-vegetated wetland units were 23 $\pm$ 2 $^{\circ}$ C and 27 $\pm$ 3 $^{\circ}$ C respectively. The mean annual rainfall measured at the study site varied over the experimental period from 3.3 to 4.3 mm day^{-1}. Average evapotranspiration rates of 6.1 mm day^{-1} and 5.6 mm day^{-1} were determined for *Cyperus papyrus* and *Phragmites mauritianus* plants compared to 3.8 mm day^{-1} in the control open ponds. The open water evapotraspiration rate of 4.1mm day^{-1} was determined in earlier studies at the Nakivubo natural wetland fringing Lake Victoria.

The changes in the environmental parameters namely oxygen, pH and temperature during each of the three operating phases was measured. Reduced oxygen concentrations less than 2 mg/l and a well-buffered pH in between 7 – 7.5, characterised the wetland units when fully covered by the vegetation. In the control open ponds, these parameters showed strong diel variations with day peak dissolved oxygen concentrations of 27 mg/l, pH values of 10.4 and maximum water temperatures of 30 $^{\circ}$C. Algal photosynthesis resulted into these conditions in the open ponds. However, with reduced open surface areas, intermediate conditions prevailed; maximum day oxygen concentration of 12 mg/l, pH values of 7.5 and temperature ranged between 22 – 23 $^{\circ}$C. The influence of the three distinct characteristics in each of the operating conditions greatly impacted the processes and extent of reduction of the pollutants achieved in each case.

The results from all the three phases showed that the COD, BOD and TSS removal efficiencies by both the *Cyperus papyrus* and *Phragmites mauritianus* wetland units were high. Removal of over 70% of the input COD (350 $kg.ha^{-1}$) and BOD (100 $kg.ha^{-1}$) and 80% of TSS input (150 $kg.ha^{-1}$) into the single units in the first phase was realised. In the serial combined wetland units, over 80% of the inputs of all these bulk parameters were removed. Their effluent concentrations were consistently low and did not exceed the Uganda regulatory discharge limits of 50 mg/l BOD and 100 mg/l COD and TSS. In the control open ponds however, effluent concentrations of these parameters measured for the whole experimental period were beyond the standard limits. This was linked to the secondary products of the decaying algal biomass. Although both macrophytes are concluded to be suitable for use in constructed wetlands to remove the bulk parameters, *Cyperus papyrus* showed a comparative advantage over *Phragmites mauritianus*. The low residual background BOD concentration of 12 mg/l in papyrus units as compared to 17 mg/l in *Phragmites* units further illustrated this difference. Furthermore, the rate of BOD decay in papyrus was higher (0.084 m d^{-1}) than in *Phragmites* (0.039 m d^{-1}).
Removal of nitrogen and phosphorus from wastewater by the different wetland units was dependent on the growth stage of the plants, the biomass yield and the gravel substratum properties. Maximum dry weight standing biomass yields for the two plant types were

similar, 108 ton.ha^{-1} and 104 ton.ha^{-1} for papyrus and *Phragmites*, respectively. The average papyrus standing biomass yields over all the growth phases was higher, 57.6 ton ha^{-1} as compared to *Phragmites'* 34.4 ton ha^{-1}. The rooting biomass yield on the other hand averaged 66 ton.ha^{-1} for papyrus and only 4 ton.ha^{-1} for *Phragmites*. Nutrient storage in the papyrus rooting biomass was correspondingly high; 1600 kg N.ha^{-1} and 508 kg P.ha^{-1} for papyrus and 31 kg N.ha^{-1} and 5.7 kg P.ha^{-1} in *Phragmites*. Plant biomass is therefore an important nutrient reservoir.

Plant uptake by the standing biomass was a significant nutrient removal route when the plants were in the exponential growth stages. Uptake rates determined over this period were 2600 kg N ha^{-1}yr^{-1} and 93 kg P ha^{-1} yr^{-1} in papyrus, and 3796 kg N ha^{-1} yr^{-1} and 26 kg P ha^{-1} yr^{-1} in *Phragmites*. Mass balance consideration at the exponential phase also showed that 15% N and 10 % P in papyrus units and 58% N and 37% P in *Phragmites* units was removed via plant uptake. This contribution in both plant systems declined drastically to less than 4% when the plant reached a steady state growth stage. These results showed that nitrogen and phosphorus removal via plant uptake by these tropical plants was negligible in long established wetlands where plants are in a steady state growth phase.

Other nutrient removal routes apart from plant uptake were important. Nitrification-denitrification was deduced to be significant based on the measurements carried out in the three phases. Nitrogen reduction of only 25 % of the input ammonium (27 kg N ha^{-1}) was determined during the first phase when the shoot density was high (80 m^{-2}), most of it was attributed to plant uptake mentioned above. Low oxygen concentrations that prevailed in the vegetated wetland units limited ammonium removal via the nitrification-denitrification pathway. On the other hand, high pH and oxygen concentrations in the open ponds favoured volatilisation of ammonia gas instead. In the modified wetland configurations, 90% of the ammonium N input (26 kg N ha^{-1}) into the wetland series with small alternating vegetated and non-vegetated zones (S 1-2) was removed. The effluent ammonium concentrations were less than the Uganda regulatory discharge limit of 10 mg/l. The environmental conditions that existed in this series favoured nitrification-denitrification nitrogen removal pathway, which was calculated to have contributed up to 77 % of the nitrogen lost from the systems. This wetland unit configuration was deduced to be the most suitable one for removal of nitrogen from wastewater in the tropical environments. It not only gave high quality effluent but it was also cost effective since no external energy input for aeration was used; only the solar energy driven processes were exploited in a controlled manner.

Phosphorus removal via other routes other than by plant uptake was limited in both plant systems and did not vary with the change in the wetland configurations. The substratum used in the *Phragmites* wetland units had a low retention capacity; the average phosphorus mass reduction in these units was 45 %, which was not significantly different from that in papyrus units (34%). In the latter units, release of phosphorus was occasionally found. This was linked to release of stored P in the growth attached to the wall of the wetland units that periodically got sloughed out into the water. Improvement in the P removal may be achieved by incorporating high phosphorus adsorbing laterite gravel.

The removal of the faecal coliforms was significant, over 99.9 % (3 log units) in all the individual wetland units and in the serial configurations. Effluent faecal counts were less than the regulatory discharge limits of 100 fcu/ml. This high reduction was correlated with the removal of TSS and particulate organic matter within the wetland units. The lethal effect of direct sunlight and the generated secondary effects such as high pH especially the open wetland units were considered significant. This was particularly important given the daily sunshine duration of 11 hours throughout the year. At retention times above 7 days, faecal removal above 4 log units was realised. A shorter retention time of 4 days applied nevertheless gave more an acceptable removal of 3 log units. Since retention time has impact on both, the wetland surface area and hydraulic loading, a retention time between 4 – 7 days was considered optimum for faecal reductions up to the regulatory levels. The average rate constants for faecal coliform decay in the wetlands were 1.2 day^{-1} and 1.1 day^{-1} in *Cyperus papyrus* and *Phragmites mauritianus* wetland units, respectively. These high values indicate that the Kirinya pilot wetland environment was suitable for the rapid reduction of faecal coliforms. The rates are comparable to the decay rate ranges reported for the North American wetlands. The faecal decay rates determined in *C. papyrus* were consistently greater than those derived from *P. mauritianus* units. This manifests a better faecal retention capacity of the former plants due to large surface area of the open mesh massive root-rhizome structures.

These results overall showed the technical viability of using a constructed wetland within the tropical environments with respect to the removal of COD, BOD, TSS, pathogens and nutrients. An economic valuation of the constructed wetland considering the wetland design and cost implications augmented this potential. The main attraction for using constructed wetlands, has not only been promoted because of their technical functionality but also because of the favourable economic cost of setting them up. The construction costs derived from the several constructed wetlands built in United States and Europe indicated wide variability which was influenced by the local conditions and the price of land.

In Uganda and other countries within the sub-region, waste stabilisation pond systems are the technology of choice for municipal wastewater treatment. The principle reason is the low investment and operation costs as compared to the conventional treatment systems, and their ability effectively to reduce the organic pollutants and pathogens. Constructed wetlands are considered to have the same potential but with added advantages and efficiency. In the Kirinya findings, nitrogen was removed to concentrations that meet the discharge standards unlike in the existing waste stabilisation pond. To assess the full potential of using the constructed wetland technology in Uganda, an economic appraisal of a constructed wetland and waste stabilisation pond systems was done. The comparison was based on the design of a population of 4000 which is typical population served in a medium sized urban centre in Uganda, a per capita flow and BOD contribution of 100 litres/day and 40g/day respectively, and effluent concentration of 30 mg BOD /l. Land area requirements for the two treatment systems were within the same range. The total annual costs of US $11,407 and US $ 14,483 were calculated for constructed wetland and waste stabilisation pond systems, respectively. The annual recurrent costs for both systems were however similar, about US $ 4,000. The deduction from this comparison is that constructed wetlands can be established competitively with waste stabilisation ponds in these environments.

Wetlands posses additional non-market ecological values which are of immense significance. The concept of use and non-use ecological values was used for additional evaluation of wetlands and other treatment systems. Some of these values included fisheries, fuel and building wood/stems, bio-diversity and scientific study. A ranking method applied to compare the significance of these values in constructed wetlands and waste stabilisation pond systems showed more positive gains in constructed wetland systems. The monetary equivalent of these values, though not quantified from the data of Kirinya, is projected to be high. Nuisance related consequences of the two systems, particularly, with regard to mosquitoes and other biting insects was higher in the wetland systems than in the stabilisation ponds. Special attention is necessary at the design and construction stages in order to minimise these effects.

The final conclusion drawn from this evaluation and from other previous performance assessment is that application of constructed wetlands in Uganda and the sub-region, can be considered both technically and economically a viable option for municipal wastewater treatment.

This capacity of wetlands to treat wastewater as demonstrated in the relatively bigger Kirinya pilot wetlands was further investigated at a single household level under similar tropical environments. There are presently many different low-cost onsite sanitation systems in use in different countries. These systems, although effective in reducing the risk from pathogens, are potential non-point pollution sources for groundwater or surface water. This risk is particularly high if they are built in areas with poor soil percolation rates, high water table and heavy rains.

The household wetland in this study was located in a low-density residential area of Jinja municipality in Uganda. The household wetland comprised of two parallel units operated as surface flow wetlands with *Cyperus papyrus* plants as in the Kirinya pilot wetlands. It was designed for a population equivalent of 20, influent BOD concentration into the wetland units of 180 mg/l, assuming 40% BOD is removed by the septic tank and effluent BOD concentration of 30 mg/l. The results obtained over the two-year operation showed an effective removal of COD, BOD and TSS to concentrations that meet the Uganda wastewater discharge criteria. The performance levels were similar to that obtained in the larger Kirinya pilot wetland. However, the nutrient reduction was low and the effluent concentrations exceeded the limits by more than 100 % through out the experimental period. Oxygen limitations imposed by the high shoot density cover minimised the reduction of ammonium. Phosphorus concentration in the wetland influent wastewater was high (15.5 mg/l) compared to the maximum of 6.5 mg/l received at Kirinya. The high concentrations were directly linked to laundry washings estimated to have contributed 0.2 m^3 day^{-1} of the flow. Similarly, low faecal coliform reduction, only 1.13 log units was obtained over the time of measurements. As a result of the high faecal coliform counts in the effluent, its re-use potential was therefore limited. The minimal reduction of faecal coliforms bacteria was associated with the malfunctioning of the septic tank causing faecal overload in the systems. The faecal coliform counts loaded into the wetland were of the same order as raw sewage. Good pre-treatment is an essential requirement for this type of systems to function properly.

The nuisance and risk of malaria infection by the mosquitoes that were breeding in the wetland systems was recognised as a key factor in the design and use of household wetlands in the tropical environments. The surface flow wetland had exposed water surfaces. Therefore it is concluded that this type of wetland systems are not suitable for use in residential areas. A subsurface wetland type is the option recommended for these types of environments and locations.

8.3 Guidelines for design, operation and management

The findings from this study at the Kirinya pilot constructed wetland and the household wetland provided a base for future research and application of the wetland technology for wastewater treatment in the region. The lessons and experience learnt over the time of experimentation and operation of these units was used to generate recommended guidelines for future design, operation and management of these systems in similar environments. This particularly covers aspects of design pre-requisites, objective and purpose of the constructed wetland, loading regimes, aquatic plant management strategies and construction provisions.

8.3.1 Wetland design and operation

Pre-requisite

The Kirinya pilot wetland units were loaded with four days pre-settled sewage as its influent. Effective removal of organic matter and suspended solids from the vegetated *Cyperus papyrus* and *Phragmites mauritianus* wetland was attained. On the other hand, in the household wetland, which was characterised by poor pre-treatment, reduced efficiency, was obtained. Therefore, for any wetland design, a settling basin with a capacity to remove over 40% of BOD from the incoming raw sewage is essential and has to be included as part of the wetland system.

Purpose

The results obtained from the Kirinya units and the demonstration wetland clearly showed that achieving good and simultaneous reduction of bulk pollutants and nutrients under similar operating conditions was not feasible.

A high reduction of organic matter and suspended solids was achieved when the shoot density cover in the wetland units was both low and high. However, nitrogen removal was low when the shoot density cover was high. Reduction of BOD and TSS was still high at hydraulic loading rates of 10 cm day^{-1} (> 60%). Linear relationship between mass loading and mass removal were still found at mass loading rates of 350 kg COD ha^{-1} day^{-1}, 100 kg BOD ha^{-1} day^{-1} and 250 kg TSS ha^{-1} day^{-1}. In contrast, at this hydraulic loading, ammonium reduction was negligible. Optimal removal of both the bulk pollutants and nutrients in the same operating conditions could not be achieved. It is therefore recommended that the intended objective of treatment be clearly defined at the wetland design stage in order to avoid over expectations of treatment performance of the systems.

In the study, by making some changes in the structure of the wetland, enhanced removal of nitrogen was attained even at high load rates. Nitrate concentration in the influent wastewater was low but ammonium was very high in comparison. The modification was aimed at increasing nitrate concentration. This was achieved through the creation of several small open zones with surface areas of between 5-10 m^2 in alternation with the vegetated zones of same size. This structural modification created suitable pH and oxygen concentrations that stimulated nitrogen removal by simultaneous nitrification-denitrification. The ponds with larger open surface areas on the other promoted ammonia loss by volatilisation process. This alternating structural arrangement of a wetland is thus recommended to be an integral part of a surface flow wetland designed for reduction of ammonium.

Wetland type
Surface flow wetlands were applied in this study at Kirinya and at the demonstration site. Their applicability in a domestic or near a residential area was found unsuitable not because of treatment performance but due to risk of malaria infection and nuisance from the mosquitoes breeding in the exposed water. The tropical climate favours mosquito growth and therefore for such critical areas, sub-surface flow wetland types are recommended.

Feeding regimes
Discontinuous and continuous wastewater loading format were both applied in this study. The treatment performance in both cases was more or less dictated by the operating conditions and status of the plants. In Uganda and in the region at large, it is envisaged that these systems may be applied at institutions with or without adequate supply of water. Both types of loading cycles are therefore recommended and may be applied interchangeably.

8.3.2 Optimising nutrient removal and vegetation management

The growth of plants in aquatic environments in the tropics is characterised by a whole year pattern, which are not subject to the climatic fluctuations as in the temperate zones. As a result changes in the plant biomass yields and nutrient uptake obtained in both plant species used was linked to the exponential and stationery growth phases, which were different. Nutrient removal efficiency from the papyrus units increased by 15 % N and 10% P and in *Phragmites* by 57% N and 37% P through harvesting the shoot biomass at 8^{th} (papyrus) and 10^{th} (*Phragmites*) month after planting, at the exponential growth stage. Because of the large nutrient reservoir in the rooting biomass of papyrus, faster re-growth was only realised in papyrus after the initial harvest but declined in subsequent times. Hence, complete and repetitive harvesting could compromise the plant vitality. Therefore for maximising on nutrient uptake removal route, it is recommended that plants could be introduced in the wetland in a staggered way to allow for differential growth patterns and consequently, an alternate harvesting regime.

8.3.3 Wetland Construction

The use of concrete and extensive brickwork as at Kirinya wetlands, is not recommended for general applications. Instead, the sides and bottom of the wetland can be lined with a non-permeable layer after clay compacting. This is essential to prevent leakage of the wastewater into the ground water where it may cause negative effects. Provision of walkways for easy plant harvesting as well access points for monitoring are essential and should be provided for at the design stage.

8.4 Limitations and general recommendations

This study approach was aimed at stimulating investigations on the use of the constructed wetland technology in the region. The capacity of the system to remove the common pollutants present in municipal wastewater namely organic matter, suspended solids, nutrients and faecal coliforms was investigated under similar operating conditions. This methodology had limitations since it was based on the premise that the conditions were suitable for removing all pollutants at the same time but was not the case. In-depth study of the processes for all the pollutants could not be done under the framework of study. It is thus recommended that future studies should be focussed on determining the performance of the constructed wetland with regard to specific pollutant(s) under conditions that suit it or their removal only. Further investigations are also recommended using other types of indigenous aquatic macrophytes such as *Typha* spp. In addition, research based on sub surface wetland should be targeted at since it has a reduced risk and nuisance from mosquitoes that was one weakness of surface flow wetlands under the prevailing climate.

Finally, this study has shown that constructed wetlands are cost effective and have a capacity for wide application in the region. Attention of policy makers in governments, the private sector and academics is therefore drawn to this potential. It is the cheaper option that may be the answer to the problem of municipal wastewater treatment dogging down many urban centres and settlements in the region.

Curriculum Vitae

Tom Okia Okurut was born on 6th January 1960 in Ngora Kumi District Uganda. He finished his ordinary level education at Gulu High School in 1977 and his advanced level education at Teso College, Aloet in 1979. In July 1980, he joined Makerere University in Uganda from where he was awarded a Bachelor of Science Degree in Chemistry in January 1984. From October 1983 to September 1987, he worked as a Teaching Assistant in the Department of Chemistry Makerere University and as an Analytical Chemist with the Geological Survey Department. In October 1987, he obtained a UNESCO scholarship under the African National Scientific and Technical Institutions (ANSTI) program to pursue a Master of Science degree in Physical Chemistry at the University of Ibadan, Ibadan Nigeria which he was awarded in December 1988. In March 1989, he was appointed as a Lecturer in the Department of Chemistry, Makerere University. In September 1991, he was awarded a NUFFIC Fellowship to undertake a Diploma Course in Environmental Science and Technology at the International Institute for Infrastructural Hydraulic and Environmental Engineering (IHE-Delft) which he obtained with a distinction in September 1992. He proceeded to pursue a Master of Science Degree in the same field and at the same Institute, which he obtained with a distinction in April 1993. His thesis was entitled ' Characterization of Wastewater Purification by *Cyperus papyrus* Floating in a Segmented Channel'. In April 1995 he started to pursue a PhD program at IHE Delft in collaboration with Wageningen Agricultural University and with the financial support provided by the Netherlands Institute for Inland Water Management and Wastewater Treatment (RIZA). The research was carried out in Uganda using a pilot constructed wetland and was aimed at determining the technical and economic viability of constructed wetland technology in tropical environments. Since 1994, he has worked as a Water Quality Manager at the National Water and Sewerage Corporation (NWSC). He is the Project Coordinator for the Industrial and Municipal Waste Management Component of the Lake Victoria Environmental Management Project (LVEMP) for Uganda. He is also a gazetted Environmental Inspector for Water Resources sector by the National Environmental Management Authority (NEMA).