AN INVESTIGATION INTO THE TECHNICAL FEASIBILITY OF USING VEGETATED SUBMERGED BED CONSTRUCTED WETLANDS FOR THE TREATMENT OF LANDFILL LEACHATE

By

Jonathan Simon Olufsen

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ABSTRACT

Landfill leachate treatment in South Africa is still in its early stages; research has been conducted but primarily at pilot scale level. Current legislation in South Africa does not prohibit the discharge of landfill leachate into the sewer line, despite the high risk of methane explosions and corrosion of the sewer pipes. Thus, to date, the off site channelling of landfill leachate into the sewer lines for further dilution in municipal wastewater treatment plants is the most common practice. Due to the development of stricter environmental regulations, the design of *sustainable* landfills is leading to the 'treatment at source' concept. Increasing public pressure is also forcing new landfills to be situated in remote areas where there is no avaliable sewer line to discharge into and 'treatment at source' will be required. Due to these developments, coupled with the lack of full scale leachate treatment experience in South Africa, Durban Solid Waste (The waste service unit of the Durban metropolitan), in an attempt to develop the knowledge and practical experience required for leachate treatment, undertook a research project to investigate the use of nitrification/denitrification pilot scale sequencing batch reactors (SBR) to treat leachate from the Bisasar Road and Mariannhill Landfills. The successful completion of the trials proved that the full removal of nitrogen compounds could be easily achieved, under South African climatic conditions, in a single sludge SBR system. The system was found to be simple to operate and required low maintenance. However, the final effluent required further treatment before it could meet the general discharge standards into natural watercourses. Being South Africa, a 'low gross income' country, it became necessary to consider an appropriate, cost effective and technically feasible 'polishing' treatment system. It was decided that a pilot scale treatment trial, using vegetated submerged bed constructed wetlands, be undertaken to assess the applicability and feasibility of such a passive system for the 'polishing' of the effluent from the pilot scale sequencing batch reactors. The wetland systems were found to be affected by many interrelating climatic factors. The trials concluded that the wetlands could not achieve the required discharge standards, in terms of concentration. However, it also showed that the effluent organics posed no oxygen demand or toxic threat to a receiving environment. The trials showed the ability of the wetlands to behave as mass removal systems, which could achieve the required mass removal efficiency in terms of mass output per day.

DEDICATION

This dissertation is dedicated to my beloved son Joshua

PREFACE

The work presented in this dissertation was carried out under the supervision of Doctor Cristina Trois and Professor Deneys Schreiner of the School of Civil Engineering, Surveying and Construction, University of Natal, Durban, South Africa. This research dissertation represents, unless specifically indicated to the contrary in the text, my own work that is submitted in fulfilment of the academic requirements for the degree of Master of Science in Engineering and has not been submitted in part, or in whole to any other University.

Research into the primary treatment of the Bisasar Road landfill leachate using a pilot scale single sludge nitrification/denitrification sequencing batch reactor was carried out by Lindsay Julian Strachan of Durban Solid Waste in collaboration with EnvirosAspinwall of the UK. The trials proved to be successful. However, with the increasingly stringent discharge standards imposed by the Department of Water Affairs and Forestry, the final effluent from the reactors required further treatment before it could be discharged into a natural watercourse. This research was conducted in order to assess the technical feasibility of using vegetated submerged bed constructed wetlands as a final 'polishing' step before discharge. A collaboration between the School of Civil Engineering, Surveying and Construction and Durban Solid Waste was formed and funded by Durban Solid Waste in order to conduct the research.

The research included the design, construction, operation, maintenance and monitoring of a pilot scale vegetated submerged bed constructed wetland system, set up at the Bisasar Road Landfill site. It also included a series of local and international site visits to full-scale constructed wetlands.

Jonathan Simon Olufsen
March 2003

.... Date: ....

Doctor Cristina Trois
Research Supervisor

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ν	kinematic viscosity, m^2/d
μ	viscosity of water, kg/ms
K_{BOD}	BOD temperature dependant first order rate constant, 1/d or m/d
ρ_s	density of solids, kg/m^3
ΔU_S	change in soil cover moisture storage
ΔU_W	change in moisture content of the refuse components
	0.4, for sparse low standing vegetation, y>0.4m
	1.0-2.0 for depths less than 20cm
	1.6, for moderately dense vegetation, y = 0.3m
	6.4, for very dense vegetation and litter layer, y<0.3m
a	resistance factor, $sm^{1/6}$
A_{BOD}	fraction of BOD not removed by settling at the head of the CW
A_c	cross-sectional area perpendicular to flow, m
SLR	surface area loading rate, $g/m^2/d$
A_s	wetland top surface area, m^2
A_V	specific surface area avaliable for microbial activity. m^2/m^3
BOD_x	x day biochemical oxygen demand, mg/l
BOD_e	effluent BOD_5 concentration, mg/l
BOD_I	influent BOD_5 concentration, mg/l
C_D	drag coefficient
C_H	humidity coefficient
C_I	concentration of pollutant in the influent, g/m^3
C_o	concentration of pollutant in the effluent, g/m^3
C_P	concentration of pollutant in rain, g/m^3
C_S	sunshine coefficient
C_T	temperature coefficient
C_W	wind coefficient
C_Z	sample z
D	average particle diameter of the media, m
d	suspended solid particle diameter, m
D_c	dispersion coefficient, m^2/d
ρ	density of water, kg/m^3

N	dimensionless distribution factors for laminar and turbulent flow
EP	pan evaporation, mm/d
ET	evapotranspiration rate, mm/d
f	friction factor related to the coefficient of drag around the particles
K	first order rate constant, 1/d
g	acceleration of gravity, m^2/d
H	relative humidity, percent
h_f	friction loss through bed, m
HLR	hydraulic loading rate, m/d
HRT	hydraulic residence time, d
H_{we}	elevation of the water surface, m
k	hydraulic conductivity, m/d
L	length of wetland cell, m
L_G	leachate generated
m	0.33-0.5 for depths greater than 20cm
n	porosity
n_M	Manning's resistance coefficient, $s/m^{1/3}$
n_o	number of observations
P	precipitation rate, mm/d
Q	the average flow through the wetland, m^3/d
Q_b	bank loss rate, m^3/d
Q_c	catchment runoff rate, m^3/d
Q_{gw}	infiltration to groundwater, m^3/d
Q_i	input polluted water flow rate, m^3/d
Q_o	output polluted water flow rate, m^3/d
Q_R	flow rate of rainfall through the system, m^3/d
Q_{sm}	snowmelt rate, m^3/d
$\overline{R}$	average range
R	surface run-off
Re	Reynolds number
S	percentage of possible sunshine, percent
S_0	hydraulic gradient or slope of water surface, m/m
S_D	standard deviation for duplicates
S_M	standard deviation for multiple analyses of a single sample

T	temperature, °C
t	time, d
T	turbulence factor, d^2/m^2
u	superficial velocity, m/d
V	water storage in wetland, m^3
v	actual velocity, m/d
VLR	volumetric loading rate, $g/m^3/d$
V_S	settling velocity of the suspended solid particle, m/s
W	width of wetland cell, m
W_i	wind speed, m/s
x	longitudinal distance, m
y	depth of water in the wetland cell, m
y_f	depth of filter, m

LIST OF ABBREVIATIONS

SLR	Surface area loading rate
UND	University of Natal Durban
Std. Dev.	Standard deviation
Min.	Minimum
Max.	Maximum
#	Number
HDPE	High density polyethylene
BRE	Control effluent flow
BR Cl^-	Estimated effluent chloride concentration for control cell
RE	*Leersia hexandra* effluent flow
RE Cl^-	Estimated effluent chloride concentration for the *Leersia hexandra* cell
BLE	*Vetiver zizanioides* effluent flow
BL Cl^-	Estimated effluent chloride concentration for the *Vetiver zizanioides* cell
OE	*Phragmites australis* effluent flow
O Cl^-	Estimated effluent chloride concentration for the *Phragmites australis* cell
BATEA	Best available technology economically achievable
BATNEEC	Best avaliable technology not entailing excessive costs
BOD_x	x Day biochemical oxygen demand
$C_5H_7O_2N$	Formulae for organic biomass
COD	Chemical oxygen demand
CSIR	Center for Scientific and Industrial Research
CW	Constructed wetlands
DC	Dissolved carbon
DIC	Dissolved inorganic carbon
DO	Dissolved oxygen
DOC	Dissolved organic carbon
DSW	Durban Solid Waste
DWAF	Department of Water Affairs and Forestry
EC/EWPCA	European Community/European Water Pollution Control Association
EP	Class-A-pan evaporation
EPA	Environmental Protection Agency (USA)
ET	Evapotranspiration

FFS	Fixed filterable solids
FS	Filterable solids
FSS	Fixed suspended solids
FWS	Free water surface flow
GLB^+	Large general waste landfill with significant leachate generation
HLR	Hydraulic loading rate
HRT	Hydraulic residence time
MLSS	Mixed liquor suspended solids
MLVSS	Mixed liquor volatile suspended solids
MSW	Municipal solid waste
NFB	Nitrification filter bed
Norg	Organic nitrogen
NPDES	National pollution discharge elimination system
PC	Particulate carbon
RBC	Rotating biological contractor
RO	Reverse osmosis
SBR	Sequencing batch reactor
SF	Sub-surface flow
SS	Suspended solids
TDS	Total dissolved solids
TFS	Total fixed solids
TKN	Total Kjeldahl nitrogen
TOC	Total organic carbon
TS	Total Solids
TSS	Total suspended solids
TVS	Total volatile solids
UES	Uniform effluent standards
UK	United Kingdom
US	United states
USA	United States of America
UV	Ultra Violet
VF	Vertical flow
VFA	Volatile fatty acids
VFS	Volatile filterable solids

VLR	Volumetric loading rate
VSB	Vegetated submerged bed
VSS	Volatile suspended solids

CHAPTER 1

"The relationship between modern man and the planet....
has been that not of symbiotic partners,
but of the tapeworm and the dog,
of the fungus and the blighted potato."
(Aldous Huxley, Ape and Essence)

1 INTRODUCTION

Under natural conditions, Earth's life forms live in equilibrium with their environment and the available resources govern the numbers and activities of each species. Hominids, however, have developed the ability to gather and process resources from beyond their immediate surroundings, which has allowed their population to thrive and flourish beyond natural constraints. The waste products generated and released into the biosphere by these increasing populations have upset the natural equilibrium. When considering the age and life of the Earth compared to the age of the human species, it may be said that humans are 'arrogant' in believing that we could destroy the Earth. The reality of the matter is that we are destroying our surrounding environment on which we depend on for survival. By understanding the Earth's track record and its ability to persevere through dominant species extinction, it has to be said that the only entity we are destroying is ourselves and unless we protect our natural environment on which we depend without being selfish and blinded by the greed of money, we will become another part of the earth's history lessons.

1.1 Modern landfills and the environmental concerns

Through the ages of solid waste disposal by land emplacement, the common practices and the terminology have changed: starting with the uncontrolled 'dumps' where the solid waste was placed directly on top of the virgin soil, to modern day engineered landfills equipped with drainage, liner and capping systems, to provide reasonable control of landfill emissions. Landfilling techniques have developed, through scientific research and sometimes catastrophic errors, giving us a better understanding of the landfill emissions and their environmental impacts. This research has focused on the treatment of landfill leachate from a municipal solid waste landfill.

1.2 The development of landfill leachate treatment systems

Treatment technologies currently available for landfill leachates have been derived from extensively used sewage wastewater treatment systems. Certain manipulation of the processes to accommodate for the variations in leachate strength and characteristics are required. The main criteria for the selection of a treatment system are the treatment objectives, characteristics of the leachate, economic feasibility and technical feasibility. To date, leachate treatment systems have ranged from the simplest evaporation ponds to the highly complex reverse osmosis and ozonation systems, the success of which is based on site-specific conditions. In the case of South Africa, the most common treatment practice has been and still is, despite the high risk of methane explosions and corrosion of the sewer pipes, the channeling of leachate into a nearby sewer line without any pretreatment. Due to the development of stricter environmental regulations the design of *sustainable* landfills is leading to 'the treatment at source concept'. Increasing public pressure is also forcing new landfills to be sited in remote locations where there is no available sewer line for discharge and the need for on site treatment for the discharge back into the natural environment would become necessary.

2 OBJECTIVES AND GENERAL OVERVIEW

In an attempt to develop the knowledge and practical experience required for leachate treatment, in South Africa, a collaboration between Durban Solid Waste and EnvirosAspinwall (UK) was formed and research into the use of nitrification/denitrification pilot scale sequencing batch reactors (SBR) to treat leachate from the Bisasar Road and Mariannhill Landfills was conducted. The successful completion of the trials proved that the full removal of nitrogen compounds could be easily achieved, under South African climatic conditions, in a single sludge SBR system (Strachan, 1999 and Olufsen, 1999) (Chapter 4, pp. 86-90). The system was found to be simple to operate and required low maintenance, however, the final effluent required further 'polishing' of certain contaminants, especially residual COD, before it could meet the general discharge standards (Table 2.2) for a natural watercourse (e.g. Umgeni River). South Africa is a 'low gross income' country and it was necessary to consider an appropriate, cost effective and technically feasible 'polishing' treatment system. It was decided that a pilot scale treatment trial using vegetated submerged bed constructed wetlands be undertaken to assess the applicability and feasibility of such a passive system for the 'polishing' of the effluent from the pilot scale sequencing batch reactors.

The preliminary objectives were to:

- Design and construct a set of pilot scale vegetated submerged bed constructed wetlands at the Bisasar Road Landfill site.
- Operate and monitor the pilot scale vegetated submerged bed constructed wetland system and maintain and operate the pilot scale sequencing batch reactor system in order to obtain an influent for the wetlands.

With the main objective to:

- Assess the technical feasibility of such a system in terms of meeting current discharge standards with the future possibility of a full-scale system and to investigate the influence of various plant species on the treatment performance.

Chapter 2 focuses on landfill leachate production and generation and includes the environmental impacts of specific pollutants. It also briefly covers the legislative framework for the discharge of polluted waters into a natural watercourse and the different types of treatment technologies available. Chapter 3 is a continuation of the literature review which focuses solely on constructed wetlands. Chapter 4 includes full scale case studies from around South Africa and the United Kingdom.

Chapter 5 is the start of the experimental part of the research. It covers the design and construction of the pilot scale vegetated submerged constructed wetlands. It is followed by Chapter 6, which covers the operation and maintenance of the pilot scale system and includes the aspects of the experimental procedures.

Chapter 7 is the core of the experimental part of the research. It includes the processed results of the vegetated submerged system and the treatability trials. Chapter 8 covers the conclusions and recommendations.

CHAPTER 2

2.1 Landfill leachate

Landfill leachate is the polluted water that emanates from a landfill site. The Department of Water Affairs and Forestry (DWAF) of South Africa define landfill leachate as: "An aqueous solution with a high pollution potential, arising when water is permitted to percolate through decomposing waste. It contains final and intermediate products of decomposition, various solutes and waste residues. It may also contain carcinogens and/or pathogens" (DWAF, 1998). Landfill leachate is, in other words, a highly complex wastewater formed by water percolating through the decomposing waste body and accumulating particulate and dissolved contaminants from the waste. In most cases the landfill leachate is highly polluted and may, through subsurface migration, contaminate groundwater resources and down stream surface waters (Christensen, 1989 and Robinson et al, 1992). Migrating leachate from a landfill site also has the potential to transport explosive landfill gas outside the boundaries of the site (Robinson et al, 1992). Landfill leachates may contain dissolved methane (CH_4) in concentrations of 10 to 15 mg/l, where a concentration of dissolved methane as low as 1.4 mg/l is capable of producing explosive atmospheres (Robinson, 2001a).

2.2 Landfill leachate generation

In order for leachate to be generated certain hydrological requirements need to be met. There, of course, needs to be a net surplus of water when taking precipitation, evaporation and surface run-off into account, but there is also another requirement which needs to be met; the waste and the soil cover material have a moisture storage capacity (field capacity) which needs to be exceeded before leachate will emanate from the waste body (Knox, 1991; Blakey, 1992; Blight et al, 1992 and Qasim and Chiang, 1994).

The waste body may be regarded as a large bioreactor in which many biodegradation processes take place (Christensen and Kjeldsen, 1989, Robinson, 1989 and Robinson, 1996). Experiences from full-scale landfills and laboratory studies have led to a theoretically idealised sequence for the biodegradational processes occurring in the waste body. The sequence involves five distinct phases and the leachate produced during these phases is characteristic to its production phase (Christensen and Kjeldsen, 1989 and Robinson, 1989).

The phase of decomposition under which methane is formed is of particular interest because it lasts longer than the other phases and provides for the decomposition of most of the decomposable waste. There will be an initial period after the refuse is placed during which methane will not be produced or will be produced in small amounts. Depending on the characteristics of the landfill, such as oxygen access, temperature, moisture content, moisture access and refuse decomposition, this lag phase may continue for a period ranging from a few weeks to many years even decades (Robinson, 1989). In a humid climate the lag phase is generally of three to nine months (Ham, 1988 and Bowers, 1999) and in a dry climate a landfill may never attain the methanogenic decomposition phase (Ham, 1988).

2.3 Impacts on the receiving environment

Leachate that emanates from a municipal solid waste (MSW) landfill site is in most cases heavily polluted and may be just as toxic as leachates from landfills in which residential and hazardous wastes were codisposed (Christensen, 1989 and Qasim and Chiang, 1994). This leachate may through subsurface migration, cause extensive pollution of the surrounding receiving environment. Ground water pollution is of major concern at landfills demanding extensive control systems and monitoring. Subsurface migration of landfill leachate is, however, slow and visual impacts on the surrounding environment may not be possible for decades making remediation of the areas polluted extremely costly (Christensen, 1989).

The impacts of MSW landfill leachate on the receiving environment are highly complex and cannot be related to one single factor. This section is not intended to cover every possible cause but to highlight the most common and serious, which require attention when considering a leachate treatment process in order to ensure that the leachate treatment system selected will remove the respective pollutants before subsequent discharge into the receiving environment.

2.3.1 Nutrients and overenrichment

Nutrients are of vital importance, since they are the essential elements for growth and reproduction of plants and animals. There is a large number of minerals and trace elements that may be considered as nutrients but the most important in terms of impacts are: carbon, nitrogen and phosphorus. Overenrichment with nitrogen, phosphorus and oxygen depletion due to carbon overloading, have adverse consequences on the receiving environment (Peavy et al, 1985 and Reed et al, 1995).

2.3.1.1 Nitrogen

Nitrogen compounds in landfill leachate occur in a variety of organic and inorganic forms. The most polluting inorganic nitrogen forms are: ammoniacal nitrogen (NH_4-N), nitrite nitrogen (NO_2-N) and nitrate nitrogen (NO_3-N).

Ammoniacal nitrogen

Aqueous ammoniacal nitrogen exists as either the ammonium ion (NH_4^+, ionised ammonia) or as ammonia (NH_3, unionised ammonia), depending on temperature and pH (Figure 2.1), in agreement with the equilibrium equation 2.1 (Peavy et al, 1985, Tchobanoglous and Burton, 1991, Kadlec and Knight, 1996 and Hammer and Hammer, 2001):

$$NH_3 \uparrow + H_2O \Leftrightarrow NH_4^+ + OH^- \qquad (2.1)$$

Under high temperatures and basic pH (pH>7) the equilibrium is shifted to the left, where unionised ammonia becomes the most predominant species, conversely at low temperatures and acidic pH (pH<7), ionised ammonia becomes the predominant species (Kadlec and Knight, 1996). Ammoniacal nitrogen in landfill leachates can range from 1 to 3610 mg/l (Andreottola and Cannas, 1992 and Department of the Environment (UK), 1995), which may be readily oxidised in natural waters, resulting in an oxygen demand on the natural water (± 4.3 gO_2/gNH_4-N). Unionised ammonia is toxic to many aquatic species (especially fish) at very low concentrations, typically at concentrations greater than 0.2 mg/l (Reed et al, 1995, Kadlec and Knight, 1996 and Hammer and Hammer, 2001).

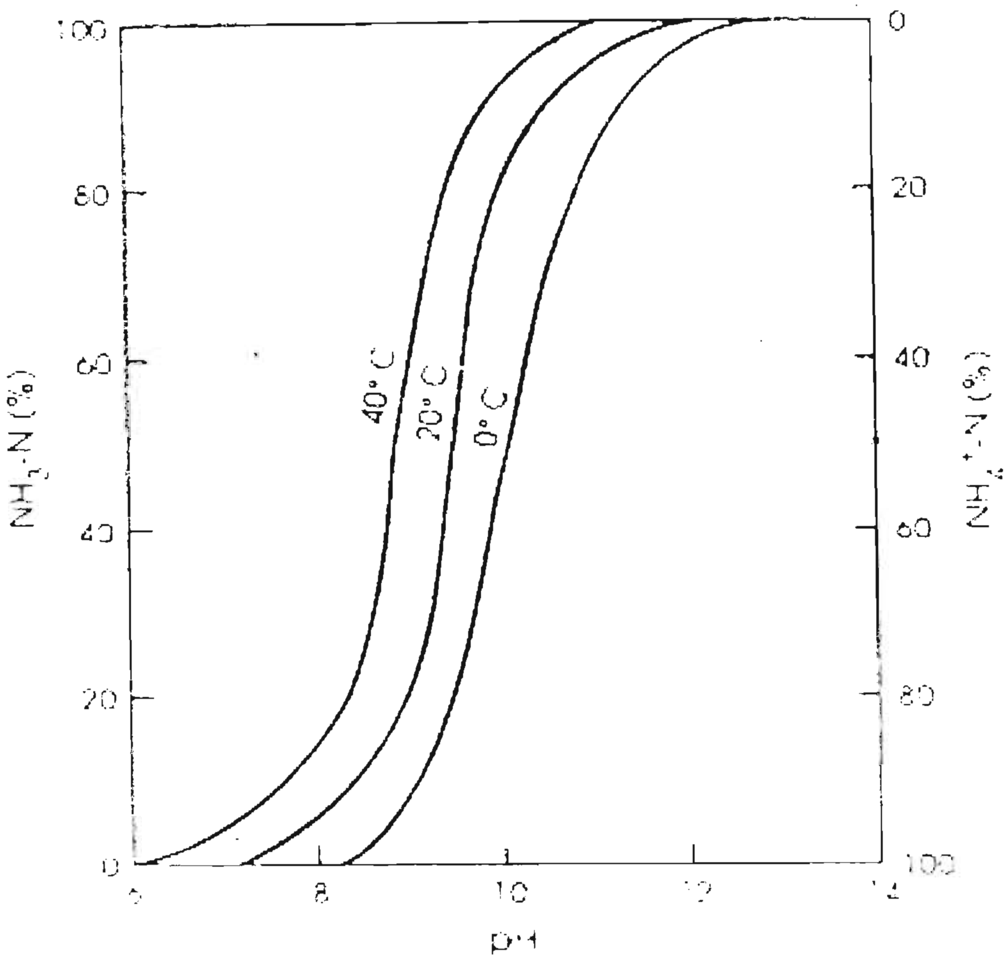


Figure 2.1: Effect on pH and water temperature on the fraction of total ammonia in the unionised and ionised forms, from Kadlec and Knight (1996).

Nitrate and nitrite nitrogen

Under aerobic conditions, ammoniacal nitrogen is oxidised to nitrite and then to nitrate, nitrite is not chemically stable and is usually found at trace levels in polluted waters (Ranging from 0 to 25 mg/l in landfill leachates (Andreottola and Cannas, 1992 and Department of the Environment, 1995), nitrate, however, is chemically stable and persists unchanged in polluted waters (Ranging from 0.1 to 50 mg/l in landfill leachates (Andreottola and Cannas, 1992 and Department of the Environment, 1995) unless biologically transformed (Kadlec and Knight, 1996). Nitrite nitrogen may be analytically determined by colorimetric techniques or by ion chromatography (Clesceri et al, 1989 and Hammer and Hammer, 2001). Nitrate and nitrite nitrogen causes nitrate poisoning (methaemoglobinemia) in infant animals and humans. Methaemoglobinemia can be extremely dangerous and even fatal. Nitrate nitrogen is converted by nitrate nitrogen reducing bacteria within the lower acidity of the infant's intestinal tract to nitrite nitrogen, which is the actual etiological agent of methaemoglobinemia. The nitrate nitrogen is absorbed into the blood stream where it replaces oxygen in the blood complex as it has a greater affinity for haemoglobin than does oxygen. A deficiency of oxygen occurs and, in the extreme case, it can lead to suffocation (Peavy et al, 1985; Clesceri et al, 1989 and Kadlec and knight, 1996). Nitrous acid, which is formed from nitrite nitrogen in acidic solution, may react with secondary amines to produce nitrosamines, which are known to be carcinogens (Clesceri et al, 1989). Nitrate nitrogen in excessive concentrations in surface waters leads to eutrophication (Kadlec and Knight, 1996 and Hammer and Hammer, 2001). Eutrophication is the accelerated production of plant life due to excessive nutrient inputs. In the presence of sunlight, algae metabolize the nutrients (Nitrate nitrogen and phosphorous) while obtaining energy from the sunlight. The algae multiply very quickly, covering the entire water surface, preventing the penetration of sunlight, which is essential for other aquatic species. When sunlight is available and the algae metabolize the nutrients, oxygen is released as a waste product, however if sunlight is not available algae catabolize stored food for energy and use oxygen in the process, thus creating an oxygen demand on the natural water. The oxygen demand in natural waters that are nutrient rich may have an adverse effect on the aquatic environment. In addition to accelerated growth, algae also die quickly. The high numbers combining with the quick death leads to an accumulation on the bed of the aquatic system. The material formed is called 'necron mud', which continues to consume oxygen slowly as it decays. This further reduces the oxygen level and fills up the aquatic system, making it shallower. Animals that are adapted to the original aquatic system depth also begin to stress. Eventually the aquatic system will be stressed to the point where all the aquatic species, except for algae, disappear from the natural water (Peavy et al, 1985 and http://www.thegeographyportal.net). Other adverse effects on water quality due to algae include taste and odour problems (Peavy et al, 1985).

Organic nitrogen

Organic nitrogen (Norg) is defined as organically bound nitrogen, upon the death of plants, animals and food wastes, the organic matter is broken down through bacterial decomposition, to form organic nitrogen products, such as amino acids, urea and uric acid, and purines and pyrimidines (Peavy et al, 1985; Clesceri et al, 1989 and Kadlec and Knight, 1996). The organic nitrogen is determined by digestion of the organic matter, to form ammoniacal nitrogen, which is then analytically measured (Hammer and Hammer, 2001). The sum of both ammoniacal nitrogen and organic nitrogen in the polluted water sample is referred to as total Kjeldahl nitrogen (TKN). Organic nitrogen in landfill leachates range from 1 to 2000 mg/l with total nitrogen ranging from 50 to 5000 mg/l (Andreottola and Cannas, 1992). Techniques are available to directly measure organic nitrogen and to calculate it from the difference of the determined TKN and ammoniacal nitrogen. Ammonification is the biological transformation of organic nitrogen to ammoniacal nitrogen, which may occur under both aerobic and anaerobic conditions, releasing ammoniacal nitrogen into the receiving environment (Kadlec and Knight, 1996 and EPA, 2000).

2.3.1.2 Phosphorus

Phosphates do not pose a direct health threat to humans or other organisms; however, they can indirectly affect water quality. Phosphate, like nitrate nitrogen, is a limiting nutrient in surface waters and when the phosphate concentration becomes excessive eutrophication can occur. Total phosphorus in landfill leachate ranges from 0.1 to 30 mg/l (Andreottola and Cannas, 1992 and Department of the Environment (UK), 1995).

2.3.1.3 Carbon

The organic fraction in landfill leachates is most commonly measured in terms of chemical oxygen demand (COD) and biochemical oxygen demand (BOD), with COD ranging from 150 to 150000 mg/l, BOD_5 ranging from 100 to 90000 mg/l and BOD_{20} ranging from 110 to 125000 mg/l (Andreottola and Cannas, 1992 and Department of the Environment (UK), 1995). The organic fraction in landfill leachates is characterised into biodegradable and non-biodegradable organics, COD gives a measure of the sum of both and BOD gives a measure of only the biodegradable fraction. The ratio of BOD to COD for a landfill leachate will tend to decrease over the age of a landfill as the less biodegradable humic and fulvic fractions become a greater portion of the COD (Andreottola and Cannas, 1992). The potential effects of these less biodegradable organics on the receiving environment cannot be predicted, and there is still discussion on whether it is sensible to even remove these substances before discharge into the natural environment (Christensen et al, 1998). The biodegradable fraction constitutes the main polluting potential, especially in young landfills where BOD to COD ratios can reach values of 0.58 compared to methanogenic leachates of 0.06 (Ehrig, 1989). Biodegradable organics that are discharged or migrate uncontrolled into a natural receiving environment, containing dissolved oxygen, undergo aerobic metabolic processes that convert the organics, using the dissolved oxygen as the terminal electron acceptor; the quantity of oxygen required for this, is measured by the BOD test (Peavy et al, 1985). The dissolved oxygen used in this process will have to be replaced by atmospheric reaeration or algae photosynthesis; otherwise anaerobic conditions will develop severely affecting the ecology of the receiving environment (Peavy et al, 1985). Atmospheric reaeration is driven by deficient concentration gradients between the atmosphere and the water body, the equilibrium concentration of oxygen in water is related to temperature and salinity of the water. If the concentration of oxygen in the water body is below the equilibrium concentration, then the flux of oxygen will be from the atmosphere into the water (Peavy et al, 1985). During photosynthesis algae metabolise inorganic compounds, and release oxygen as one of the waste products (Equation 2.2)(Peavy et al, 1985 and EPA, 2000):

$$CO_2 + 2H_2O \xrightarrow{Light} CH_2O + O_2 + H_2O \qquad (2.2)$$

The oxygen released during algae photosynthesis could be available to replenish the deficiency, however, because algae use the waste products of bacterial metabolism, which are transported downstream of the discharge point by the river flow, they usually grow away from the area where oxygen is required (Peavy et al, 1985). The disproportionate algae growths in the presence of excessive nutrients do contribute to eutrophication and the diurnal switch from photosynthetic metabolism, during the day, to endogenous catabolism (Equation 2.3), during the night, contributes to the oxygen demand and may deplete the dissolved oxygen causing anaerobic conditions which can be detrimental to aquatic diversity (Peavy et al, 1985).

$$CH_2O + O_2 \longrightarrow CO_2 + H_2O \qquad (2.3)$$

If the rate of oxygen replenishment is equal to the rate of consumption then the natural system's capacity for self purification will not be exceeded. However, conversely, if the rate of replenishment is exceeded by consumption, the natural system will suffer a shift from equilibrium to an active phase of self purification, following the *ecological response model* (Figure 2.2), proposed in Peavy et al (1985). The model divides the receiving stream into four zones: zone two is referred to as the degradation zone, where the water becomes turbid with sludge deposits and floating debris, oxygen is reduced to approximately forty percent of saturation, system diversity starts to decrease with fish and green algae populations declining, microbial populations start to increase. Zone three, or the 'active decomposition' zone, is characterised by greyish waters, dissolved oxygen levels reaching anaerobic conditions, microbial populations flourish, anaerobes establishing first in the anaerobic portion of zone three, followed by aerobes, due to decomposition activity decreasing slowly down stream and reintroduction of dissolved oxygen followed further downstream by fungi. There may be fly and mosquito larvae; there is no fish life. The recovery zone, zone four, is characterised by a limited amount of fungi and protozoa, rotifiers, crustaceans and algae begin to appear, system diversity begins to increase with more resistant life forms beginning to appear. The water is clearer, and dissolved oxygen levels begin to move back to equilibrium and nitrates are present. Zones one and five, are regarded as the clean water zones where natural stream conditions exist before discharge and after completion of the self-purification process (Peavy et al, 1985).

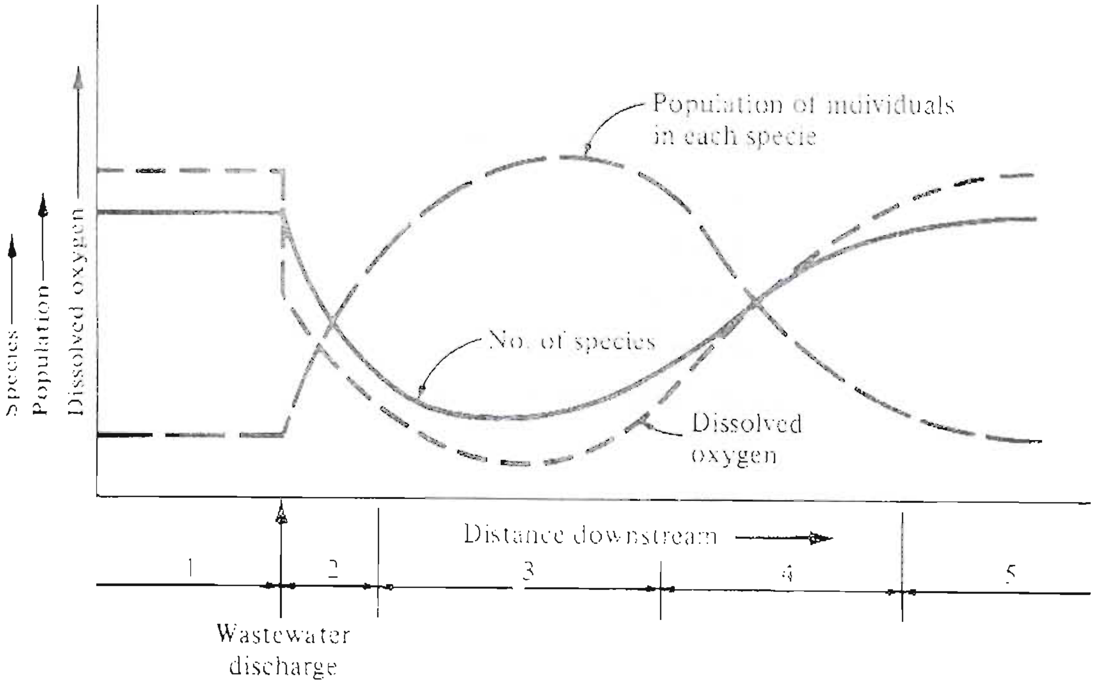


Figure 2.2: Ecological response curve, caused by polluted water discharge into a natural receiver, from Peavy et al (1985).

2.3.2 Alkalinity

The sources of alkalinity in landfill leachate are carbonates, bicarbonates, silicates, borates, ammonia, organic bases, sulfides, and phosphates with concentrations ranging from 300 to 15870 mg/l as $CaCO_3$ (Andreottola and Cannas, 1992; Qasim and Chiang, 1994 and Department of the Environment, 1995). At high concentrations, alkalinity gives a bitter taste to water (Peavy et al, 1985), it may also form precipitates with certain cations in the leachate, contributing to the 'clogging' of leachate collection systems by anaerobic bacterial activity (Cossu, 1998 and Cossu et al, 2000). The high concentrations of alkalinity in landfill leachate constitute a beneficial economical attribute to the treatment of landfill leachate containing high concentrations of ammoniacal nitrogen. Nitrification requires 7.14 mg/l of alkalinity (as $CaCO_3$) per 1 mg/l of ammoniacal nitrogen oxidised (Robinson et al, 1998b). The supply of alkalinity from the leachate may be sufficient enough or may contribute in minimising the input amount of external alkalinity, required to 'buffer' the treatment system against decreases in pH values below the optimal range for nitrification to take place (between pH 8 and 8.5) (Robinson et al, 1998b).

2.3.3 pH value

The pH value of a leachate ranges from acidic conditions (4.5 to 7.5), during the acetic phase of the landfills life to basic conditions (7.5 to 9) during the methanogenic phase (Ehrig, 1989 and Department of the Environment, 1995). The pH value is an indicator for optimal environmental conditions for bacterial growth; these conditions are applicable to both treatment systems (See alkalinity) and natural receiving waters. If a natural system receives polluted water with a pH value below or above its equilibrium range, that system's pH value may be altered, changing environmental conditions for the natural consortium of microbes. The pH value is also an indicator of the solution's ability to mobilise heavy metals (Christensen and Kjeldsen, 1989), which if transported through the migration of leachate into a natural receiver, may have toxic effects on the receiving fauna and flora (Peavy et al, 1985).

2.3.4 Electrical Conductivity

The electrical conductivity, typically ranging from 1000 to 52000 μS/m for landfill leachates (Ham, 1988 and Department of the Environment, 1995), is a measure of the quantity of ionized materials in a polluted water sample (Kadlec and Knight, 1996). Fish species are sensitive to sudden changes in conductivity (Pulles et al, 1996). Conductivity may also be used as a flow tracer; however, ionic salts have been shown to be altered to a certain degree by biological and physical environmental conditions, leading to inaccuracies (Kadlec and Knight, 1996).

2.3.5 Chlorides

Chlorides are not affected by attenuation processes (Qasim and Chiang, 1994) and have been used for dilution estimates in landfill leachate (Andreottola and Cannas, 1992), as well as for estimating the duration of leachate emissions (Ehrig, 1989). Chloride concentrations in landfill leachate typically range from 30 to 5000 mg/l (Andreottola and Cannas, 1992 and Department of the Environment, 1995). High concentrations of chlorides discharged into a natural receiver may have an effect on the receiving water's conductivity, thus affecting the osmotic balance between the aquatic organisms and their surrounding environment, which may be fatal (Pulles et al, 1996). The high concentrations of chlorides in landfill leachates pose a problem for the use of landfill leachate in irrigation (Pulles et al, 1996).

2.3.6 Metals

Heavy metals in MSW landfill leachates have generally been found in low concentrations (Robinson and Gronow, 1998). Heavy metals, however, if present at suitably high concentrations, do pose a potential harmful impact on humans and other organisms (Peavy et al, 1985 and Pulles et al, 1996). Table 2.1 summarises the types of heavy metals found in landfill leachates from landfills classified as large, high waste input rate, relatively dry (Robinson and Gronow, 1998).

Table 2.1: Heavy metals found in large, high waste input rate, relatively dry landfill leachates, from Robinson and Gronow (1998).

Heavy metal	**Concentration (mg/l)**
Cr	0.07 to 0.12
Ni	0.14 to 0.23
Cu	0.07 to 0.07
Zn	0.78 to 6.85
Cd	0.01 to 0.01
Pb	0.13 to 0.30
As	0.009 to 0.010
Hg	<0.0001 to 0.003

Upper limit concentrations represent an acetogenic leachate.
Lower limit concentrations represent a methanogenic leachate.

2.3.7 Toxicity

Toxic substances found in MSW landfill leachates include both organic and inorganic compounds, as heavy metals, pesticides, chlorinated hydrocarbons (Qasim and Chiang, 1994). MSW landfill leachates have been shown to be just as toxic as leachate from landfills practicing co-disposal of residential and hazardous wastes (Qasim and Chiang, 1994). These substances can cause significant and even fatal damage to both humans and other organisms (Hammer and Hammer, 2001).

2.3.8 Colour

The colour of landfill leachate is attributed to tannins, humic acids and humates formed from the decomposition of leaves, weeds and wood, present in the waste body, these substances give the leachate its yellowish-brown hues (Peavy et al, 1985). Reddish leachates may contain iron oxides and brown or blackish leachates may contain manganese oxides, other colours may be attributed to dyes from industrial sources (Peavy et al, 1985). Coloured discharge effluents are not aesthetically acceptable, in fact, 'true colour', colour that is created by dissolved solids after the removal of suspended solids, is not considered 'unsafe', however, the organic compounds responsible for the colour may pose a chlorine demand, decreasing the cost effectiveness of a chlorine disinfectant, if required (Peavy et al, 1985). Colour changes in natural waters may be caused by 'true colour' and 'apparent colour', the latter being colour partly due to suspended solids or turbidity and may influence light sensitive processes such as photosynthesis and may interfere with light penetration (Peavy et al, 1985 and Hammer and Hammer, 2001).

2.3.9 Temperature

Landfill leachate exits the landfill at elevated temperatures, reflecting elevated temperatures within the landfill (30 to 45°C) (Christensen and Kjeldsen, 1989). If the temperature of a natural receiving water body is elevated due the discharge of the warmer leachate; dissolved oxygen levels will decrease, algae growth and microbial metabolism may be accelerated, solubility levels, reaction rates of certain chemicals and the properties of the water, such as viscosity and density will be affected, which may have a 'subtle' effect on planktonic microorganisms in natural water bodies (Peavy et al, 1985).

2.3.10 Solids

Solids found in landfill leachate are usually defined as suspended or filterable, these two fractions are then further subdivided into volatile, usually referred to as the organic fraction, and fixed solids, referred to as the refractory fraction. The sum of all suspended and filterable solids is referred to as the total solids (Figure 2.3) (Tchobanoglous and Burton, 1991).

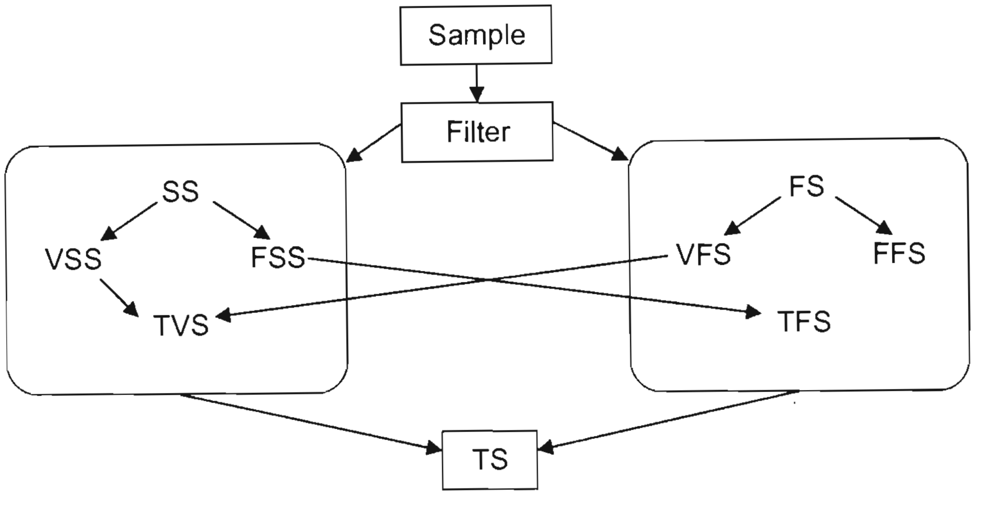


Figure 2.3: Solid fractions, TS = total solids; VSS = volatile suspended solids; FSS = fixed suspended solids; TVS = total volatile solids; TFS = total fixed solids; VFS = volatile filterable solids; FFS = fixed filterable solids; FS = filterable solids; SS = suspended solids.

Suspended solids increase turbidity and hence affect apparent colour, they may also be associated with active microbes, including pathogens (Peavy et al, 1985). Organic suspended solids, specifically volatile suspended solids, may be biologically degraded, resulting in unacceptable by-products and an oxygen demand on the receiving water (Peavy et al, 1985 and EPA, 2000). Total dissolved solids or filterable solids, approximately measured by electrical conductivity, give the leachate its true colour (Peavy et al, 1985).

2.3.11 Pathogens

Pathogens are organisms capable of infecting, or transmitting diseases to humans, they require an animal host for growth and reproduction and many are able to survive and maintain there infectious capabilities outside of the host and in water for significant periods of time (Peavy et al, 1985). These infectious organisms include bacteria, viruses, protozoa and helminths (Peavy et al, 1985 and Hammer and Hammer, 2001). MSW landfills contain a variety of pathogenic sources including animal carcasses, sewage sludges, human and animal faeces and even illegally disposed materials such as medical wastes (Strachan, 1999). Microbiological studies on landfill leachate have shown the presence of pathogenic organisms, however, environmental factors inside the landfill waste body, such as pH value, temperature and primarily the lack of oxygen have also been shown to inhibit the further development of pathogens, once inside the fill, and pathogen populations may even decrease with landfill age (Strachan, 1999). Viruses have only very rarely been reported in landfill leachates, with the rare occurrence of enteroviruses (Andreottola and Cannas, 1992). The only pathogenic fungi observed in landfill leachate have been *Allescheriaboydii*, which may cause *madura foot abscesses.* Parasites such as protozoa, helminths and nematodes may be observed in landfill leachates, due to the presence of animal and human faeces in the fill (Andreottola and Cannas, 1992).

2.4 Water quality standards and legislation

Water quality requirements are based on a known or assumed need. Water quality standards are limits on impurities allowed in water that is intended for a particular use (Nathanson, 2000). Over the years as scientific knowledge has grown water quality standards have evolved, internationally and locally. In the United States of America (US), the Environmental Protection Agency (EPA) has set minimum standards. There are three different types of water quality standards set by the USEPA: stream (or instream) standards, effluent standards and drinking water standards (Peavy et al, 1985; Nathanson, 2000 and Hammer and Hammer, 2001). Figure 2.4 schematically puts the three standards in perspective (Nathanson, 2000). Stream standards are aimed at maintaining the natural water system at as high a quality level as possible and often reflect the beneficial use made of the stream. Drinking standards are set to protect the health of the consumer and are much more stringent than both effluent and instream standards (Peavy et al, 1985; Nathanson, 2000 and Hammer and Hammer, 2001).

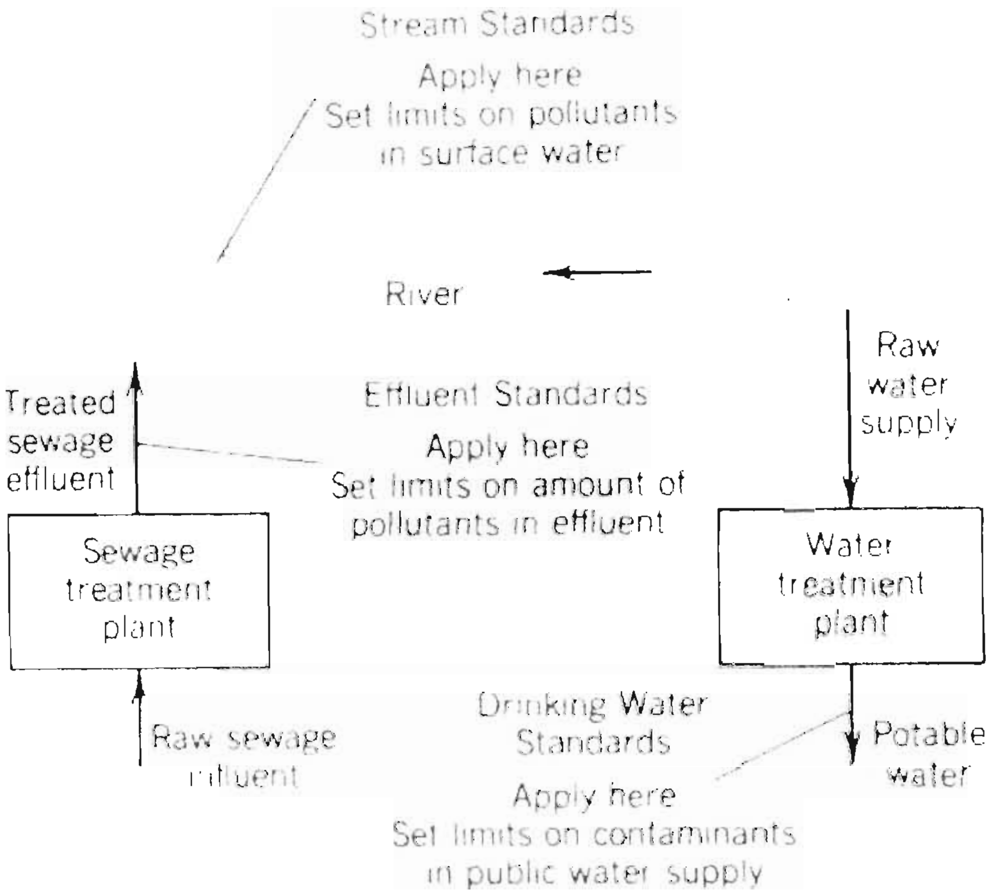


Figure 2.4: Schematic of USEPA water quality standards, from Nathanson (2000).

Effluent water quality standards are set in order to protect the receiving water body and usually take into account its beneficial use, for example more stringent standards will be enforced for receiving waters that are used as drinking water resources and for recreational purposes. In setting effluent discharge limits two aspects need to be satisfied: the first being 'Best Available Technology Economically Achievable' (BATEA) and the second being the state and use of the receiving water body (Hammer and Hammer, 2001). This leads to water quality based standards and technology-based standards. Technology based standards do not guarantee that a treated wastewater will not pollute a receiving water body as they do not take into account the state of the receiving water body. Water quality based standards apply to the waters receiving wastewater discharges (Hammer and Hammer, 2001). In the US the emphasis was initially on the stream standards, which regulates the amount of pollutants in the receiving waters. With the implementation of the National Pollution Discharge Elimination System (NPDES) permit program the emphasis has been redirected and the focus is now on individual discharges and a system of effluent standards for water pollution control has been formulated. This method of regulating pollution is much easier to enforce and it is now possible to determine the responsible polluter. The Uniform Effluent Standards (UES) has been in place for over twenty years in South Africa and is aimed to regulate the input of effluents into the receiving water body, and takes into account Best Available Technology Not Entailing Excessive Costs (BATNEEC) (Pulles et al, 1996). This approach has its drawbacks, it ignores the possibility of existing high background concentrations in the river system, also the ability of the river system to assimilate pollutants (Pulles et al, 1996). The USEPA's response to these drawbacks is evident in their objective, acknowledging the current state of specific water bodies the NPEDS aims not to further pollute these systems and in the future reestablish them as swimmable and fishable water bodies (Nathanson, 2000 and Hammer and Hammer, 2001). Although the application of the UES has led to the decrease in the rate of pollution, deterioration has still continued (Pulles et al, 1996). In 1990 the Department of Water Affairs and Forestry (DWAF) realised that a more advanced approach would be necessary and included the following principals in its approach (Pulles et al, 1996):

- "The desired quality of a water resource is determined by its present and/or intended uses. This quality should be stated as a list of water quality objectives."
- "It is accepted that the water environment has a certain, usually quantifiable, capacity to assimilate pollutants without detriment to predetermined quality objectives."
- "The assimilative capacity of a water body is part of the water resource and, as such, must be managed judiciously and shared in an equitable manner amongst all water users for the disposal of their wastes."
- "For those pollutants which pose the greatest threat to the environment, because of their toxicity, extent of bio-accumulation and persistence, a precautionary approach aimed at minimising or preventing inputs to the water environment should be adopted."

The Government Gazette No. 20526 8 October 1999 indicates the actual requirements for discharge of waste or water containing waste into a water resource through a pipe, canal, sewer or other conduit. In this document, rivers are classified according to use and either excluded from or included in the general authorisation for discharge. Those that are included are further subdivided into rivers accepting effluents that have complied with the special limit values or the general limit values. It also gives an indication of the monitoring requirements for domestic wastewater discharges. Figure 2.5 summarizes the general requirements for compliance with this document. The General and Special limit values are presented in Table 2.2.

Table 2.2: Discharge limit values applicable to discharge of wastewater into a water resource, from Government Gazette (1999).

Parameter	General limit	Special limit
Faecal Coliforms (per 100 ml)	1000	0
COD (mg/l)	75*	30*
pH	5.5 to 9.5	5.5 to 7.5
Ammoniacal Nitrogen (mg/l)	3	2
Nitrate/Nitrite nitrogen (mg/l)	15	1.5
Free Chlorine (mg/l)	0.25	0
Suspended Solids (mg/l)	25	10
Electrical Conductivity (mS/m)	70 mS/m above intake to a max. of 150 mS/m	50 mS/m above background receiving water to a max. of 100 mS/m
Ortho-Phosphate (mg/l)	10	1 (median) and 2.5 (max.)
Fluoride (mg/l)	1	1
Soap, oil or grease (mg/l)	2.5	0
Dissolved Arsenic (mg/l)	0.02	0.01
Dissolved Cadmium (mg/l)	0.005	0.001
Dissolved Chromium (mg/l)	0.05	0.02
Dissolved Copper (mg/l)	0.01	0.002
Dissolved Cyanide (mg/l)	0.02	0.01
Dissolved Iron (mg/l)	0.3	0.3
Dissolved Lead (mg/l)	0.01	0.006
Dissolved Manganese (mg/l)	0.1	0.1
Dissolved Selenium (mg/l)	0.02	0.02
Dissolved Zinc (mg/l)	0.1	0.04
Mercury and its compounds (mg/l)	0.005	0.001
Boron (mg/l)	1	0.5

*After the removal of algae

Discharge of waste or water containing waste into a water resource through a pipe, canal, sewer or other conduit.

Authorisation does:

- Apply throughout the Republic of South Africa, except for the areas set out in Table A1 (Appendix A).
- Apply to a person who owns or lawfully occupies property registered in the Deeds Office as at the date of this notice or lawfully occupies or uses land that is not registered or surveyed.
- Applies to the discharge of domestic and industrial wastewater into a water resource.

Authorisation does not:

- Apply to discharge into sea, aquifer or any other ground water resource.
- Apply to the discharge of a Complex Industrial Wastewater, classified by DWAF.
- Apply to any water user under Schedule 1 of the National Water Act (1998).
- Apply to areas set out in Table A1 (Appendix A).
- Replace any existing authorisation recognized under the National Water Act (1998).
- Exempt a person from compliance with section 7(2) of the Water Service Act, 1997, or who uses water from compliance with any other provision of the National Water Act unless stated otherwise by any applicable law, regulation, ordinance or by-law.

Discharge limit values and compliances for discharge into a water resource that is not listed in Table A2 (Appendix A):

- Discharge up to 2000 m^3/day
- Comply with General Limit Values (Table 2.2).
- Discharge may not alter the natural ambient water temperature of the receiving water resource by more than 3°C.

Discharge limit values and compliances for discharge into a water resource listed in Table A2 (Appendix A):

- Discharge up to 2000 m^3/day
- Comply with Special Limit Values (Table 2.2).
- Discharge may not alter the natural ambient water temperature of the receiving water resource by more than 2°C.

Other requirements:

- The discharge of the wastewater into a water resource must be registered, in terms of the schedule in the Government Gazette (1999).
- The registered user must ensure the establishment of monitoring programs prior to the commencement of discharge.
- The registered user must follow acceptable precautionary practices.
- Inspections by an authorised person must be allowed.

Figure 2.5: Requirements for the discharge of domestic and industrial wastewaters into a water resource, from Government Gazette (1999).

2.5 Leachate management and treatment technologies

Due to the possible adverse impacts which landfill leachate may pose to the receiving environment, leachate generated from the landfill needs to be managed in an appropriate and cost effective way. This may simply entail off-site channeling of collected leachate into a nearby sewer line for treatment downstream at the sewage treatment works. However, the possibility of methane explosions occurring in the sewer lines due to the dissolved methane present in older methanogenic leachates is becoming more apparent and air stripping of dissolved methane should be a common practice before discharge into sewer lines (Robinson, 2001a). For landfills located in areas without an available sewer line, advanced leachate treatment processes may be required. The management of leachate does not only include its collection and treatment, it also entails landfill management, operation and leachate generation. Leachate quality and quantity are attributed to a number of interrelated factors such as type of waste stream, depth and age of landfill, amount of water ingress, landfill design and operation and the activity of biological, physical and chemical processes within the waste body (Qasim and Chiang, 1994). In large individual sites, typical of modern landfilling, the high rate of compaction of the waste and cover material have encouraged the rapid onset of anaerobic conditions leading to the production of leachates with high biodegradable organic concentrations which slowly decompose into large quantities of landfill gas (Robinson et al, 1992). High quality cover materials and peripheral cut-off drains aid in reducing the amount of water ingress into a landfill and hence reduce the rate of leachate generation (Robinson et al, 1992). These modern landfill management techniques can, however, increase the time scale of leachate and gas generation to beyond a century, during which generated leachate will build up in the base of the site and extraction and appropriate treatment will inevitably be required (Robinson et al, 1992).

2.5.1 Treatment technologies

The selection of a treatment process is highly complex due to the characteristics of landfill leachate, changing over a specific landfills age and from landfill to landfill, it should be site specific and be appropriate both economically and technologically (Qasim and Chiang, 1994). Advanced leachate treatment includes biological, physical and chemical processes as well as combinations of the three (Qasim and Chiang, 1994 and Robinson, 2001a).

2.5.1.1 Biological treatment processes

In biological treatment processes the leachate comes into contact with a mixed consortium of microorganisms. The chemical reactions that take place during the treatment process are biologically mediated. Biodegradable organics and other nutrients of concern are used as a substrate by the microorganisms for subsistence, growth and synthesis of new cells (metabolism) (Peavy et al, 1985; Tchobanoglous and Burton, 1991; Nathanson, 2000 and Hammer and hammer, 2001).

Aerobic biological treatment processes include suspended growth systems, attached growth systems and combinations of the two (Tchobanoglous and Burton, 1991; Qasim and Chiang, 1994 and Department of the Environment (UK), 1995).

A **suspended growth** system involves the mixing of leachate containing BOD, contaminants, solids and nutrients with a large population of active microorganisms suspended in an aeration basin. Hydraulic retention times ranging from 10 to 20 days have been found to be capable of greater than 90 percent removal of ammoniacal nitrogen and COD in landfill leachates. In the activated sludge process, concerned with the removal of organics, microorganisms are mixed thoroughly with the leachate so that they can grow and stabilize the substrate. As the microorganisms grow in the presence of oxygen and are mixed by agitation, the individual organisms flocculate to form an active mass of microbial floc called 'activated sludge'. The mixture of the activated sludge and leachate in the aeration basin is called 'mixed liquor suspended solids' (MLSS), and the volatile organic fraction is called 'mixed liquor volatile suspended solids' (MLVSS). The MLSS passes from the aeration basin into a secondary clarifier where the activated sludge is settled. A portion of the settled sludge is returned to the aeration basin to maintain the correct food to microorganism ratio. There is usually an excess amount of sludge produced due to cell synthesis, this excess is wasted. The extended aeration plants used for leachate treatment are similar in principle, however, they differ in their operation; the short hydraulic retention times of the activated sludge plants only enable reduction of COD with limited ammoniacal nitrogen removal while the longer retention times in the extended aeration plants are capable of greater than 90 percent removal of ammoniacal nitrogen and COD. Typical aeration basins are plug flow, completely mixed and arbitrary flow reactors. In a plug flow reactor the leachate particles flow through the tank and are discharged in the same sequence in which they enter. The particles remain in the tank for a time equal to the theoretical retention time. In a completely mixed (or continuous flow reactor) reactor the leachate particles are completely mixed when they enter the reactor and are dispersed immediately throughout the reactor. The particles leave the tank in proportion to their statistical population. An arbitrary flow is any degree of partial mixing between plug flow and complete mixing.

During aeration ammoniacal nitrogen is oxidized to nitrate nitrogen (nitrification) by a population of autotrophic nitrifiers (Haandel and Marais, 1981). BOD removal can also be achieved during nitrification due to the possibility of a mixed population of aerobic heterotrophs being present in the MLSS, typical of extended aeration treatment plants.

The aerobic treatment may also take place in a lagoon (aerated lagoon), which is a large aeration basin with several days of hydraulic retention period. The aerated lagoon uses the same microbial reactions as the activated sludge process, however there is no sludge recycle. A sequencing batch reactor (SBR) is a fill and draw treatment system where the food and microorganism contact, organics stabilization, sedimentation and discharge of a clear effluent called a supernatant occur in a single basin. Flow of leachate may occur during the fill period or during the react period depending on the strength of the leachate.

An **attached growth** system involves a population of active microorganisms that are supported over solid media. The solid media may be natural or synthetic materials. There are two main types of attached growth systems:

1. Trickling filters. In a trickling filter the leachate is sprayed through the air to absorb oxygen and then allowed to trickle through a bed of natural or synthetic material coated with a slime of microbial growth. The use of this system for high strength leachates is limited due to increasing organic and inorganic loadings which cause 'clogging' of the filter medium through the build-up of slims and the precipitation and build-up of inorganic salts.
2. Rotating biological contractor (RBC). The RBC consists of a series of circular plastic discs mounted over a shaft that rotates slowly. A portion of the disks remains submerged in the tank and portions of the disks are exposed to the air. The biological growth develops over the disks, which receive alternating exposures to leachate and air. RBC's have been proven to be more successful in the treatment of leachates, with high COD and ammoniacal nitrogen concentrations, than trickling filters.

Anaerobic/anoxic treatment processes may also be divided into suspended growth systems, attached growth systems and combinations of the two. The removal of ammoniacal nitrogen, which is regarded as the major long-term contaminant in many landfill leachates, cannot be achieved in an anaerobic/anoxic system (Tchobanoglous and Burton, 1991; Qasim and Chiang, 1994 and Department of the Environment (UK), 1995).

A **suspended growth** anaerobic/anoxic system involves the mixing of leachate with biological solids under anaerobic/anoxic conditions. Anaerobic/anoxic suspended growth processes include:

1. Conventional. High organic leachate, typical of the acetogenic phase of biodegradation, is stabilized in a digester.
2. Contact process. The leachate is digested in a completely mixed anaerobic reactor. The digested solids are settled in a clarifier and then returned to the digester.
3. Upflow anaerobic sludge blanket. Leachate enters the bottom and flows upward through a blanket of biologically formed granules or solids.
4. Denitrification. Denitrification refers to a biological redox reaction in which nitrate, an inorganic nitrogen compound, is reduced to nitrogen gas (Haandel and Marais, 1981 and Robinson et al, 1998b).
5. Combined anoxic, anaerobic and aerobic system. Nitrogen and phosphorus are removed along with BOD in an anoxic, aerobic and anaerobic environment treatment system. The treatment may take place in separate reactors or in the same SBR.

An **attached growth** anaerobic system involves a microbiological film that is supported over a solid medium. The organic matter is stabilised as the leachate comes in contact with the attached growth. Anaerobic attached growth processes include:

1. Anaerobic filter. A reactor is filled with a solid medium and the leachate flows upward through the bed.
2. Expanded or fluidized bed. A reactor is filled with media such as sand, coal, or gravel. The influent and recycled effluents are pumped from the bottom. This process has been used to dilute leachate.
3. Rotating biodisks. Circular disks are mounted on a central shaft and rotated while completely submerged in an enclosed housing. Biofilm grows over the disks and stabilizes the leachate.

4. Denitrification. Attached growth in an anoxic environment and in the presence of a carbon source reduces nitrate nitrogen and nitrite nitrogen to nitrogen gas.

Generally, biological leachate treatment is the most favorable treatment technique for landfill leachate (Christensen et al, 1998). Christensen et al (1998) pointed out that the design criteria for sewage treatment plants could not be used for landfill leachate and there were certain inherent points that needed to be considered:

- Excessive foam production.
- Clogging of pipe works due to the precipitation of certain constituents.
- Low phosphorus concentrations experienced in landfill leachate
- Changing BOD to COD ratios and ammoniacal nitrogen strengths with landfill age.
- Halogenated hydrocarbons.

Mavinic (1998) stated that the most difficult type of leachate to treat would be a low carbon, high nitrogen leachate, with a variety of metals, low temperature and varying pH-value.

2.5.1.2 Physical and chemical treatment processes

Chemical and physical treatment processes will be briefly discussed here for the purpose of completeness (Tchobanoglous and Burton, 1991; Qasim and Chiang, 1994 and Department of the Environment (UK), 1995).

Physical Treatment

1. Equalization. Flow and mass loadings are equalized by means of utilizing in-line or off-line equalization chambers.
2. Screening. Suspended and floating debris are removed by straining action.
3. Flocculation. Fine particles are aggregated by utilizing gentle stirring.
4. Sedimentation. Solids are removed by gravity.
5. Flotation. Solids are floated by fine air bubbles and skimmed from the surface.
6. Air stripping. Air and leachate are contacted in countercurrent flow in a stripping tower. Air stripping may be used for both ammonia and dissolved methane removal. During the air stripping of ammonia, the pH of the leachate is adjusted to values of 11 or above to convert the ammoniacal nitrogen to gaseous ammonia, large amounts of air are then passed through the leachate to aid in the removal of the gaseous ammonia, which is then released to the atmosphere.

7. Filtration. Suspended solids are removed in a filter bed or micro screen.
8. Membrane processes or reverse osmosis (RO). Dissolved and suspended solids, ammoniacal nitrogen, heavy metals may be removed by membrane separation and COD and BOD_5 may be reduced. This treatment technique is suitable for leachates with high inorganic loading and low volumetric flow rates, it does not treat or degrade any contaminants, but, through ultrafiltration, is able to concentrate soluble constituents of the leachate into a 'brine' and produce a permeate which can achieve high standards treatment.
9. Natural evaporation. The leachate is stored in basins that have an impervious liner and the liquid is evaporated. Usually, acid is added to the leachate, to convert volatile ammonia into soluble ammonium salts. As with RO, this technique is not a treatment, but rather one of concentration.

Chemical Treatment

1. Coagulation. Colloidal particles are destabilized by rapid dispersion of chemicals such as lime, sodium and magnesium hydroxide.
2. Precipitation. Solubility is reduced by chemical reaction.
3. Gas transfer. Gases are added or removed by mixing, air diffusion and change in pressure.
4. Chemical oxidation. Oxidizing chemicals such as hydrogen peroxide or hypochlorite are used to oxidize organics, hydrogen sulfide, ferrous and other metal ions. This technique is often used in situations where odours caused by sulphides are a particular problem. Ozone has also been used to control odour, oxidise pesticides and improve biodegradability of other organic compounds.
5. Chemical reduction. Metal ions are reduced for precipitation, recovery and conversion into a less toxic state.
6. Disinfection. Pathogens are destroyed using strong oxidizing agents or ultraviolet light.
7. Ion exchange. The removal of inorganic compounds from a liquid is achieved.
8. Activated carbon adsorption. Used for the reduction of residual BOD, COD, toxic and refractory organics. The suspended solids need to be removed prior to treatment to prevent 'clogging' of the carbon filter. Once the adsorption capacity of the carbon filter has been saturated with adsorbent, it may be regenerated. This treatment technique is suitable for effluent polishing situations and can be highly effective with up to 99 percent

removal; however, it is expensive if significant quantities of residual COD require treatment.

Research into the use of chemical and physical treatment processes have shown that they can be useful for leachates from older landfills with low BOD to COD ratios and as a polishing step for biologically treated leachates (Qasim and Chiang, 1994). Activated carbon has been used successfully to remove residual refractory organics and was actually found to give better removal than chemical precipitation (Qasim and Chiang, 1994). Ozonation has also proven to be successful; an ozonation plant at the Buckden South Landfill in the United Kingdom was successfully used, following biological pretreatment using an SBR, to degrade the herbicides, mecoprop and isoproturon into biologically degradable organic materials, which were polished in a final constructed wetland (Robinson and Harris, 2001). Air stripping of methane gas in landfill leachates before discharge to sewerline, as mentioned before, has also been proven to be very successful (Robinson, 2001a).

Other techniques of leachate treatment include discharge to sewer, as mentioned above, leachate recirculation back onto the landfill, land and reed bed treatment. Recirculation uses the landfill as a large bioreactor to reduce high strength acetogenic leachates, however, the removal of inorganic material and ammoniacal nitrogen has been found to be low (Qasim and Chiang, 1994). It may also make more effective use of the absorptive capacity of the landfill wastes and improve moisture distribution within the waste body, enhancing the production of landfill gas. The removal of ammoniacal nitrogen before recirculation has shown some promise in pilot scale studies (Department of the Environment (UK), 1995). Land treatment implies the spray irrigation of low strength leachate onto grassland, coniferous and broadleaf woodland and peat slopes. The spray irrigation of stronger acetogenic leachates is unlikely to be a successful method, however, for weaker and pretreated leachates irrigation has significant potential for effluent polishing (Department of the Environment (UK), 1995). Reed bed systems, described in further detail in Chapter 3, have also shown excellent removal potential for suspended solids, iron and COD and BOD_5, but, poor removal of ammoniacal nitrogen is also a common finding (Department of the Environment (UK), 1995; Reed et al, 1995; Kadlec and Knight, 1996 and Mulamoottil et al, 1998).

CHAPTER 3

3 CONSTRUCTED WETLANDS

3.1 Introduction to wetlands and constructed wetlands

Wetlands are defined as land areas where geological and hydrological conditions promote the formation of hydric soils long enough to alter soil properties chemically, physically and biologically and to support vegetation typically adapted for life in saturated soil conditions (Rogers et al, 1985; Mitsch and Gosselink, 1993; Reed et al, 1995; Kadlec and Knight, 1996 and Brix, 1997). Natural wetlands have been used as discharge sites for wastewater since the early 20th Century (Kadlec and Knight, 1996). However this practice and its effect on the receiving environments were not sufficiently monitored. Only during the nineteen sixties and seventies was intense monitoring initiated at some of the discharge sites. This developed the awareness for the potential of wetlands to treat polluted waters (Kadlec and Knight, 1996). During the past twenty years scientists have gained sufficient understanding of these natural systems to utilise them to improve the quality of polluted waters (Lehman and Rodgers, 2000). Wetlands have a higher rate of biological activity than most ecosystems due to their abundance in water and plants (Wetzel, 1993 and Kadlec and Knight, 1996). The use of natural wetlands for polluted water treatment has, however, been constrained as natural wetlands are considered by most regulatory authorities as receiving waters and any discharge into such waters is required to meet specified discharge standards (Wetzel, 1993 and Reed et al, 1995). The deliberate use of natural wetland systems for the treatment of polluted waters in South Africa is prohibited and the policy of the governmental water agency is that natural wetlands should be maintained to buffer diffuse source pollution (Rogers et al, 1985). The functional components of natural wetlands are extremely variable which makes it difficult to accurately predict responses to polluted water applications (Brix, 1993). On the other hand, the use of constructed wetlands (CW) avoids the legislative requirements and allows for a greater degree of control (Brix, 1993). CW are aimed at simulating natural wetlands by optimising their treatment properties in order to achieve higher treatment efficiencies (Wetzel, 1993). To meet increasingly stringent discharge standards CW have become the most economical option for secondary or tertiary treatment of a variety of polluted waters, including municipal, commercial and industrial wastewaters (Tchobanoglous and Burton, 1991, IAWQ, 1994, 1995, 1997, Reed et al, 1995 and Kadlec and Knight, 1996). Unlike most advanced wastewater treatment systems, which require intensive inputs during construction and operation (such as: concrete, steel, chemicals and fossil fuels), CW rely more on natural energies (such as: the sun, wind, soil, plants and animals), to accomplish treatment goals (Reed et al, 1995, Kadlec and Knight, 1996 and Lehman and Rodgers, 2000).

CW are divided into two main categories, Free Water Surface Flow (FWS) and Subsurface Flow (SF). Hybrid combinations of these two categories are also often used to achieve treatment of specific pollutants.

3.1.1 Subsurface flow constructed wetlands

SF CW can be further subdivided into horizontal and vertical flow. In the horizontal subsurface flow constructed wetland (Figure 3.1), also referred to as vegetated submerged bed (VSB) (EPA, 2000), an excavated basin is filled with a porous medium and the free water level is maintained at or below the top of the medium. The flow of the polluted water occurs horizontally through the bed substrate where it comes into contact with a mixture of facultative microbes living in association with the substrate and plant roots. The VSB is planted with emergent aquatic vegetation and is lined, if necessary, in order to eliminate hydraulic losses and to protect the surrounding environment from any migration of polluted water. The substratum and plant roots provide the attachment surfaces for the microbial growth. The vertical flow (VF) constructed wetland (Figure 3.2) presents a similar bed as for the horizontal flow, however the polluted water is applied uniformly over the top of the bed, and the effluent flows out through a perforated pipe on the bottom of the bed parallel to the VF CW long axis (Brix, 1993a, Crites, 1994, Reed et al, 1995, Wood, 1995 and Kadlec and Knight, 1996).

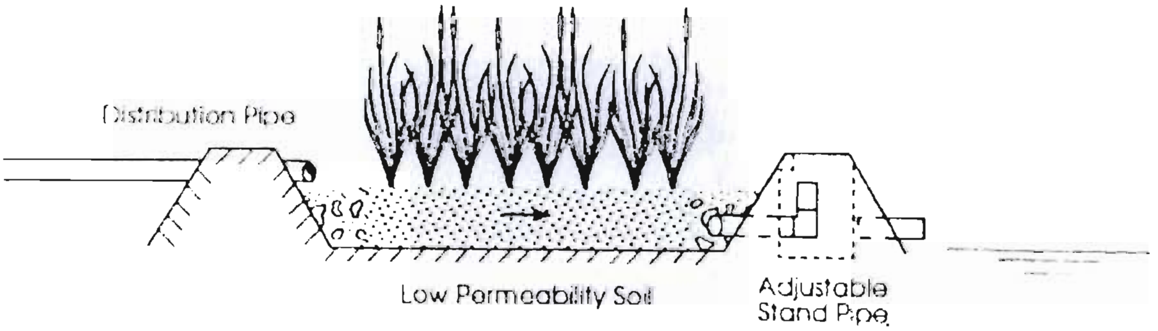


Figure 3.1: Typical cross sectional layout of a VSB, from Kadlec and Knight (1996).

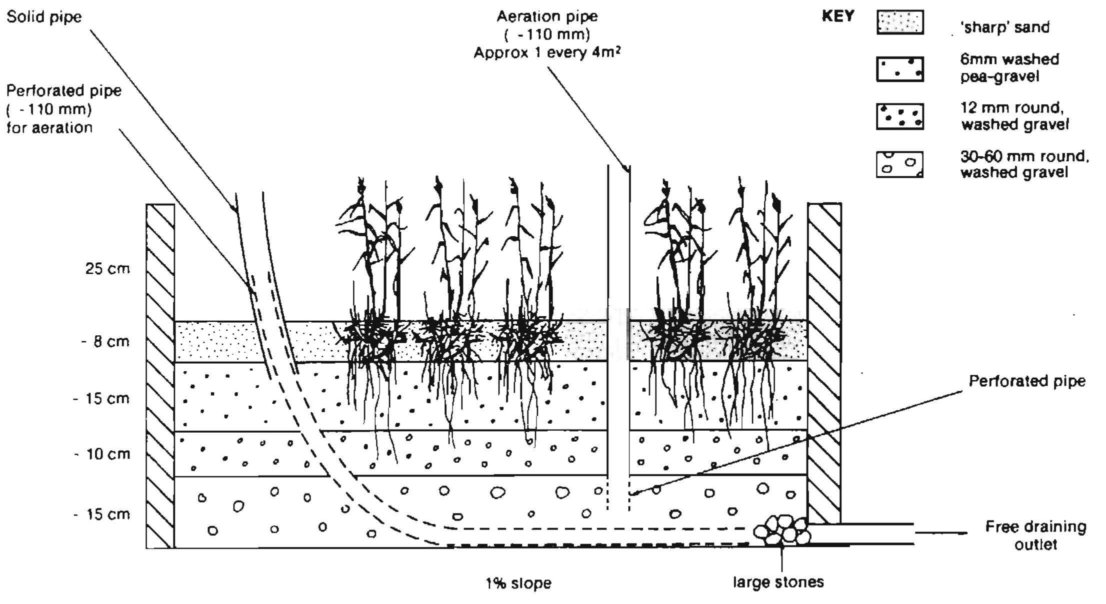


Figure 3.2: Typical cross sectional layout of a VF constructed wetland, from EC/EWPCA (1990).

3.1.2 Free water surface constructed wetlands

This constructed wetland (Figure 3.3) is characterized by the water surface being exposed to the atmosphere. The constructed wetland is planted with emergent aquatic vegetation (emergent macrophytes). The base of the constructed wetland is usually lined to reduce hydraulic losses and prevent migration of polluted waters into the surroundings. Treatment takes place as the polluted water comes into contact with the macrophytes submerged stems, leaves and litter, which create attachment sites for an active bacterial population (Kadlec et al, 1993, Brix, 1993, Crites, 1994, Reed et al, 1995, Wood, 1995 and Kadlec and Knight, 1996).

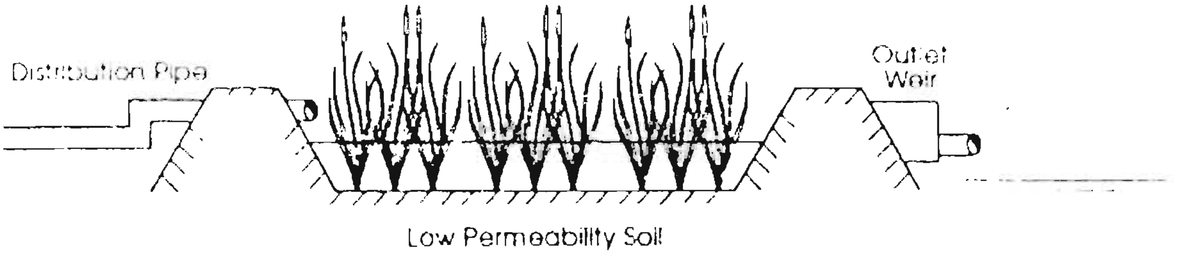


Figure 3.3: Typical cross sectional layout of a FWS constructed wetland, from Kadlec and Knight (1996).

3.1.3 Hybrid constructed wetland systems

Hybrid systems feature combinations and/or modified forms of FWS, VSB and VF constructed wetlands. They may be all included in one 'cell' or sequentially in series. These systems are designed for specific treatment needs and depend on the characteristics of the polluted water, climate and the amount of available land (Brix, 1993a and Lehman and Rodgers, 2000). An example of a modified constructed wetland is the nitrification filter bed (NFB) (Figure 3.4) (Reed et al, 1995).

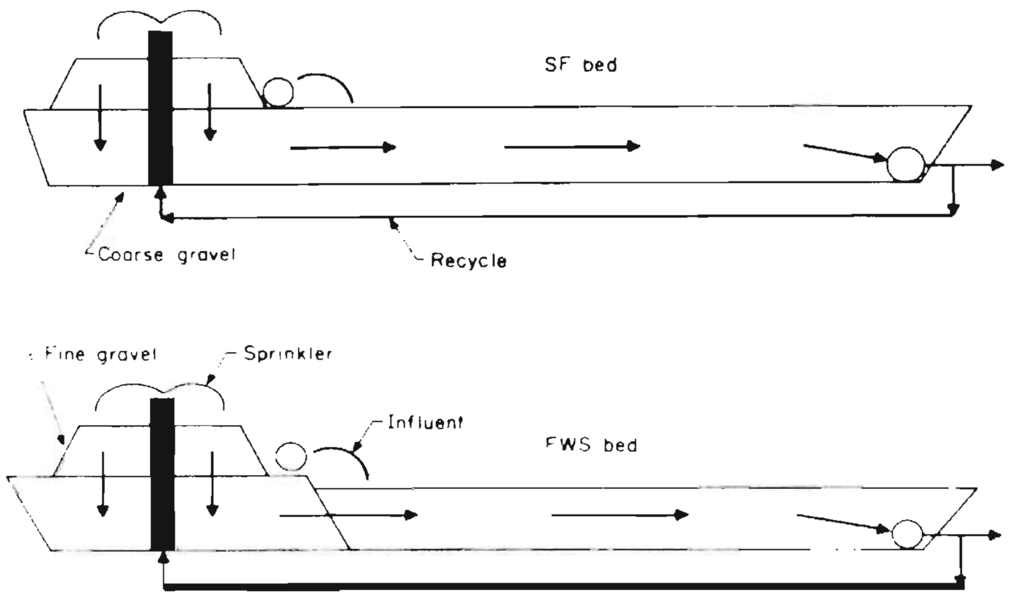
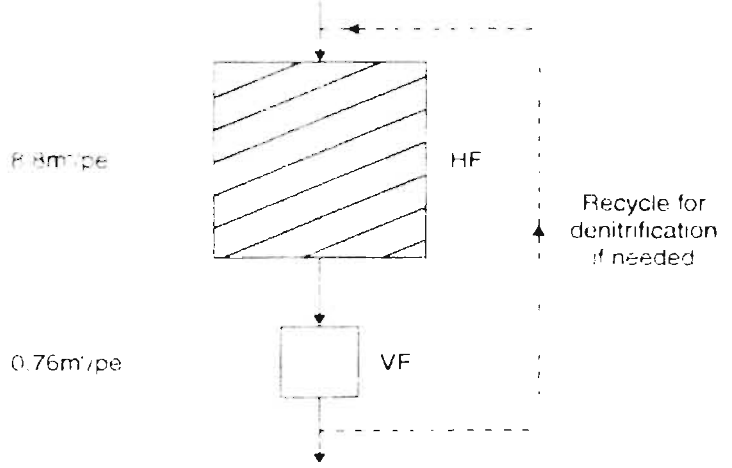


Figure 3.4: Schematic diagram of nitrification filter bed, from Reed et al (1995).

This type of bed was developed to overcome the difficulties in meeting ammonia discharge limits. The system consists of a VF gravel filter bed on top of an existing VSB or FWS wetland bed (Reed et al, 1995). The NFB system design is based on trickling-filters and rotating biological contractor attached growth concepts. In order to achieve a successful nitrification performance, biodegradable organics need to be low (BOD/TKN<1), aerobic conditions must be maintained in the attached film of nitrifying organisms, the attachment surfaces must remain moist at all times to sustain microbial activity, and there must be sufficient alkalinity to support nitrification (10 g alkalinity/ 1 g ammonia nitrified) (Reed et al, 1995). Other modified forms include the fill and draw system developed to enhance nitrification, denitrification and organics removal (Reed et al, 1995, Green et al, 1997, Cooper, 1999 and Cooper et al, 1999). Cooper (1999) classified hybrid systems into two main types depending on system layout; the first system, described by Johansen and Brix

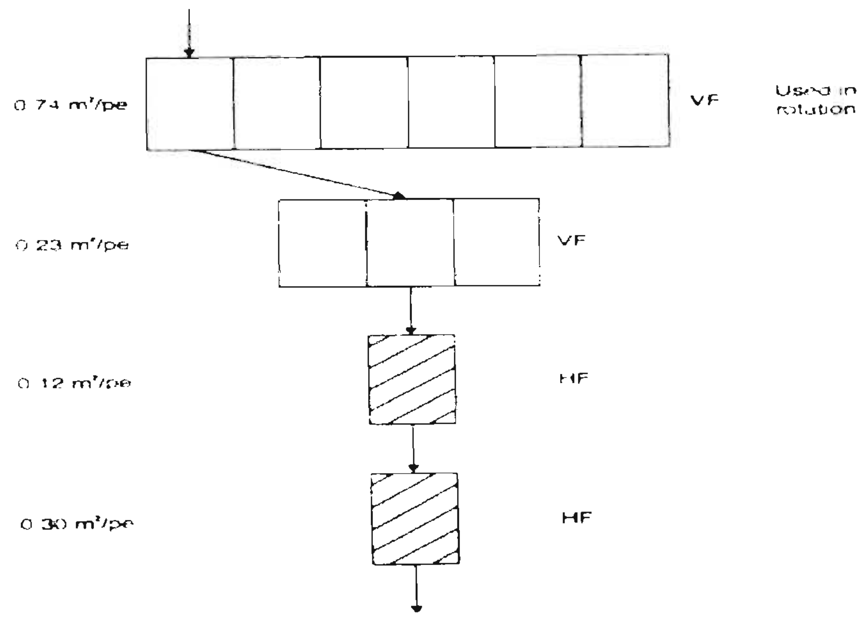
(1996), is based on the idea that the biodegradable organics will be removed in the horizontal flow bed and nitrification would then take place in the vertical flow bed. This system is characterized by having a horizontal flow system first followed by a vertical flow



system (Figure 3.5).

Figure 3.5: Hybrid system with horizontal flow bed followed by a vertical flow bed, from Cooper (1999).

The second system (Figure 3.6) is characterised by the vertical flow system first followed



by horizontal flow beds (Cooper, 1999).

Figure 3.6: Hybrid system with vertical flow system followed by horizontal flow beds in

series, from Cooper (1999).

3.2 Constructed wetland hydrology and hydrological design parameters

The hydrology of a constructed wetland system is strongly related to the type of bed and the local climatic conditions. An overall understanding of the hydrology of the CW system is essential as the hydrological design is critical to its successful performance (Reed et al, 1995).

3.2.1 Definitions of hydraulic terms

Hydraulic loading rate

Hydraulic loading rate is the measure of the rate of application of a volume of water over the surface area of the constructed wetland (equation 3.1) (Kadlec and Knight, 1996).

$$HLR = \frac{Q_i}{A_S} \qquad (3.1)$$

where HLR = Hydraulic loading rate, m/d

Q_i = input polluted water flow rate, m^3/d

A_S = wetland top surface area, m^2

Hydraulic residence time

It is the measure of the average time taken for one constructed wetland bed volume to be replaced (equation 3.2) (Kadlec and Knight, 1996).

$$HRT = \frac{V}{Q} \qquad (3.2)$$

where HRT = hydraulic residence time, d

Q = the average flow rate of polluted water through the wetland, m^3/d

V = water storage in wetland, m^3

The water storage (V) in the wetland may be calculated as (Reed et al, 1995) (equation 3.3):

$$V = LWyn \qquad (3.3)$$

where L = length of wetland cell, m

W = width of wetland cell, m

y = depth of water in the wetland cell, m

n = porosity

The average flow compensates for water losses or gains (equation 3.4) (Reed et al, 1995):

$$Q = \frac{Q_i + Q_o}{2} \tag{3.4}$$

where Q = average flow, m^3/d

Q_o = output of polluted water flow rate, m^3/d

Surface area loading rate

The surface area loading rate is a measure of the mass of pollutant applied to the surface of the constructed wetland over a period of time (equation 3.5) (EPA, 2000):

$$SLR = \frac{C_i \times Q_i}{A_S} \tag{3.5}$$

where SLR = Surface area loading rate, $g/m^2/d$

C_I = Concentration of pollutant in the influent, g/m^3

Volumetric loading rate

The volumetric loading rate is a measure of the mass of pollutant applied to the pore volume of the VSB system over a period of time (equation 3.6) (EPA, 2000).

$$VLR = \frac{C_i \times Q_i}{LWyn} \tag{3.6}$$

where VLR = Volumetric loading rate, $g/m^3/d$

Hydroperiod and water regime

The hydroperiod is the time that a wetland soil or gravel is saturated or flooded. It is usually expressed as a number of days or a percentage of the time in flooded conditions during the year. Continuous systems, usually, have a hydroperiod of 365 days or 100% of the year. The water regime refers to both the hydroperiod as well as the water depth (Kadlec and Knight, 1996).

Actual velocity

The actual velocity is the velocity that would be measured by a velocity measuring instrument, such as a current meter (Massey, 1995) (Kadlec and Knight, 1996) (equation 3.7):

$$v = \frac{Q}{nA_c} \qquad (3.7)$$

where v = actual velocity, m/d

A_c = Cross sectional area perpendicular to the flow, m

Superficial velocity

The superficial velocity is the apparent flow velocity through the entire cross sectional area of the bed (Reed et al, 1995) (equation 3.8):

$$u = \frac{Q}{A_c} \qquad (3.8)$$

where u = superficial velocity, m/d

3.2.2 Hydrological balance

The general movement of water inside a constructed wetland follows the same pattern for both FWS and SF systems. The over-all water balance may be expressed by equation 3.9 (Kadlec and Knight, 1996):

$$Q_i - Q_o + Q_c - Q_b - Q_{gw} + Q_{sm} + P \cdot A - ET \cdot A_S = \frac{dV}{dt} \qquad (3.9)$$

where ET = evapotranspiration rate, m/d

P = precipitation rate, m/d

Q_b = Infiltration rate out of the system through the side walls, m^3/d

Q_c = catchment runoff rate, m^3/d

Q_{gw} = infiltration to groundwater, m^3/d

Q_{sm} = snowmelt rate, m^3/d

t = time, d

The relative significance of the terms in equation 3.9 is dependent on the system type and local climatic conditions (Kadlec and Knight, 1996). For the climatic conditions experienced in Durban and for the operational conditions used in this research equation 3.9 may be simplified to equation 3.10:

$$Q_i - Q_o + PA - ET \cdot A_S = \frac{dV}{dt} \qquad (3.10)$$

Equation 3.10 is more representative of a lined CW with no snowfall and peripheral cut off drains or walls.

Precipitation

For a conceptual design, average annual precipitation data may be used, however for a more detailed design, monthly average precipitation values should be used (Table 3.1) (Kadlec and Knight, 1996).

Table 3.1: Average annual and monthly precipitation in mm, from 1961 to 1990, for selected major towns in South Africa (courtesy of the South African Weather Bureau).

Month	Durban	Cape Town	Johannesburg	Pretoria	Nelspruit	Bethlehem
JAN	134	15	125	136	127	96
FEB	113	17	90	75	108	77
MAR	120	20	91	82	90	94
APR	73	41	54	51	51	58
MAY	59	69	13	13	15	9
JUN	28	93	9	7	9	12
JUL	39	82	4	3	10	7
AUG	62	77	6	6	10	27
SEP	73	40	27	22	26	35
OCT	98	30	72	71	75	83
NOV	108	14	117	98	115	96
DEC	102	17	105	110	131	86
Annual Av.	**1009**	**515**	**713**	**674**	**767**	**680**

Evapotranspiration

Evapotranspiration (ET) is the combination of evaporation from the CW water surface exposed to the atmosphere and transpiration from vegetation. The factors governing evapotranspiration include energy supply, water vapour transport and moisture available at the evaporative surface (Chow et al, 1988). Vegetation has a shading effect; it reduces the wind and increases humidity near the surface that, in turn, reduces evaporation. The transpiration of the plants may, however, cancel these reducing effects and cause the loss rate to be roughly the same or higher, than unvegetated beds (Reed et al, 1995, Kadlec and Knight, 1996 and EPA, 2000). The determination of the ET for FWS and SF systems differ in principal and it will be presented separately later on.

Free water surface constructed wetlands

There are three main methods for estimating the ET. The 'energy balance' method (Chow et al, 1988, Kadlec and Knight, 1996), 'aerodynamic' method (Chow et al, 1988) and 'pan factor' method (Reed et al, 1995, Kadlec and knight, 1996). ET rates are difficult to accurately measure in FWS wetlands (Rogers et al, 1985, EPA, 2000) and the 'aerodynamic' and 'energy balance' methods are highly involved and require a fair amount of input data that may not be readily available (Kadlec and Knight, 1996). Thus, it has become common practice to use the 'pan factor' methods, which are much simpler and may be used for design purposes (Reed et al, 1995 and EPA, 2000). The simplest 'pan factor' method assumes that the constructed wetland evapotranspiration is equal to the lake evaporation rate, which is in turn, equal to 0.7 to 0.8 times the Class A pan evaporation (Rogers et al, 1985, Geiger et al, 1993, Reed et al, 1995, Kadlec and Knight, 1996, DWAF, 1998, EPA, 2000). Christiansen (1968) proposed further refinements to the 0.7 to 0.8 in the form of multipliers (Equations 3.11 to 3.15).

$$ET = 0.755 \times EP \times C_T \times C_W \times C_H \times C_S \qquad (3.11)$$

where

$$C_T = 0.862 + 0.179\left[\frac{T}{20}\right] - 0.041\left[\frac{T}{20}\right]^2 \qquad (3.12)$$

$$C_W = 1.189 - 0.24\left[\frac{W_i}{1.86}\right] + 0.051\left[\frac{W_i}{1.86}\right]^2 \qquad (3.13)$$

$$C_H = 0.499 + 0.62\left[\frac{H}{60}\right] - 0.119\left[\frac{H}{60}\right]^2 \tag{3.14}$$

$$C_S = 0.904 + 0.008\left[\frac{S}{80}\right] + 0.088\left[\frac{S}{80}\right]^2 \tag{3.15}$$

where
C_H = humidity coefficient
C_S = sunshine coefficient
C_T = temperature coefficient
C_W = wind coefficient
EP = pan evaporation, mm/d
ET = evapotranspiration rate, mm/d
H = relative humidity, percent
S = percentage of possible sunshine, percent
T = temperature, °C
W_i = wind speed, m/s

Christiansen's method was developed for well-watered grass surfaces, however, it has been found to be adequately representative for wetlands in Nevada and Michigan, United States of America (Kadlec and Knight, 1996). The seasonal variation in ET, for the Durban area, may be seen in Figure 3.7 (Raw data presented in Table B1 and B2, Appendix B). During the winter months in Durban relative humidity, temperature and percentage sunshine, decrease, and hence so does the ET.

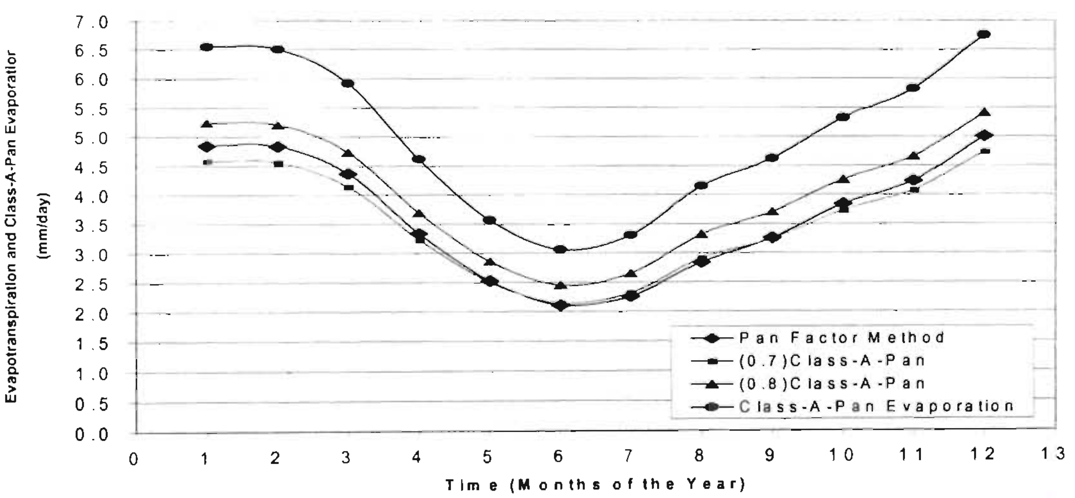


Figure 3.7: Monthly averages of estimated evapotranspiration from 1957 to 1987, Durban International Airport, raw data courtesy of the South African Weather Bureau.

Subsurface flow constructed wetlands

The ET from SF constructed wetland differs from previous case because water vapour must first diffuse through the dry upper layer of gravel or soil and then be transferred by wind penetrating through the vegetation and out of the system. Heat transfer to the water must also pass through the porous medium. The heat storage capacity of the medium is also related to the ET. Due to these changes in moisture and heat transfer, estimations of the ET by the 'energy balance method' are not possible (Kadlec and Knight, 1996). Bavor et al (1988) used water budgets to estimate the ET in gravel bed wetlands; the proposed correlations were found as follows (Equations 3.16 to 3.18).

$$ET = 1.128\ EP + 0.072\ mm/d \qquad (3.16)$$
$R^2 = 0.72$ (Cattail/gravel)
$12\ °C < T_{air} < 25\ °C$

$$ET = 0.948\ EP + 0.0027\ mm/d \qquad (3.17)$$
$R^2 = 0.93$ (Bulrush/gravel)
$12\ °C < T_{air} < 25\ °C$

$$ET = 0.0757\ EP + 0.028\ mm/d \qquad (3.18)$$
$R^2 = 0.15$ (No plants/gravel)
$12\ °C < T_{air} < 25\ °C$

Bavor's findings indicate a tendency for a linear relationship between EP and ET for vegetated gravel beds. The no plants/gravel bed however, does not show this linear relationship. The findings also indicate the importance of the transpiration component of ET for vegetated gravel beds (Kadlec and Knight, 1996). In comparison with the estimated ET rates for FWS CW, graphically presented in Figure 3.8 using the above correlations for the SF CW, the cattail and bulrush estimations were found to be higher. This is not what would be expected, as the changes in moisture and heat transfer would have been expected to reduce these rates. Other authors have also shown ET rates for VSB systems to be higher than that for FWS CW, ET rates of 1.5 to 2 times the pan evaporation have been reported (EPA, 2000). The type of vegetation and density has also been found to affect the ET rates for VSB systems (EPA, 2000).

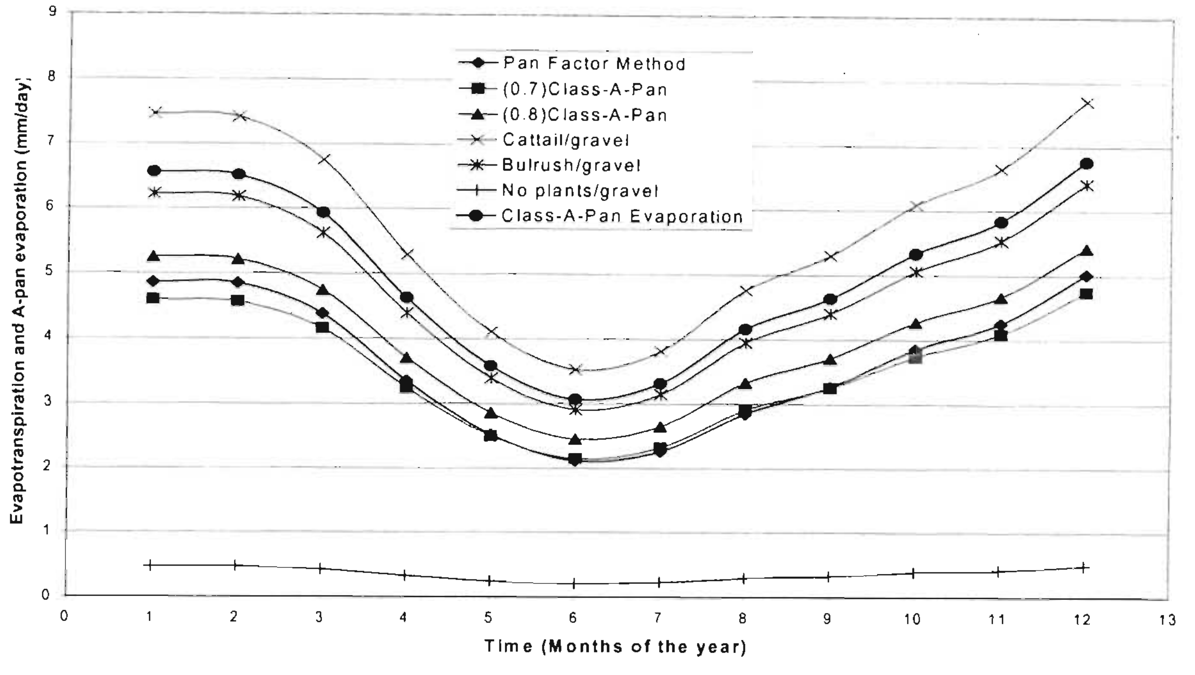


Figure 3.8: Estimated vegetated submerged gravel bed wetland evapotranspiration rates in the Durban area, calculated from equations 3.16 to 3.18, raw data courtesy of the South African Weather Bureau.

Evapotranspiration and precipitation impacts

ET and precipitation are seasonally variable (Kadlec, 1999), but they combine over extended periods to either dilute or concentrate the polluted water as it passes through the CW (Reed et al, 1995, Kadlec and Knight, 1996 and EPA, 2000). Heavy rainfall may cause dilution of pollutants in the CW, reductions in pollutant mass removal and it may reduce the hydraulic retention time (HRT) (Kadlec and Knight, 1996 and EPA, 2000). A free water surface above the top of the SF CW bedding medium, which may be caused by heavy rainfall and increased flows, is a major design concern, as the design flow paths of the polluted water are changed and 'short circuited', leading to shorter HRT and decreased performance (Wood, 1996 and EPA, 2000). Precipitation may also cause re-suspension of solids into the water column, in FWS CW (Kadlec and Knight, 1996). ET concentrates the pollutants and decreases the HRT (Reed et al, 1995, Kadlec and Knight, 1996 and EPA, 2000). It has been found that for non-conservative pollutants, such as biodegradable organics, the increased HRT may modify the removal rate and may either partially offset or enhance the concentrating effects of ET (EPA, 2000).

3.2.3 Flow modelling in horizontal subsurface flow constructed wetlands

The hydraulics of VSB systems may be designed by using modelling techniques of fluid flow through a porous medium (Reed et al, 1995, Kadlec and Knight, 1996 and Kadlec, 1997). The flow in a VSB is driven by gravity and the hydraulic gradient is controlled by the set outlet elevation (EPA, 2000). Darcy's equation may be used to model this type of flow (Equation 3.19) (Kadlec and Knight, 1996):

$$-\frac{dH_{we}}{dx}=\frac{1}{k}\cdot u+\omega\cdot u^{2} \qquad (3.19)$$

where H_{we} = elevation of the water surface, m

k = hydraulic conductivity, m/d

ω = turbulence factor, d^2/m^2

x = longitudinal distance, m

u = superficial water velocity, m/d

The contribution of turbulence may be regarded as negligible or ignored with very small error at Reynolds numbers up to 10 and the flow may be regarded as laminar (equation 3.20) (Reed et al, 1995 and Kadlec and Knight, 1996).

$$\mathrm{Re}=\frac{D\cdot u}{\nu} \qquad (3.20)$$

where Re = Reynolds number

D = average particle diameter of the medium, m

ν = kinematic viscosity, m^2/d

If the Reynolds number is greater than 10 then the flow must be regarded as turbulent and designed accordingly (Reed et al, 1995 and Kadlec and Knight, 1996). For most VSB systems the Reynolds number will be less than 10 and the use of Darcy's equation without the contribution of turbulence may be used for the design (Reed et al, 1995, Kadlec and Knight, 1996 and EPA, 2000). By using Darcy's equation under laminar or turbulent flow conditions, the following assumptions are made:

- the medium is isotropic and homogeneous;

- there is no capillary action;
- steady state flow exists.

Since the bed is not isotropic and homogeneous and the flow may vary due to ET, precipitation and short-circuiting, the mathematical assessment of the flow using Darcy's equation must be used with caution (Reed et al, 1995, Bell, 1993 and Kadlec and Knight, 1996). Capillary action effects can confidently be neglected as the medium used in VSB usually has a much higher pore size than required for capillary action to occur (Kadlec and Knight, 1996). In the case of turbulent flow, Ergun's equation (equation 3.21) will be more appropriate for the design (Reed et al, 1995 and Kadlec and Knight, 1996).

$$-\frac{dH_{we}}{dx}=\frac{150\nu(1-n)^2}{g\cdot n^2\cdot D^2}u+\frac{1.75(1-n)}{g\cdot n^3\cdot D}u^2 \qquad (3.21)$$

where g = acceleration of gravity, m^2/d

Ergun's equation was developed for spheres of single size and because the medium used for VSB is usually angular and not of uniform size, the use of equation 3.21 for VSB systems has been found to over estimate depths by about 10cm (Kadlec and Knight, 1996). Efforts to correlate equation 3.21 to the actual hydraulics of VSB constructed wetlands have lead to the use of a recommended preliminary design equation for clean media (equation 3.22) (Kadlec and Knight, 1996):

$$-\frac{dH_{we}}{dx}=\frac{255\nu(1-n)}{g\cdot n^{3.7}\cdot D^2}u+\frac{2(1-n)}{g\cdot n^3\cdot D}u^2 \qquad (3.22)$$

Selecting a design hydraulic conductivity and porosity

The final design must be based on actual measurements of porosity and hydraulic conductivity (Reed et al, 1995 and Kadlec and Knight, 1996). Published data such as those shown in figures 3.9 and 3.10 may be used as guidelines for the preliminary design (Reed et al, 1995 and Kadlec and Knight, 1996). The porosity of the medium used may be measured in the laboratory using the standard SABS methods (SABS method 844, 1994 and SABS method 845, 1994).

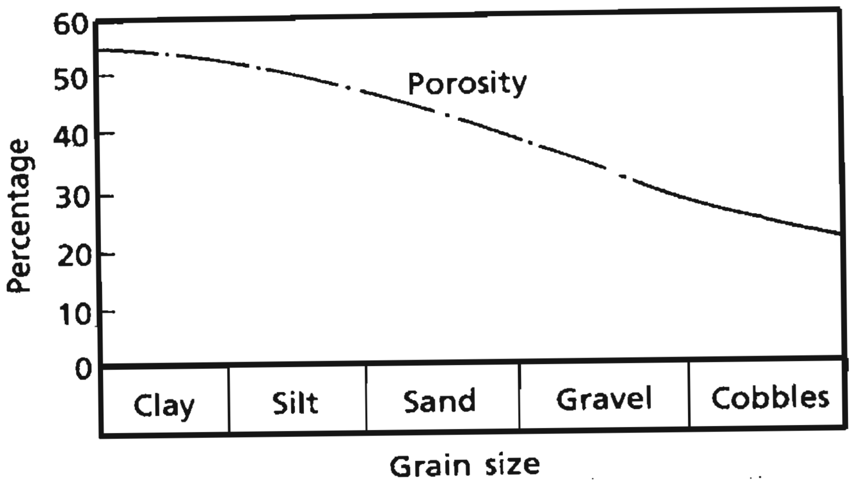


Figure 3.9: Porosity variations with grain size for a well-sorted material (Modified from Bell, 1993).

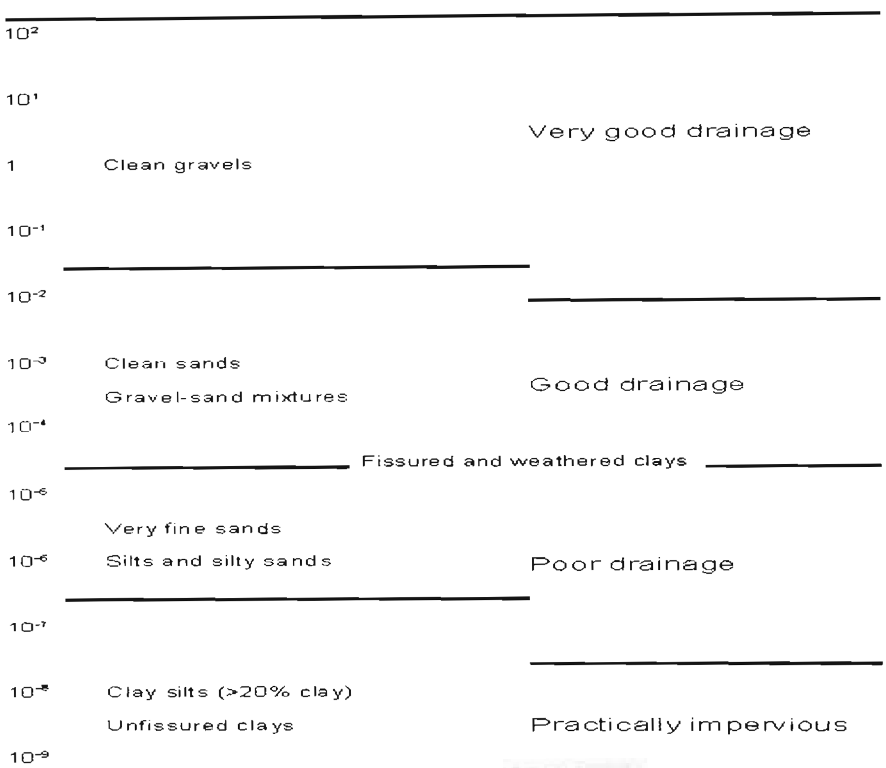


Figure 3.10: Average values of hydraulic conductivity for various soils, note that the unit for hydraulic conductivity shown is m/s (Modified from Whitlow, 1995).

The Hydraulic conductivity of granular materials for CW may be measured either in the laboratory, using the Rowe cell (Vickers, 1983) or the Permeameter trough (Figure 3.11) (Reed et al, 1995), or in the field, using the trench technique (Kadlec and Knight, 1995).

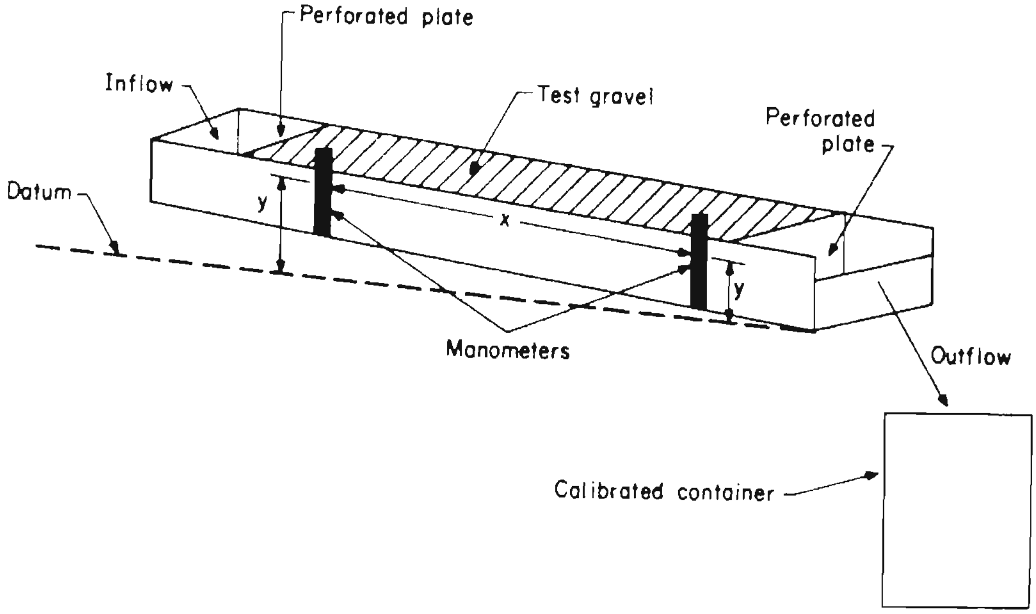


Figure 3.11: Permeameter for measuring hydraulic conductivity, from Reed et al (1995).

The permeameter trough shown in Figure 3.11 is a modification of the standard laboratory permeameter, and has been successfully used to estimate the hydraulic conductivity of a range of gravel sizes (Reed et al, 1995). The trough has a total length of 5 m with perforated plates located at 0.5 m from each end. The medium to be tested is placed between the perforated plates, manometers are used to observe the water levels in the trough and are spaced at about 3 m apart (Reed et al, 1995). In the trench method, the medium to be tested is placed in a sealed horizontal trench, flow measurements are taken and the head loss is measured across the test length by differential manometry (Kadlec and Knight, 1996). When calculating the hydraulic conductivity of the design medium under turbulent flow conditions, dynamic similarities between the full-scale system and the lab scale system should be considered (Massey, 1995). By matching the Reynolds numbers between the two systems, the corresponding hydraulic conductivity could be calculated. The designer must acknowledge that the above mentioned measurements and assumptions are particularly suitable for clean media; in constructed wetlands used for the treatment of polluted waters (in particular landfill leachate) the void spaces between the grains will be gradually filled by deposition of solids and plant roots, altering the ideal measurement of the hydraulic conductivity (Tchobanoglous, 1993 and Kadlec and Knight, 1996). In the past, researchers believed that the plant roots created subsurface channels for the flow and increased the beds hydraulic conductivity (Beven and Germann, 1982), however, more recent scientific data has failed to prove this and, in fact, the hydraulic conductivity has been found to decrease with time (EC/EWPCA, 1990, Brix, 1994 and EPA, 2000). Several authors have not only noticed a reduction in hydraulic conductivity but have also showed relative differences along the initial one quarter to one third of the wetland bed (EPA, 2000). These observations have lead to conservative recommended design values of hydraulic conductivity for long-term operation of VSB CW (EPA, 2000):

- For initial 30% of SF CW k = 1% of clean k
- For final 70% of SF CW k = 10% of clean k

Clean k is measured in the laboratory or field, using the techniques mentioned above, or taken from published data, such as Figure 3.10.

3.2.4 Flow modelling in vertical flow constructed wetlands

The VF constructed wetland typically consists of a number of stages with VF CW in parallel and in series (EC/EWPCA, 1990, Cooper, 1993 and Reed et al, 1995). The main advantage of these beds can be attributed to their ability to restore aerobic conditions during periods of resting, as the beds are loaded alternatively. Typically two days loading and then rested for 4-8 days (EC/EWPCA, 1990 and Reed et al, 1995). During the resting periods air is reintroduced into the bed as the bed drains, the bed is then refilled with polluted water during the loading period. These systems operate under similar conditions to gravel bed filters and the hydraulics could be modelled using the Carmen-Kozeny equation (equation 3.23) (Peavy et al, 1985):

$$h_f = \frac{f \cdot y_f \cdot (1-n) \cdot (HLR)^2}{n^3 \cdot g \cdot D} \qquad (3.23)$$

where h_f = friction loss through bed, m

f = friction factor related to the coefficient of drag around the particles

y_f = depth of filter, m

3.2.5 Flow modelling in free water surface constructed wetlands

FWS CW function as open channels and can be modelled using the Manning equation (Reed et al, 1995, Kadlec and Knight, 1996, Nuttall et al, 1997 and EPA, 2000). The general form of Manning's equation is (equation 3.24) (Roberson et al, 1995):

$$S_O^{1/2} = \frac{Q \cdot n_M}{A_C \cdot y^{2/3}} \qquad (3.24)$$

where S_0 = hydraulic gradient or slope of water surface, m/m

n_M = Manning's resistance coefficient, s/m$^{1/3}$

Unlike typical open channel flow where the resistance to flow only occurs at the channel's wetted perimeter, this type of wetland experiences further resistance within the water column due to the emergent vegetation (Reed et al, 1995, Kadlec and Knight, 1996 and EPA, 2000), therefore the Manning's coefficient is a function of the water column depth. Research has shown that even the highest published value for this coefficient in typical open channel flow, n_M = 0.29 s/m$^{1/3}$ (French, 1985) and n_M = 0.1 s/m$^{1/3}$ (Chow et al, 1988), is one order of magnitude less than values determined from actual constructed wetland data (Kadlec and Knight, 1996). Numerical relationships for Manning's coefficient in FWS CW have been published by various authors, for example equation 3.25 (Kadlec and Knight, 1996) and equation 3.26 (Reed et al, 1995). The numerical relationships are strongly dependant on the depth of the water column and density of vegetation (Reed et al, 1995 and Kadlec and Knight, 1996, EPA, 2000).

$$\frac{n_M}{n_{M1}} = \left(\frac{y_1}{y}\right)^m \qquad (3.25)$$

where m = 0.33-0.5 for depths greater than 20cm

= 1.0-2.0 for depths less than 20cm

$$n_M = \frac{a}{y^{1/2}} \qquad (3.26)$$

where a = resistance factor, $sm^{1/6}$

= 0.4, for sparse low standing vegetation, y>0.4m

= 1.6, for moderately dense vegetation, y = 0.3m

= 6.4, for very dense vegetation and litter layer, y<0.3m

Equation 3.25 was first proposed in the Florida 'emergent marsh' studies (Kadlec and Knight, 1996) and requires a single value of Manning's coefficient at a known depth. In equation 3.26 it is acceptable to assume for design purposes that the value of 'a' lies between 1 $sm^{1/6}$ and 4 $sm^{1/6}$ (Reed et al, 1995). By substitution of equations 3.24 and 3.26, equation 3.24 may be written as equation 3.27 (Reed et al, 1995):

$$S_O^{1/2} = \frac{Q \cdot a}{A_c \cdot y^{7/6}} \qquad (3.27)$$

Thus for a selected hydraulic gradient the FWS CW maximum length can be calculated by rearranging and substituting the terms in equation 3.27 to produce equation 3.28 (Reed et al, 1995):

$$L = \left[\frac{(A_S)(y)^{8/3}(m)^{1/2}}{a \cdot Q} \right]^{2/3} \qquad (3.28)$$

were the equations substituted were:

$$A_c = Wy \qquad (3.29)$$

$$W = \frac{A_S}{L} \qquad (3.30)$$

$$S_O = \frac{m \cdot y}{L} \qquad (3.31)$$

and m = increment of depth serving as head differential.

3.2.6 Sizing of constructed wetland beds

There are two main facets that must be considered in the hydraulic design of CW. The first is the flow characteristics of the specific CW, such as sub-surface, free water surface and vertical. Design of the specific types of wetlands, must ensure that the flow characteristics of the wetland remain generally constant, allowing for small-uncontrolled fluctuations (such as seasonal variations (Kadlec, 1999)) throughout the operational life of the system (Wood, 1996). The second facet is based on the required treatment efficiency of the constructed wetland, which dictates the hydraulic retention time of the system. The calculated HRT then allows the designer to size the CW. However, researchers have not yet achieved a total consensus on the best suitable approach for the hydraulic design of constructed wetlands (Reed et al, 1995). There are three design approaches that are currently used: the first depends on multiple regression analysis of performance data from operating systems, the second utilises a surface area or volumetric loading approach and the third bases the design on the assumption that CW behave as attached-growth biological reactors (Reed et al, 1995). The main features of these design approaches are presented as follows.

Attached growth biological reactor approach

This method is the most commonly published and assumes that the CW behave like an attached-growth biological reactor. The most common way to model this type of reactor in the past has been based on idealised flow patterns such as, plug flow reactors (PFR) and continuous stirred or mixed flow reactors (CSTR) (Kadlec et al, 1993). The concentration of pollutants in a PFR is represented by the ideal first order irreversible steady-flow kinetic reaction presented in equation 3.32 (Tchobanoglous and Burton, 1991, Kadlec et al, 1993 and Kadlec and Knight, 1996):

$$\frac{C_o}{C_i} = \exp[-\mathrm{K}(HRT)] \qquad (3.32)$$

where C_o = Concentration of pollutant in the effluent, g/m^3

K = first order rate constant, 1/days

The concentration of pollutant in a CSTR is described by a first order kinetic equation presented in equation 3.33 (Tchobanoglous and Burton, 1991, Kadlec et al, 1993 and Kadlec and Knight, 1996):

$$\frac{C_o}{C_i} = \frac{1}{1 + K(HRT)} \qquad (3.33)$$

Tracer studies conducted on both VSB and FWS CW have proved that CW do not strictly behave as ideal reactors, instead they tend to behave as a combination of the two states (Kadlec et al, 1993, Netter, 1994, Reed et al, 1995, Kadlec and Knight, 1996, Batchelor and Loots, 1997 and EPA, 2000). The assumption that a constructed wetland behaves ideally can lead the designer to gross errors (Kadlec et al, 1993). Many tracer studies conducted did, however, show that the flow tended to be closer to plug flow than to completely mixed (Reed et al, 1995) and equations to describe this non-ideal intermediate case have been developed (Equation 3.34) (Kadlec et al, 1993):

$$\frac{C_i}{C_o} = \exp[-K(HRT)] \cdot \exp\left[\frac{D_c}{uL}[K(HRT)]^2\right] \qquad (3.34)$$

where D_c = dispersion coefficient, m^2/d

Equation 3.34 is valid for CW with very low dispersion, where plug flow is predicted to prevail, such as almost all VSB systems (Kadlec et al, 1993). Where very large dispersion can take place, such as in open water areas experienced in FWS systems, equation 3.34 cannot be used (Kadlec et al, 1993). The evaluation of the dispersion coefficient has proven to be difficult and the use of this model has been limited (Reed et al, 1995). Another method of modelling the non-ideal combined state was proposed by Kadlec et al. (1993) and further developed by Kadlec and Knight (1996). Their model characterises the CW as a series of continuously stirred and plug flow reactors. The constructed wetland is divided into compartments each of which is characterised by specific boundary conditions and the type of flow conditions expected to dominate; plug flow in subsurface flow and densely vegetated FWS system areas and mixed flow in open water areas (Kadlec et al, 1993). The boundary conditions for each compartment are derived from internal performance data, thus influent and effluent pollutant concentrations for each compartment is required. By using this approach it is possible to fit a combination of PFR and CSTR to any tracer study curve, once the wetland is operational. Trying to predict the correct combination for CW to be designed is much more difficult (Reed et al, 1995). This approach also requires internal performance data. This model would in effect behave and predict similar final effluent quality to that from the simpler PFR with first order kinetics (Reed et al, 1995). It is important to understand that the above equations are steady state equations, where losses and gains through evapotranspiration, precipitation and infiltration are balanced. Some CW will experience significant losses or gains depending on their climatic location and liner design (Kadlec and Knight, 1996). Under these circumstances the first order PFR irreversible reaction may be developed into equation 3.35 (Kadlec and Knight, 1996):

$$\frac{[(P-ET)C_o - PC_P + KC_o]}{[(P-ET)C_i - PC_P + KC_i]} = \left(\frac{[Q_i + (P-I-ET)\cdot W \cdot L]}{Q_i}\right)^{-\left(1+\frac{K+I}{P-I-ET}\right)} \qquad (3.35)$$

where C_P = concentration of pollutant in rain, g/m^3

The above equations are the bases of the design models used for CW. However they vary slightly from pollutant to pollutant. Usually the design of a constructed wetland is dictated by whichever pollutant requires the largest HRT in order to achieve treatment goals. The first order kinetic approach is usually applied to biodegradable organics (BOD) and nitrogen removal. Although extensive works to increase nitrogen removal efficiency in CW are avaliable (Reed et al, 1995, Cooper, 1999, Platzer, 1999 and Cooper et al, 1999), CW do not have the ability to consistently remove high concentrations of ammoniacal nitrogen to specified discharge limits (EC/EWPCA, 1990, Robinson et al, 1993, Van Oostrom and Russel, 1994, Reed et al, 1995, Kadlec and Knight, 1996, Robinson and Barr, 1998 and NADB, 2000). CW have been used to remove other pollutants such as heavy metals (Robinson et al, 1998), phosphorus and solids (Robinson et al, 1993, Reed et al, 1995, IAWQ, 1994, 1995, 1997, 1999, Kadlec and Knight, 1996, Robinson and Barr, 1998, Wood, 1999 and EPA, 2000). The removal mechanisms of these pollutants cannot be explained by first order kinetics. Solids removal or more specifically Total Suspended Solids (TSS) removal is due to physical processes and is influenced only by temperature (Reed et al, 1995) and design for TSS removal is usually done using regression models (Reed et al, 1995) which will be explained below. This research focused primarily on organics removal. Various authors have published different modifications of the ideal first-order irreversible steady-flow PFR reactions for BOD removal. However they tend to fall into two distinct categories, where either the fate of the different constituents forming the

BOD are considered (equation 3.36) (Tchobanoglous and Burton, 1991, Tchobanoglous, 1993, Reed et al, 1995 and Wood, 1999) or they are lumped together (equation 3.37) (EC/EWPCA, 1990 and Cooper et al, 1996). Other factors affecting the model include background or residual BOD in the effluent (Reed et al, 1995 and Kadlec and Knight, 1996).

$$\frac{C_e}{C_i} = A_{BOD}[-0.7 \cdot \mathrm{K}_{BOD} \cdot (A_V)^{1.75} \cdot HRT] \qquad (3.36)$$

where A_{BOD} = fraction of BOD not removed by settling at the head of the CW

A_V = specific surface area avaliable for microbial activity, m^2/m^3

K_{BOD} = BOD temperature dependant first order rate constant, 1/d

$$\frac{C_e}{C_i} = \exp(-\mathrm{K}_{BOD} \cdot HRT) \qquad (3.37)$$

It is possible that some of the residual BOD within the constructed wetland is produced, from the decomposition of plant litter and other organics (Reed et al, 1995); this background BOD has been estimated to range from 2 to 7 mg/l (Reed et al, 1995), however, a wider range of values have been estimated, ranging from 1 to 15 mg/l (Kadlec and Knight, 1996). Thus equations 3.36 and 3.37 cannot be used if an effluent concentration below the background BOD value is required (Reed et al, 1995). Equation 3.36 is limited by the difficulty in measuring the A_{BOD} and A_V factors (Reed et al, 1995). An A_{BOD} value of 0.5 for primary sewerage effluents, 0.7 to 0.8 for secondary sewerage effluents and 0.9 or higher for highly treated tertiary sewerage effluents has been proposed by Reed et al (1995). A_V on the other hand is more difficult to measure especially in FWS CW, a value of 15 to 16m^2/m^3 for FWS systems and 140 to 150m^2/m^3 for VSB systems was proposed by Reed et al (1995). Researchers have tried to develop theoretical models to approximate A_V (Kadlec and Knight, 1996, Polprasert et al, 1998 and Khatiwada and Polprasert, 1999b). Polprasert et al (1998) proposed a theoretical value of A_V = 10000 m^2/m^3 and a predicted value of A_V = 4.4 m^2/m^3 for FWS CW, Kadlec and Knight (1996) proposed a theoretical value of A_V = 360 m^2/m^3 and A_V = 24000 m^2/m^3 for VSB systems. There is a wide range of published values for the BOD rate constant K (Table 3.2). K Is presented in two ways in the literature and the reader must be aware of this. The common way of expressing K is with the direct units of 1/d while for CW it may also be expressed in m/d. When the rate constant is expressed in units of m/d, it implies that the K, in terms of the units 1/d, has been multiplied by the saturated bed depth and bed porosity (EC/EWPCA, 1990 and Kadlec and Knight, 1996).

Table 3.2: Published values of K_{BOD}.

K_{BOD5} (units as shown)	Comment and References
0.09 m/d	Average of 20 FWS systems in the US, Kadlec and Knight (1996)
0.49 m/d	Average of 14 VSB systems in the US, Kadlec and Knight (1996)
0.083 m/d	From 49 systems in Denmark, Cooper et al (1996)
0.067 to 0.1 m/d	In the UK, Cooper et al (1996)
0.06 m/d	For secondary systems in the UK, Cooper et al (1996)
0.31 m/d	For tertiary systems in the UK, Cooper et al (1996)
0.1 m/d	Recommended value by EC/EWPCA, EC/EWPCA (1990)
1.96 1/d	Average of 14 VSB systems in the US, Kadlec and Knight (1996)
$(1.104)(1.06)^{(T-20)}$ 1/d	Developed for VSB systems, Reed et al (1995)
$(0.86)(1.1)^{(T-20)}$ 1/d	Developed for VSB systems (Gravely sand medium), Tchobanoglous and Burton (1991)
$(1.35)(1.1)^{(T-20)}$ 1/d	Developed for VSB systems (Coarse sand medium), Tchobanoglous and Burton (1991)
$(1.84)(1.1)^{(T-20)}$ 1/d	Developed for VSB systems (Medium sand medium), Tchobanoglous and Burton (1991)
$(0.678)(1.06)^{(T-20)}$ 1/d	Developed for FWS systems, Reed et al (1995)
$(0.0057)(1.1)^{(T-20)}$ 1/d	Developed for FWS systems, Tchobanoglous and Burton (1991)
$(0.15)(1.04)^{(T-20)}$ 1/d	Developed for FWS systems, EPA (2000)

Multiple regression approach

This approach uses a multiple regression analysis of performance data from operating systems to estimate design criteria (Reed et al, 1995). Detailed databases are required for such analyses. There are databases avaliable, such as the *North American Treatment Wetlands Database* version 2.0 (NADB) (EPA, 2000), the *Danish Wetland Treatment System Database* (Kadlec and Knight, 1996) and the *European Wetland Treatment System Database* (Kadlec and Knight, 1996). The literature, however, encourages designers not to use such databases for direct design or modeling (Kadlec and Knight, 1996 and EPA, 2000) as the information comes from a diverse source and cannot always be verified (Kadlec and Knight, 1996). The databases should only be used to summarise characteristics and trends in the wetland systems (Kadlec and Knight, 1996 and EPA, 2000). The diversity of CW can be seen in the regression plots presented in the literature of SLR versus effluent concentrations or removal efficiency (Figure 3.12) (EPA, 2000). The scatter of results makes it almost impossible to confidently determine a trend for the regression analysis.

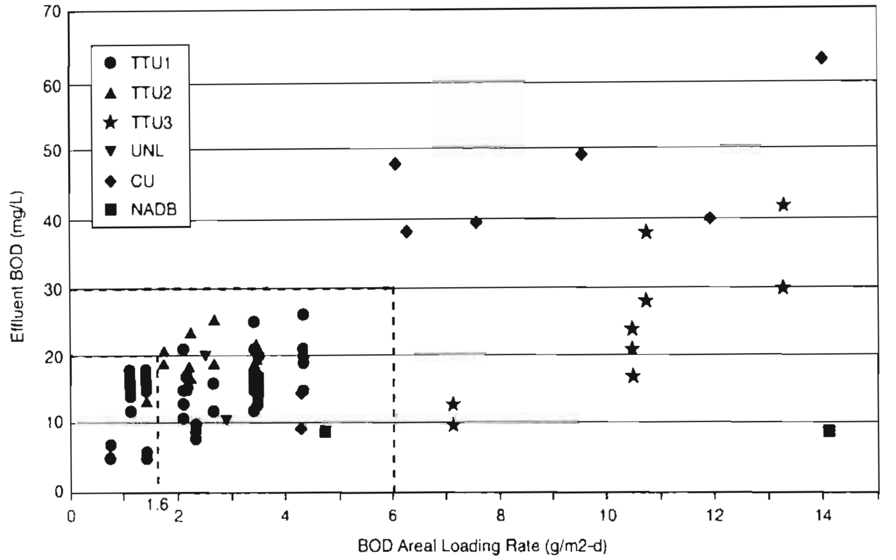


Figure 3.12: Effluent BOD vs. SLR for VSB CW treating septic tank and primary effluents, taken from EPA (2000); (TTU = Studies conducted by Tennessee Technnological University; CU = Studies conducted by Clarkson University; UNL = Studies conducted by the University of Nebraska – Lincoln; NADB = Data from the North American Wetlands for Water Quality Treatment Database) (EPA, 2000).

Regression analysis for TSS removal design is probably the most logical method as the removal of TSS in CW is not likely to be the limiting design parameter for sizing the CW, and more of a conceptual design approach is undertaken (Reed et al, 1995). Reed et al. (1995) proposed the regression equations 3.38 (FWS CW) and 3.39 (VSB CW) as conceptual design models for TSS removal:

$$C_e = C_i[0.1139 + 0.00213(HLR)] \tag{3.38}$$

$$C_e = C_i[0.1058 + 0.0011(HLR)] \qquad (3.39)$$

The models are limited to a HLR of 0.4 to 75 cm/d and should not be used to predict a final effluent less that 5 mgTSS/l, due to background TSS concentrations within the CW (Reed et al, 1995). Knight et al (1993) proposed a regression equation (equation 3.40) for BOD_5 removal, based on 324 complete data records from the NADB:

$$BOD_e = 0.97 \cdot HLR + 0.192 \cdot BOD_i \qquad R^2 = 0.72 \qquad (3.40)$$

where BOD_I = Influent BOD_5 concentration, mg/l

BOD_e = Effluent BOD_5 concentration, mg/l

Surface area or volumetric loading approach

Surface area loading design methods are used in land treatment systems where the polluted water is applied uniformly over the whole treatment area; this is not the case in CW where the influent is usually applied uniformly across the inlet of the system (Reed et al, 1995). The SLR method does not take the water depth, the related HRT and temperature into account (Reed et al, 1995). However, a rational design approach can yield plausible results (EPA, 2000). Although the EPA (2000) considers volumetric loading rates for VSB system design, because the actual pore volume is seldom known and the HRT may not be directly related to the theoretical value due to preferential flow, the use of VLR for design is limited (EPA, 2000). A comparison between the design of constructed wetland areas estimated from different design models and the surface area loading method is available (EPA, 2000). The results showed that the other design approaches estimated significantly smaller systems than that using the surface area loading approach (EPA, 2000). Table 3.3 gives a summary of recommended BOD_5 SLR and ranges of actual BOD_5 SLR used in CW design.

Table 3.3: Published BOD_5 surface area loading rates.

BOD_5 SLR ($g/m^2/d$)	Comments and References
6	For VSB systems, EPA (2000)
5.3	For VSB systems, TVA (1993)
6-7	Upper limit for FWS systems to prevent odours, Stowell et al (1985)

4	For fully vegetated zone in FWS systems (effluent concentration of 30 mg/l), EPA (2000)
4.5	For FWS systems with significant open waters between fully vegetated zones (effluent concentration of <20 mg/l), EPA (2000)
6	For FWS systems with significant open waters between fully vegetated zones (effluent concentration of 30 mg/l), EPA (2000)
8	For a population equivalent (pe) of 40gBOD$_5$/d and a recommended wetland surface area of 5m^2/pe, EC/EWPCA (1990)
0.004-11.5 (Ave. 1.3)	Ranges from the NADB, for FWS systems, NADB (2000)
0.45-41.24 (Ave. 5.4)	Ranges from the NADB, for VSB systems, NADB (2000)
0.51-130.73 (Ave. 11.96)	Ranges from the Danish database, for VSB systems, Kadlec and Knight (1996)
0.50-4.63 (Ave. 2.97)	Ranges from the British database, for VSB systems, Kadlec and Knight (1996)
6.65	For VSB and FWS systems, Tchobanoglous and Burton (1991)

3.3 Constructed wetland bedding media and layout

3.3.1 Sub surface flow systems

The selection of the VSB design media is dictated by several functional reasons. The selected media must maintain the required range of design hydraulic conductivity through out the life of the system, in order to maintain sub-surface conditions and eliminate the possibility of overland flow (EC/EWPCA, 1990, Reed et al, 1995, Wood, 1996 and EPA, 2000). Those VSB systems that have used soils as bedding media have suffered from this problem due to the clogging of the media, by bacterial growth, precipitation and sedimentation, and incorrect hydraulic design (EC/EWPCA, 1991 Reed et al, 1995, Wood, 1996, Blazejewski and Murat-Blazejewska, 1997 and EPA, 2000). The EPA (2000) recommended that soil and sand media should be avoided. The selected media should also function as a rooting material for the vegetation (EC/EWPCA, 1990, Reed et al, 1995, Kadlec and Knight, 1996 and EPA, 2000). Cattails, reeds and bulrushes have been shown to grow in a variety of soils and fine gravels (Reed et al, 1995). The EC/EWPCA (1990) recommended that if a gravel medium was selected, which is lacking in nutrients, small amounts of fertiliser should be used at planting time and once the plants were established the influent feed should contain enough nutrients to sustain the vegetation (Robinson et al, 1993). The bedding type media may also have an effect on root growth, however, slower plant growth in gravel based beds has little adverse effect on the VSB performance (Robinson and Barr, 1998). Table 3.4 gives a summary of the optimal root depth of specific emergent species (Tchobanoglous and Burton, 1991, Reed et al, 1995 and Nuttall et al, 1997).

Table 3.4 Plant Species and their optimal root depth

Emergent Species	Optimal Root Penetration (cm)
Cattail	30
Bulrushes	76
Phragmites australis	60

Wood (1999) found that *Typha* root systems, in South African gravel based systems, were limited to the top 20 cm of the bed. Kadlec (1989) showed that plant root systems do not penetrate deeper than 10 to 15 cm in gravel bed systems. The EPA (2000) specifies a root zone of 15 to 25 cm in gravel bed systems. The gravel bed hydroponic systems use a bed depth of 25 cm (Wood, 1999). Robinson et al. (1993) found that lowering the water level in gravel beds at regular intervals encourages deep rhizome growth. The media should also provide surface area for microbial growth and in fact any wetted portion of the medium can become a suitable surface for the development of microbes (EPA, 2000). The media should also function as a filter and trap for particulate matter and help to evenly distribute and collect flow at the inlet and outlet (EC/EWPCA, 1990 and EPA, 2000). The media may also be selected in order to obtain a desired treatment effect (Wood, 1999), such as phosphate removal (EC/EWPCA, 1990, Reed et al, 1995, Kadlec and Knight, 1996, Wood, 1999 and EPA, 2000) and ammonium removal (Reed et al, 1995). Media with high iron or aluminum have a higher potential for phosphorous binding (EPA, 2000) and media with some clay minerals have an ion-exchange capacity that may contribute to the removal of ammonium (Reed et al, 1995). Both these removal mechanisms are limited and are usually exhausted during the first few months of operation (EPA, 2000).

The bed layout is divided into three zones (Figure 3.13); the inlet zone, treatment zone and the outlet zone (typical inlet and outlet designs are presented later in the text) (EC/EWPCA, 1990 and EPA, 2000).

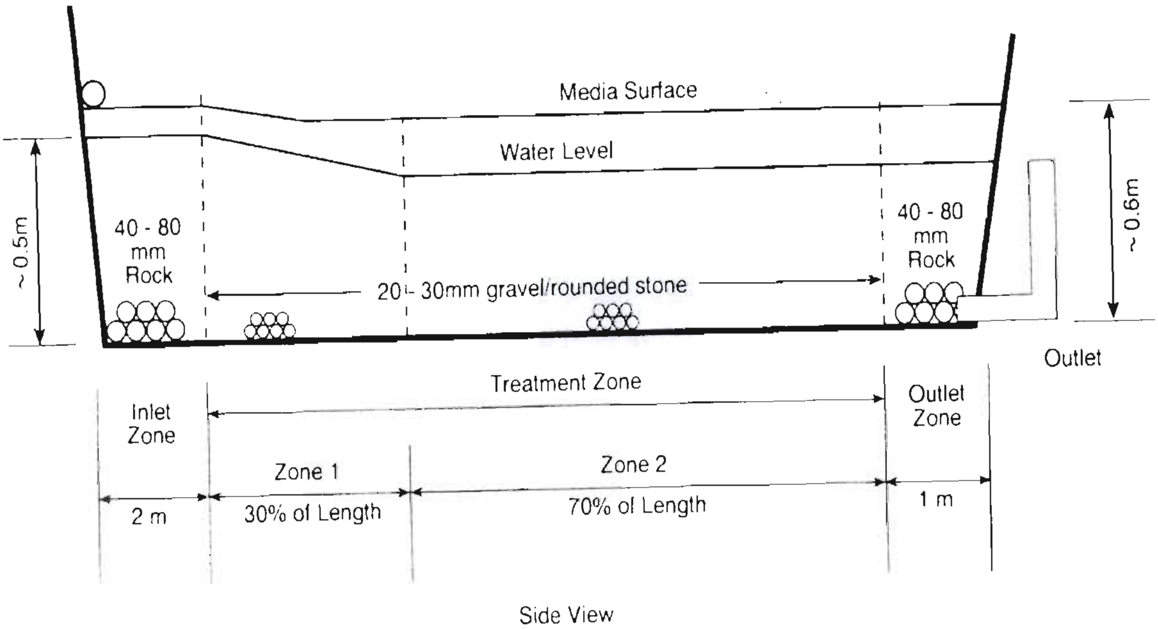


Figure 3.13: VSB system zones, from EPA (2000).

According to the EPA (2000) the inlet zone should be 2 m long and the outlet zone should be 1 m long. The EC/EWPCA (1990) recommended using 0.5 m long inlet and outlet zones. The EPA (2000) recommended an average diameter gravel for the treatment zone between 20 and 30 mm and between 40 and 80 mm for the inlet and outlet zones. The EC/EWPCA (1990) recommends the use of evenly graded stones in the range of 50 to 200 mm for the inlet and outlet zones, a size for the treatment zone was not specified. The EPA (2000) and EC/EWPCA (1990) recommended typical designs for the inlet and outlet structures. The main function of the inlet and outlet structures is to evenly distribute and collect the influent and effluent, across the entire width of the VSB (EC/EWPCA, 1990, Reed et al, 1995, Kadlec and Knight, 1996 and EPA, 2000). The use of castellated weirs is not advised for inlet structures (EC/EWPCA, 1990) that should be designed to allow for inspection and clean-out if necessary (EPA, 2000). The outlet arrangement should allow for water level manipulation in the VSB and full drainage (EC/EWPCA, 1990 and EPA, 2000), the use of stop logs, to achieve this, is not recommended as they are difficult to operate and give course control (EC/EWPCA, 1990).

3.3.2 Free water surface systems

The FWS system bedding medium is mainly selected according to its ability to behave as a vegetation substrate. The major flow path in FWS systems is above the soil surface and the hydraulic conductivity of the bedding medium is not of importance, thus soils with high humic and sand components may be used (EPA, 2000). Vegetation generally used in CW reproduces asexually via rhizomes. Studies have proven that vegetation growth and rhizome migration are more rapid in these types of soils (EPA, 2000). The EPA (2000) recommended using a well loosened loamy soil at least 150mm deep, depending on the type of liner material used. In systems with large water depth fluctuations, a sandy loam or gravely loam should be used as less dense substrates, such as a silty loam, have been found to cause large vegetation mats to float when water level fluctuations occur, reducing treatment performances (EPA, 2000). As with VSB systems the substrate may be selected for a specific treatment need, however, unlike VSB systems where the polluted water passes through the medium's structural matrix, the only contact the polluted water has with the medium is along the surface and the removal mechanisms of the medium's surface matrix are easily exhausted (Reed et al, 1995 and EPA, 2000). The main functions of the inlet and outlet systems (typical inlet and outlet designs will be discussed later in the text) for the FWS system are the same as for the VSB systems, however the design will be slightly different. The use of large stone at the inlet and outlet zones is not required, unless the outlet system is submerged, and then it should be designed as for VSB systems. The outlet system should allow for water level manipulation and complete draw down. Encroachment of vegetation may clog the inlet and outlets with plant litter and detritus (Plate 3.1) (EPA, 2000). This may be overcome by 1 m wide deep-water zones (1-1.3m deeper than the bottom of the rest of the wetland) adjacent to the inlet and outlet structures (EPA, 2000).

Plate 3.1: Over grown outlet structure, Kwazamokuhle FWS CW, Gauteng, South Africa.

3.3.3 Typical inlet and outlet designs

The EPA (2000) and EC/EWPCA (1990) have proposed typical examples of constructed wetland inlet (Figures 3.14a and b, 3.15a and b) and outlet designs (Figures 3.16a and b).

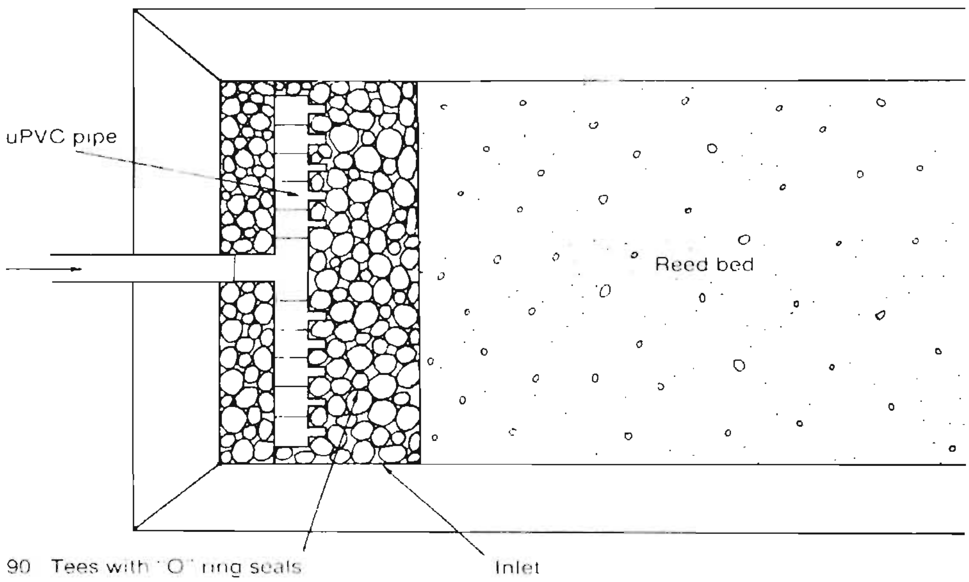


Figure 3.14a: Top View of inlet with swivelling tees.

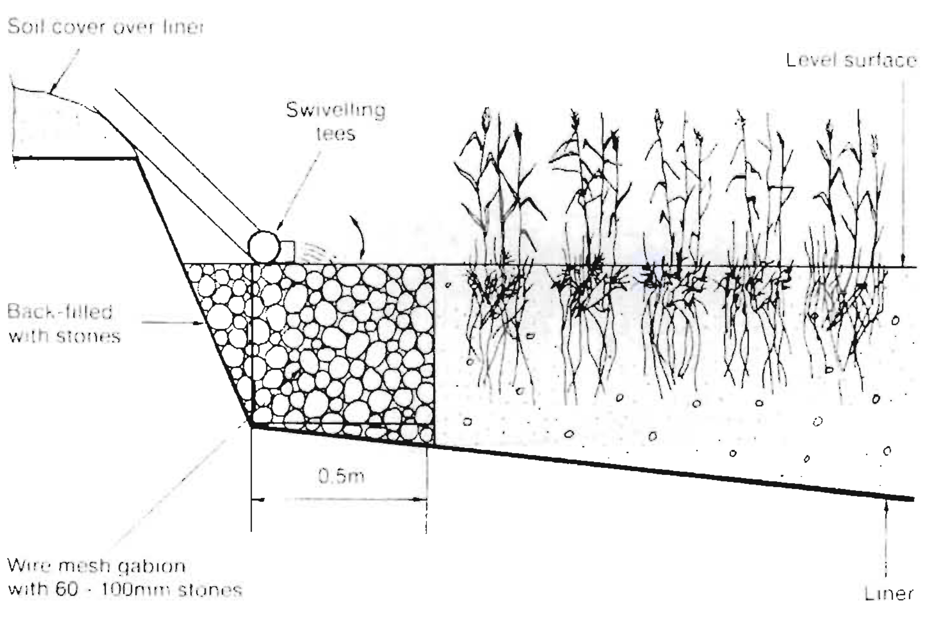


Figure 3.14b: Side View of inlet with swivelling tees.

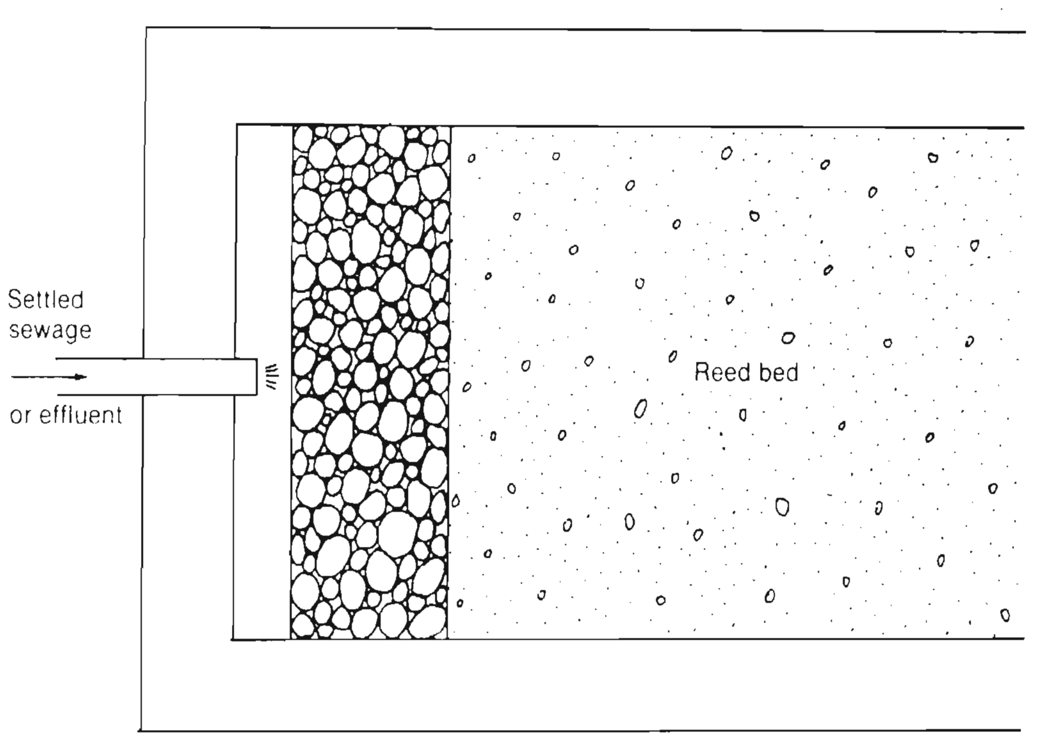


Figure 3.15a: Top View of inlet with feed behind gabion.

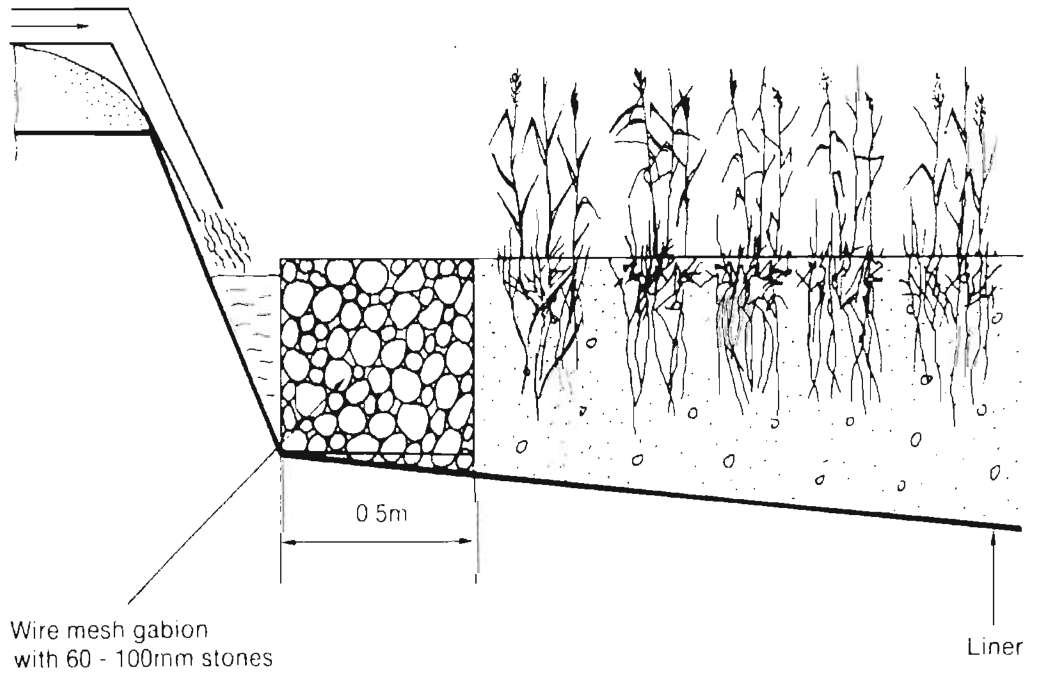


Figure 3.15b: Side View of inlet with feed behind gabion.

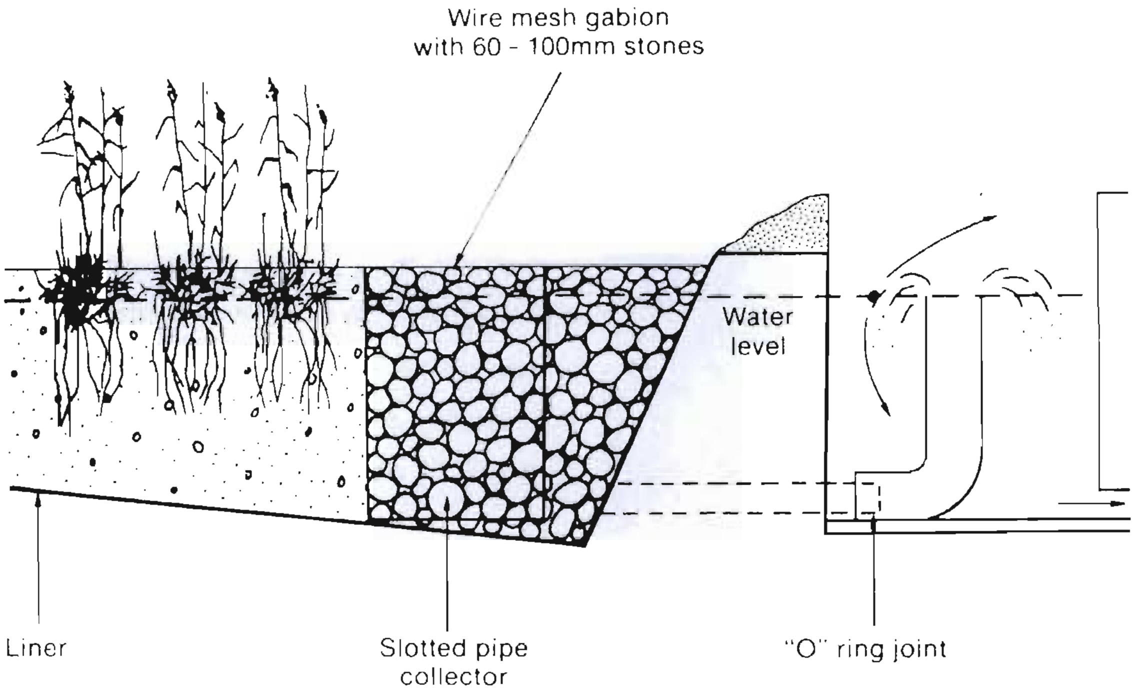


Figure 3.16a: Side View of outlet system with level controlled by interchangeable section.

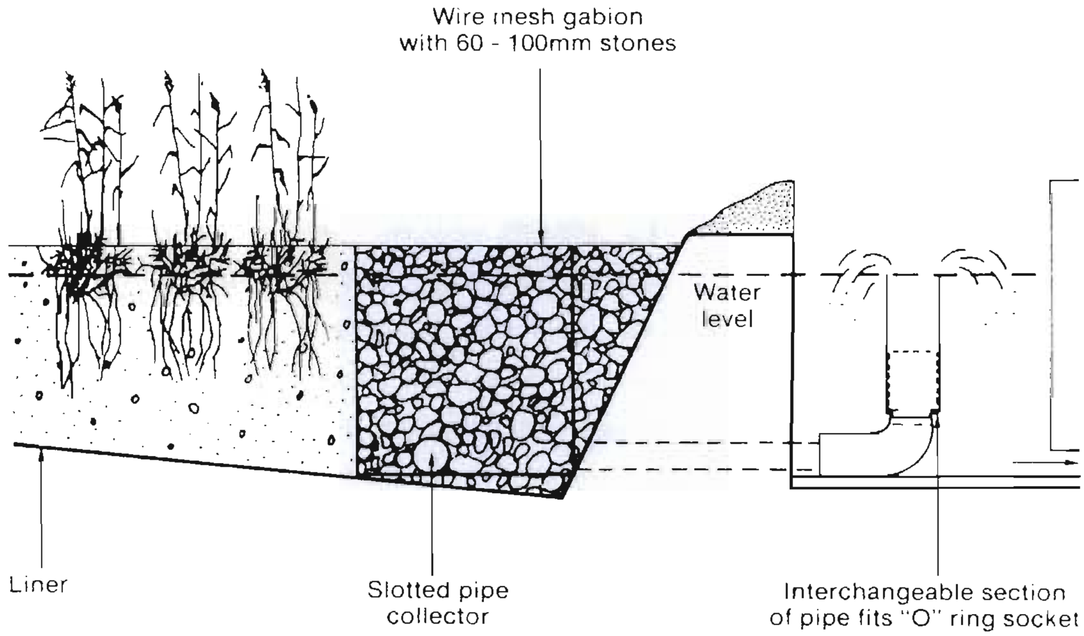


Figure 3.16b: Outlet system with 90°-elbow arrangement.

3.4 Constructed wetland vegetation

Macrophytes, which are the most commonly used plant species for CW, are vascular plants that have easily visible tissues. Unlike other types of plants that require well oxygenated soils, macrophytes are adapted to the saturated hydrological conditions experienced in CW, which causes predominately anaerobic or anoxic conditions within the rhizoshpere. Macrophytes have developed 'aerenchymous' (Vascular) tissues (Plate 3.2) that enable the transport of gasses to and from the rhizomes. 'Lenticels' (small openings) on the above water portions of the plant allow atmospheric oxygen into the 'lacunal' system (network of vascular tissues) for transport of oxygen to the rhizomes (Kadlec and Knight, 1996). Many types of macrophytes have been used in CW, generally the most common are *Phragmites spp., Typha spp., Scirpus spp., Juncus spp.* and *Carex spp.* (Reed et al, 1995). Research conducted by Wood (1999) pointed out that the typical species used in South African CW are *Typha spp.* and *Frogmouths. Phragmites spp.* is the most commonly used macrophyte in Europe and the UK in particular (Robinson et al, 1993, Robinson and Barr, 1998 and Reed et al, 1995) while *Scirpus spp.* and *Typha spp.* are the most commonly used in the United States (Reed et al, 1995). Certain species of this commonly used group of macrophytes (i.e. *Phragmites spp.)* are regarded as 'noxious' weeds in certain countries, such as south-western Australia, where other species such as Typha spp. and Juncus spp. are used (Chambers and McComb, 1994). Macrophytes have a high environmental value, for both humans and animals, they are aesthetically pleasing, they have a 'high habitat' value, by providing nesting and food source for animals (Kadlec and Knight, 1996). Macrophytes reproduce both asexually, through vegetative growth of rhizomes (Plate 3.3) and sexually, and may be either relocated by rhizome sections or established by seeds, preferably in a green house and then relocated as seedlings (EC/EWPCA, 1990 and Kadlec and Knight, 1996).


Plate 3.2: Aerenchymous tissue

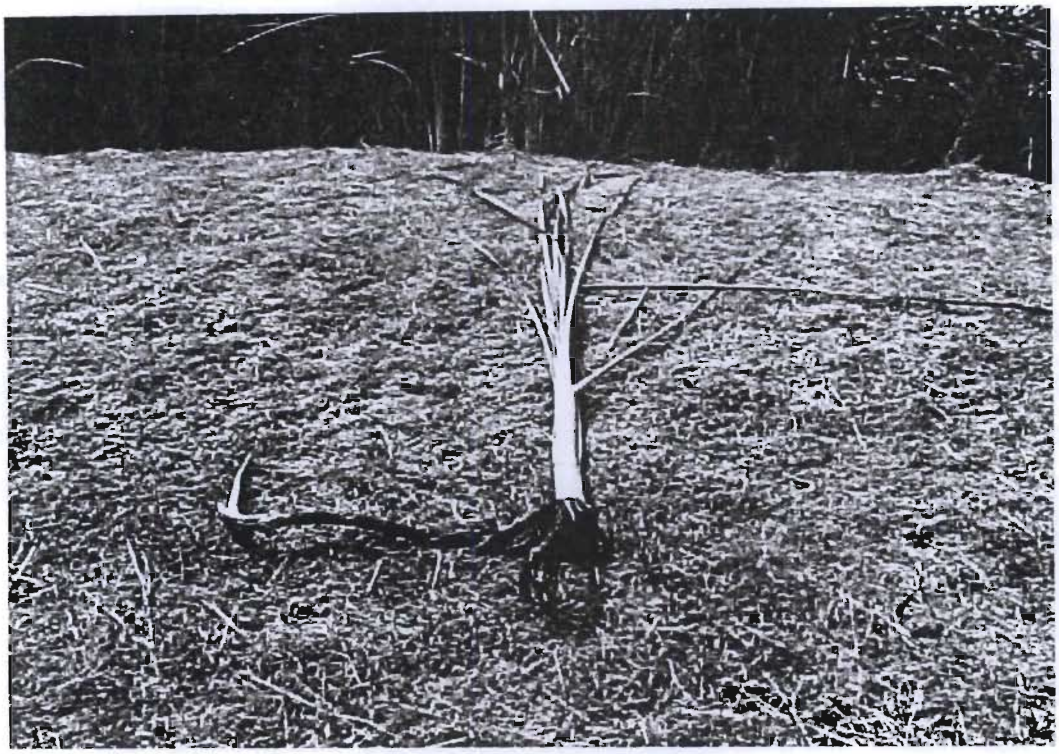


Plate 3.3: Extracted rhizome showing vegetative growth.

The compatibility of an emergent macrophyte species for a specific constructed wetland is based on its environmental value, the availability and the species treatment ability. There should be no legal restriction of the use of the selected species, and 'alien species' or 'noxious weeds' should be avoided (Chambers and McCombe, 1994). A sufficient source of 'indigenous' plants coupled with a planned method of relocation and establishment should always be ensured before construction (Chambers and McCombe, 1994). The use of indigenous species aids in blending the system in with the surroundings, increasing the systems aesthetics and economic viability (Brix, 1994 and Wood, 1999). Certain designs may also be based on 'habitat values' in addition to treatment function; this implies a greater diversity of species with an emphasis on food and nesting values for birds and aquatic life (Brix, 1994 and Reed et al, 1995).

The main criterion for the selection of a macrophyte species in a CW is its treatment capability or role. CW are assumed, in design, to be attached-growth biological reactors and are designed using first order kinetics (EC/EWPCA, 1990, Reed et al, 1995, Kadlec and Knight, 1996 and EPA, 2000). In doing so the influence from the macrophyte species on the systems treatment performance seems to be irrelevant (Wetzel, 1993). Under these assumptions, however, the only use for macrophyte species would be for aesthetics and habitat values, and if so the use of other trickling or filtration systems that biologically remove organic material at much higher loading rates than CW would be much more efficient, manageable and economical. Numerous studies measuring treatment efficiencies of systems with and without plants have concluded that performance is higher when plants are present (Wetzel, 1993 and Kadlec and Knight, 1996). However, other more recent studies have shown no significant difference in performance (EPA, 2000). In a pilot scale study, Okurut et al (1999) found that wetland cells planted with *Cyperus papyrus* and *Phragmites mauritianus* performed better in organics removal than an unplanted control. However, ammoniacal nitrogen removal efficiencies were higher in the unplanted control than in the planted cells. Reed et al (1995) found that planted VSB systems, in Santee California, performed better in both organics and ammoniacal nitrogen removal, compared with an unvegetated bed. A notable difference in suspended solids removal was not evident. Hosokawa and Furukawa (1994) pointed out, by using numerical models, that macrophyte stems reduce the polluted water flow in FWS coastal reed beds, reduction in flow increased with increasing water depth, thus promoting the settlement of suspended solids. Macrophytes are believed to play a role in the treatment performance of CW in a number of ways. The structural elements of the macrophyte in contact with the polluted water (Figure 3.17) provide attachment surfaces for the microbes that mediate most of the pollutant reductions (Kadlec and Knight, 1996).

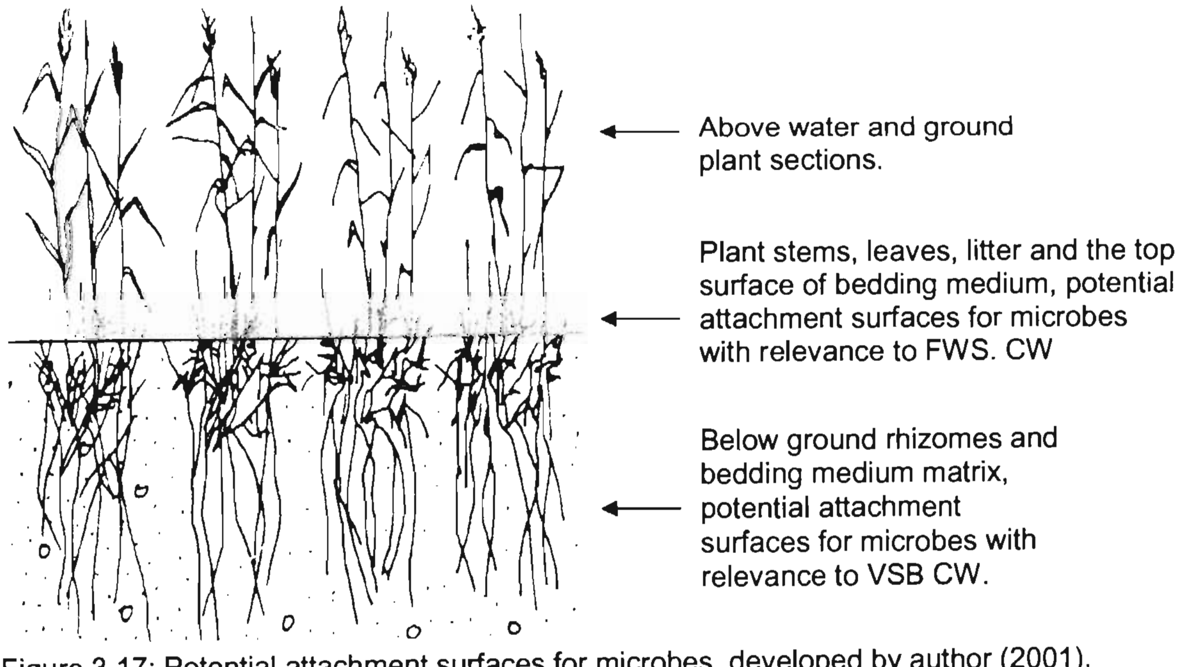


Figure 3.17: Potential attachment surfaces for microbes, developed by author (2001).

In FWS CW this refers to the above ground living plant parts and the decomposing plant litter in the water column, as well as the top surface of the bedding medium, while in VSB this refers to the rhizomes in contact with the polluted water and the matrix of the bedding medium (Reed et al, 1995 and Kadlec and Knight, 1996). In VSB systems, typically 0.6 m deep (EC/EWPCA, 1990 and EPA, 2000) with gravel media, root development throughout the entire bed depth has been proven to be suppressed, forming preferential flow paths under the root zone (EPA, 2000). Nutrient uptake of macrophytes has been found only to be of quantitative importance in low-loaded systems (Brix, 1994). These nutrients are, however, released back into the system during plant senescence (Wetzel, 1993) and permanent removal is only accomplished if the macrophytes are harvested (Reed et al, 1995). Harvesting in CW is not practical due to problems of access and labor costs (Reed et al, 1995). The majority of the nutrients removed by the macrophyte are also located in the underground tissues especially at times of maximum biomass when harvesting would usually take place, much of the nutrient content is translocated to the rhizomes for storage and use in new growth. Thus, the removed nutrients and organic carbon generated by the macrophytes is eventually returned to the system and reduces its efficiency (Wetzel, 1993). It has already been established that CW are inefficient in high ammoniacal nitrogen concentration removal (EC/EWPCA, 1990, Robinson et al, 1993, Van Oostrom and Russel, 1994, Reed et al, 1995, Kadlec and Knight, 1996 and NADB, 2000). The release of organic carbon back into the system via plant senescence has, however, the potential to support denitrification if the influent nitrogen is in the form of nitrate nitrogen and temperature conditions are favorable (Van Oostrom and Russel, 1994 and Reed et al, 1995). Van Oostrom and Russel (1994) showed that a mat of floating *Glyceria maxima* promoted a denitrification rate of 3.8 $g/m^2/d$ in a FWS pilot system at 20^0C. Approximately 5-9 g of carbon is required to denitrify 1 g of nitrate nitrogen. Theoretically the carbon available in emergent macrophytes is adequate to support denitrification (Reed et al, 1995). The anoxic conditions within both FWS and VSB systems could favour denitrification, however, FWS wetlands would have an advantage as the polluted water is in direct contact with the decaying plant litter, and the rate of decay would be much higher than that in VSB systems as the litter is always in the water (Reed et al, 1995).

The mechanisms for the internal oxygen transport mentioned earlier are passive molecular diffusion (Figure 3.18, Brix, 1993b), caused by concentration gradients within the lacunal system, and convective flow through the lacunal system (Figure 3.19, Brix, 1993b) (Brix, 1993b, Brix, 1994, Stengel, 1993 and Kadlec and Knight, 1996).

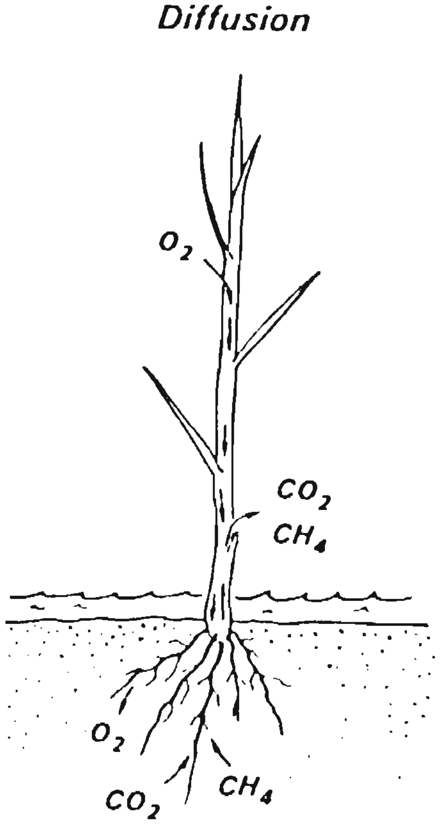


Figure 3.18: Oxygen transport by molecular diffusion.

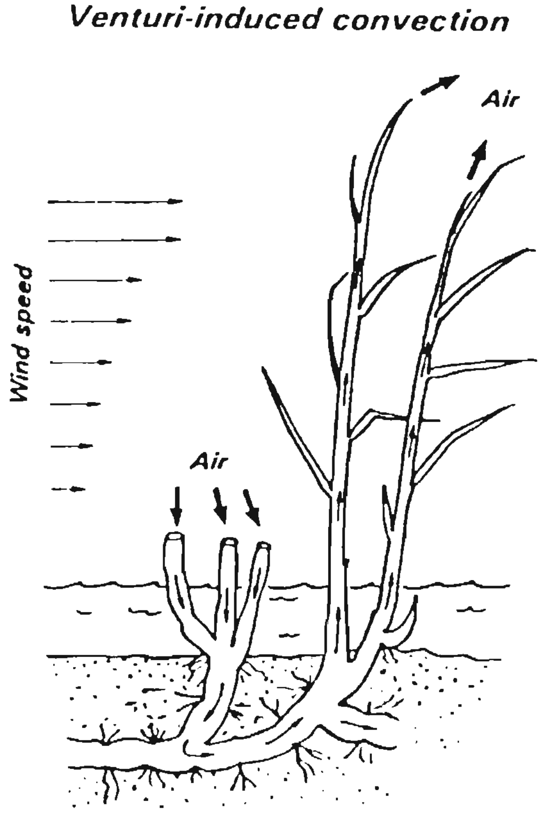


Figure 3.19: Oxygen transport by venturi induced convection.

The transportation of oxygen into the rhizome is of concern for the design of VSB wetlands. It is believed that a fraction of this oxygen can 'leak' out of the rhizome and create aerobic microzones around the rhizome, in a predominantly anaerobic or anoxic environment. These aerobic microzones would support aerobic microbes that would aid in the oxidation of organic material (Brix, 1993b, 1994). Biological reactions in water saturated soils result in the release of reduced substances that may, at suitably high concentrations, be toxic to the rhizomes; the 'leakage' of oxygen from the rhizomes would help in detoxifying these reduced products. The oxygen release rates depend on internal oxygen concentration, the oxygen demand of the surrounding medium, the permeability of the rhizome walls. Laboratory measured release rates range from 0.5 to 5.2 g/m^2/d (Brix, 1993b). Macrophytes tend to limit the radial 'leakage', in an attempt to conserve internal oxygen and to allow oxygenation of the root tip; although 'leakage' does occur at the root tip for detoxifying purposes (Brix, 1993b). Reed et al (1995) found strong correlations between treatment performances, for biodegradable organics and ammoniacal nitrogen, and depth of root penetration. The VSB system design depth is normally based on potential root depth of penetration, and the presence of aerobic microzones around the rhizomes (Reed et al, 1995). The depth of root penetration is not of concern for FWS CW, where atmospheric diffusion is the primary mechanism of reaeration, typically leading to aeration of only the top few millimeters of the water surface (Brix, 1993b). The oxygen transfer from the atmosphere is driven by concentration gradients developed between the water column and the atmosphere when dissolved oxygen levels within the water column are below saturation. This oxygen transfer depends on temperature and the rate of oxygen consumption due to aerobic microbial processes (Kadlec and Knight, 1996). The depth of the FWS wetland is generally designed on the selected vegetation's hydroperiod; common species hydroperiods are summarised in Table 3.5 (Kadlec and Knight, 1996).

Macrophytes in CW are exposed to adverse environmental conditions, Table 3.6 summarises some of the survival conditions for specific species.

Table 3.5: Hydroperiods of selected macrophytes.

Macrophyte	Water Depth (m)	Flooding Duration (%)	Reference
Typha latifolia	0.15-0.6	-	Nuttall et al (1997)
Typha spp.	0.1-0.75	70-100	Kadlec and Knight (1996)
Iris pseudacorus	0.1-0.2	-	Nuttall et al (1997)
Iris spp.	0.05-0.2	50-100	Kadlec and Knight (1996)
Sparganium spp.	0.6	-	Nuttall et al (1997)
Sparganium americanum	0.1-0.5	70-100	Kadlec and Knight (1996)
Scirpus acutus	0.6-2	-	Nuttall et al (1997)
Scirpus spp.	0.1-1.5	75-100	Kadlec and Knight (1996)
Cyperus spp.	0.05-0.5	50-100	Kadlec and Knight (1996)
Carex spp.	0.05-0.25	50-100	Kadlec and Knight (1996)
Phragmites spp.	0.05-0.5	70-100	Kadlec and Knight (1996)
Juncus spp.	0.05-0.25	50-100	Kadlec and Knight (1996)

Table 3.6: Environmental conditions for the survival of specific macrophytes.

Macrophyte	Temp (°C)	Salinity tolerance (g/l)	pH range	Reference
Typha angustifolia	10-30	15-30	4-10	Reed et al (1995)
			3.7-8.5	EPA (1988)
Typha latifolia	10-30	<1	4-10	Reed et al (1995)
			3-8.5	EPA (1988)
Scirpus acutus	18-27	0-5	4-9	Reed et al (1995)
Scirpus validus	18-27	0-5	4-9	Reed et al (1995)
			6.5-8.5	EPA (1988)
Scirpus lacustris	18-27	25	4-9	Reed et al (1995)
Phragmites communis	12-23	<45	2-8	Reed et al (1995)
Phragmites australis	12-33	<45	2-8	Reed et al (1995)
			3.7-8	EPA (1988)
Juncus spp.	16-26	0-25	5-7.5	Reed et al (1995)
Carex spp.	14-32	<0.5	5-7.5	Reed et al (1995)

Macrophytes may be planted in rhizome sections, clumps, using seeds and seedlings grown in greenhouses (EC/EWPCA, 1990 and EPA, 2000). Research conducted by the Institute of Terrestrial Ecology in the UK found that both seedlings and clumps were more successful than the rhizomes method and the use of seedlings was recommended due to their ability to give faster growth results (EC/EWPCA, 1990). Once the macrophyte has been planted, the soil should be maintained in a moist condition; weeds may be controlled by the occasional flooding of the CW, to a depth of about 5 centimetres (EC/EWPCA, 1990, Robinson et al, 1993 and EPA, 2000). Chambers and McComb (1994) showed that plant growth and establishment was the most successful when the water level approximately coincided with the sediment surface, they also concluded that clay substrates and low rates of water movement were less suitable for establishment than sands and higher rates of water movements. Certain macrophyte species tend to form growth gradients along the length of the CW. Edwards et al (1993) concluded that growth gradients for bulrushes did exist along the length of CW with almost more than four times the density near the inlet as the outlet. From an aesthetic point of view the designer needs to be aware of the possible gradients as the appearance of the bed is often viewed by the outsider as a reflection on the beds performance (EC/EWPCA, 1990). The EPA (2000) and EC/EWPCA (1990) recommend planting to take place in spring. The side walls of the CW should be approximately vertical as research has shown that macrophytes do not grow near shallow sloped edges (EC/EWPCA, 1990). The typical spacing of plants during planting is summarised in Table 3.7:

Table 3.7: Typical spacing of macrophytes during planting.

Macrophyte	Spacing	Reference
Not specified	0.3 to 1 m centres	EPA (2000)
Not specified	4 seedlings/m^2	EC/EWPCA (1990)
Not specified	2 rhizomes/m^2	EC/EWPCA (1990)
Cattail	Dense cover in less than one year with 0.6 m centres	Reed et al (1995)
Reeds	Up to 10/m^2	Robinson and Barr (1998)
Bulrush	Dense cover in one year with 0.3 m centres	Reed et al (1995)
Reeds	Very dense cover in one year with 0.6 m centres	Reed et al (1995)
Rushes	Dense cover in one year with 0.15 m centres	Reed et al (1995)
Sedges	Dense cover in one year with 0.15 m centres	Reed et al (1995)

3.5 Constructed wetland microbial communities and wildlife

Due to the abundance of water, natural and constructed wetlands are among the most productive systems in the biosphere (Rogers et al, 1985), favouring a diversity of microbial communities and wild life (Reed et al, 1995, Kadlec and Knight, 1996 and EPA, 2000). The biotic environment of a constructed wetland is highly complex with intraspecific and interspecific relationships between species. Each species occupies different niches within the complex 'food web' of the constructed wetland, starting with the producers and ending with the reducers (Lombard, 1999). Solar radiation is the primary energy source for CW, it drives photosynthesis and warms the wetland waters and media (Kadlec, 1999). It is the ability of the CW biotic components to adapt through genetic diversity and functional adaptation that allows them to use the constituents of the influent polluted water for their growth and reproduction and, in turn to modify and improve the water quality (Kadlec and Knight, 1996).

3.5.1 Procaryotes

The procaryotic group of microorganisms is highly important in CW as they catalyse a number of chemical conversions (Table 3.8) that determine the fate of certain chemicals and influence the nutrient cycle (Rogers et al, 1985, Tchobanoglous and Burton, 1991, Reed et al, 1995 and Kadlec and Knight, 1996). Bacteria not only influence the fate of the influent pollutants but play a vital role in the conversion of internally produced organic material, decomposing and converting it to useable products for higher life forms (Wetzel, 1993 and Kadlec and Knight, 1996). Bacteria in CW may be either free living or attached to surfaces, such as the solid surfaces of plants, decaying organic matter and the bed medium (Rogers et al, 1985, Kadlec and Knight, 1996).

3.5.2 Eucaryotes

Eucaryotes include plants, animals and protists, such as fungi, algae, protozoa and rotifiers (Tchobanoglous and Burton, 1991). Fungi includes yeasts, molds and fleshy fungi which are all heterotrophic protists and mediate the recycling of carbon and other nutrients in CW. They usually colonise on decaying vegetation made available through bacterial decomposition (Kadlec and Knight, 1996). Some fungi (as with bacteria) have been shown to live symbiotically with species of algae and higher plants, capturing dissolved elements and making them available for their host (Kadlec and Knight, 1996). Protozoa, a microscopic protist, and Rotifers, multicellular animals are both free living (motile) and prey on bacteria (Tchobanoglous and Burton, 1991). They use bacteria and particulate organic matter as an energy sources to grow and synthesise new cells (Tchobanoglous and Burton, 1991). Algae are important in CW as they aid in short term nutrient fixation and immobilisation, contribute to the overall cycling of nutrients and play a functional role in microbial establishment (Kadlec and Knight, 1996). The terms related to algae's functional role in microbial establishment are periphyton, aufwuchs and benthic algae (Kadlec and Knight, 1996). Periphyton describes the community of organisms that colonise the available attachment surfaces on vegetation in CW, which is initiated by the growth of filamentous and unicellular species of algae (Rogers et al, 1985 and Kadlec and Knight, 1996). Periphyton has also been found to form the basis of the grazing food chain. However, due to the poor assimilation of higher grazing organisms, much of the ingested periphyton is returned to the system in the grazers faeces (Rogers et al, 1985). Aufwuchs describes the community of organisms that colonise all the available attachment surfaces in CW, such as plant litter, living plants and the bed medium. Benthic algae specifically describes the algae component of the periphyton or aufwuchs (Kadlec and Knight, 1996).

Group	Representative Genera	Comments
Phototrophic bacteria	*Rhodospirillum, Chlorobium*	Members of these genera are nonsymbiotic N fixers
Gliding bacteria	*Beggiatoa, Flexibacter, Thiothrix*	Filamentous bacteria found in activated sludge; *Beggiatoa* oxidizes hydrogen sulfide
Sheathed bacteria	*Sphaerotilus*	Filamentous bacteria implicated in reduced sludge settling rates in sewage treatment plants and common in polluted waters
Budding and/or appendaged bacteria	*Caulobacter, Hyphomicrobium*	Aquatic bacteria growing attached to surfaces with a hold fast
Gram-negative aerobic rods and cocci	*Pseudomonas, Zooglea, Azotobacter, Rhizobium*	*Pseudomonas* spp. denitrifies NO_3^- to N_2 under anaerobic conditions and can oxidize hydrogen gas. *P. aeruginosa* causes a variety of bacterial infections in humans. *Azotobacter* spp. is a nonsymbiotic N fixer; *Rhizobium* is a symbiotic N fixer
Gram-negative facultatively anaerobic rods	*Escherichia, Salmonella, Shigella, Klebsiella, Enterobacter, Aeromonas*	*E. coli* is the predominant coliform in feces; *Salmonella* spp. cause food poisoning and typhoid fever; *Shigella* spp. causes bacillary dysentery; species in the genera *Klebsiella* and *Enterobacter* are nonsymbiotic N fixers and are in the total coliform group. *K. pneumoniae* is important in human and industrial wastes and can cause bacterial infections in humans
Gram negative anaerobic bacteria	*Desulfovibrio*	Reduces sulfate to hydrogen sulfide
Gram-negative, chemolithotrophic bacteria	*Nitrosomonas, Nitrobacter, Thiobacillus*	*Nitrosomonas* catalyze the conversion of NH_4^+ to NO_2^-; *Nitrobacter* oxidize NO_2^- to NO_3^-; *T. ferrooxidans* oxidize ironsulfides producing Fe^{+3} and SO_4^{-2}
Methane producing bacteria	*Methanobacterium*	Anaerobic bacteria of wetland sediments that convert carbonate to methane
Gram-positive cocci	*Streptococcus*	Fecal streptococci include human species (*S. faecalis* and *S. faecium*) and animal species (*S. bovis, S. equinus, S. avium*)
Endospore-forming rods and cocci	*Clostridium, Bacillus*	*C. botulinum* survives in soils and bottom sediments of wetlands and causes avian botulism; some *Clostridium* spp. are nonsymbiotic N fixers; *B. thuringiensis* is an insect pathogen; *B. licheniformis* denitrifies NO_3^- to N_2O
Actinomycetes and related organisms	*Nocardia, Frankia, Streptomyces*	Filamentous bacteria occurring aquatically and in soils; *Nocardia* is implicated in sludge bulking in sewage treatment. *Frankia* is a symbiotic N fixer with alder trees

Table 3.8: Classification of bacteria in CW, from Kadlec and Knight (1996).

3.5.3 Invertebrates and vertebrates

Invertebrates may be motile or attached, they form the initial trophic levels of the higher organisms and part of the grazing food chain, and they feed directly off the aufwuchs and vegetation. A portion of the invertebrates are detritiivores and feed on the CW detritus produced by the macrophytes, others are omnivorous and some are carnivorous (Rogers et al, 1985). The most common invertebrates in CW fall into four main groups: annelid worms, molluscs, crustaceans and insects.

Vertebrates are at the top of the trophic levels, they may be omnivorous or carnivorous (Rogers et al, 1985). They form the 'visible' wildlife of CW and may be attracted to the wetland environment through design or as an ancillary benefit (Kadlec and Knight, 1996 and EPA, 2000). The habitat values of the constructed wetland will determine the type and diversity of the wildlife. Typical vertebrate species in CW include: fish, which feed off crustaceans, insects and plant material; amphibians, which feed off algae and small foods during their larvae phase and insects during their adult phase; reptiles which tend to be insectivorous or piscivorous; birds, which feed off plant seeds, aquatic invertebrates and fish and mammals, which feed off fish and vegetation (Rogers et al, 1985, Kadlec and Knight, 1996 and EPA, 2000).

3.5.4 Vectors and human health concerns

All the positive aspects of CW are accompanied by some negative aspects, which need to be considered by the designer as they can have a social effect (Kadlec and Knight, 1996). The negative aspects may not only affect humans but also plants and wildlife. The designer needs to classify the possible negative effects and take precautionary measures. Systems that are organically overloaded tend to generate foul odours and encourage mosquito development in FWS systems (Reed et al, 1995). Mosquitoes complete part of their life cycle in the water column of FWS systems, the female mosquito uses mammal blood to support egg growth, she then lays the eggs in the water column, the eggs then hatch into larvae that live off the organic matter in the system, the larvae then move into the 'pupa' stage of the cycle and then finally the adult mosquito emerges from the 'pupa' to start the cycle again (Kadlec and Knight, 1996). Mosquitoes are not only a nuisance but also a potential vector for diseases such as malaria. They may be controlled in FWS constructed wetland systems in a number of ways: chemical and biological insecticides and fish may be used (Reed et al, 1995; Kadlec and Knight, 1996 and EPA, 1995). Other means of mosquito control include steep side slopes to reduce shallow water areas that encourage larvae development because the mosquito larva breathes atmospheric oxygen through a respiratory siphon, covering the water surface with duckweed may also help (Reed et al, 1995 and EPA, 2000). As the primary function of the constructed wetland system is to treat polluted water, the influent may also contain pathogens and viruses and the level of human and wildlife access to the area may have to be reduced especially where direct contact with the polluted water is possible. Certain 'safe' areas may be designed for educational and recreational purposes (Kadlec and Knight, 1996 and EPA, 2000).

3.6 Pollutant removal mechanisms of interest

The removal mechanisms that occur in CW are qualitatively known, however, an adequate quantitative assessment of these mechanisms is still lacking (EPA, 2000). Pollutants within CW undergo chemical, physical and biological reactions and transformations, summarised by Brix (1993a) and EPA (2000) in Table 3.9.

Table 3.9: Removal mechanisms in constructed wetlands, Brix (1993a).

Polluted water constituent	Removal mechanism
Suspended solids	Sedimentation/filtration and sorption.
Biodegradable organics	Sedimentation/filtration, sorption, volatilization and microbial degradation.
Nitrogen	Sedimentation/filtration, sorption, volatilization, ion exchange.
	Ammonification followed by microbial nitrification and denitrification.
	Plant uptake.
Pathogens	Sedimentation/filtration.
	Natural die-off.
	UV radiation.
	Excretion of antibiotics from roots of macrophytes.

The relevance of the above potential mechanisms is dependant upon the external input, internal interactions and polluted water characteristics of the constructed wetland (EPA, 2000). The specific mechanisms and their effect on polluted water quality will be explained in more detail below.

3.6.1 Suspended solids

Suspended solids removal is highly effective in both subsurface and free water surface constructed wetlands (Reed et al, 1995), usually taking place in the initial 20 to 40 percent of the bed (Kadlec, 1993). Many pollutants that are in the suspended solid form, such as metals, organic chemicals, and biodegradable organics can be associated with these removal mechanisms in Table 3.9 (equation 3.36) (Reed et al, 1995). Suspended solids enter the system during construction and through the influent polluted water, chemical precipitation, biomass generation and decay, resuspension of settled solids, rainfall and wind driven particulates (Figure 3.20) (Reed et al, 1995 and Kadlec and Knight, 1996). Deposition and accumulation of the mineral portion of the suspended solids within VSB is of design concern due to the potential for 'clogging' of the void spaces, especially near the inlet, reducing the medium's hydraulic conductivity (Reed et al, 1995; Kadlec and Knight, 1996; Wood, 1999 and EPA, 2000). The primary removal mechanisms in CW are flocculation/sedimentation and filtration/interception. In discrete settling the suspended particles settle independently and are not influenced by other particles or changes in size and density; it is directly proportional to the diameter of the particle and the difference in particle and fluid densities, and inversely proportional to the drag on the particle (Kadlec and Knight, 1996 and EPA, 2000).

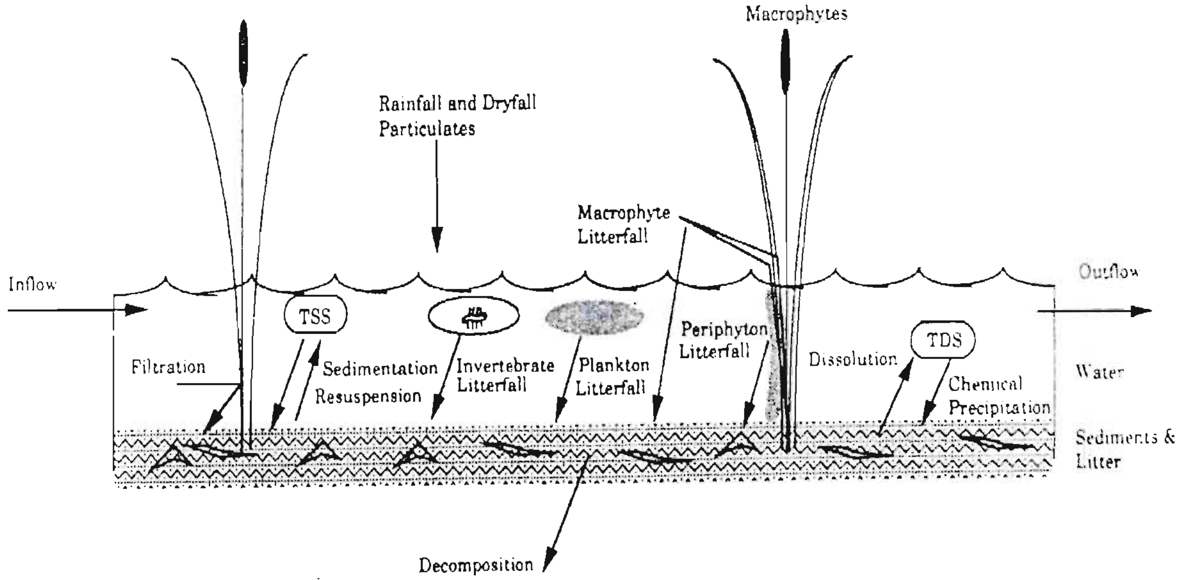


Figure 3.20: Suspended solids storages and transfers in CW, from Kadlec and Knight (1996) (TSS = Total Suspended Solids and TDS = Total Dissolved Solids).

The settling velocity of a spherical particle under turbulent conditions (Re_p > 1) may be mathematically expressed by equation 3.41 (Kadlec and Knight, 1996):

$$V_S = \sqrt{\frac{4}{3}\frac{g \cdot d}{C_D}\left(\frac{\rho_S - \rho}{\rho}\right)} \qquad (3.41)$$

where V_S = settling velocity of the suspended solid particle, m/s

d = suspended solid particle diameter, m

C_D = drag coefficient

ρ = density of water, kg/m^3

ρ_s = density of solids, kg/m^3

The drag coefficient is a function of the particle Reynolds number, expressed by equation 3.42:

$$Re_P = \frac{d \cdot V_S}{\nu} \qquad (3.42)$$

Using the calculated Reynolds number and Figure 3.21, to estimate the drag coefficient, the suspended solids particle settling velocity under turbulent conditions, may be estimated.

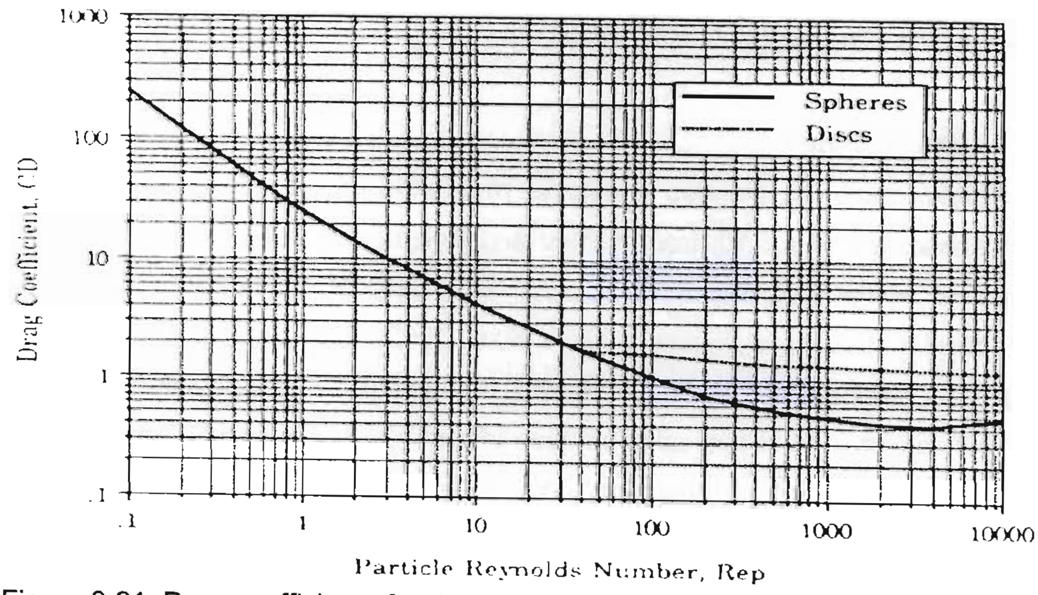


Figure 3.21: Drag coefficients for discs and spheres as a function of the Reynolds number, from Kadlec and Knight (1996).

If the Reynolds number is less than one, and laminar flow conditions apply, the drag coefficient is inversely proportional to the particles Reynolds number and the particles settling velocity may be expressed by Stokes law (Equation 3. 43) (Kadlec and Knight, 1996 and EPA, 2000):

$$V_S = \frac{g \cdot d^2}{18 \cdot \mu}(\rho_S - \rho) \qquad (3.43)$$

where μ = viscosity of water, kg/ms

Flocculent settling, which is dependent upon particle electrical charges, is caused through particulate collisions resulting in particle growth and changes in characteristics over time, that result in increased particle settling velocities (EPA, 2000). Filtration is particularly relevant in VSB as the stems of the vegetation in FWS CW are too far apart to have any significant influence; on the other hand the medium's matrix, including the rhizoshpere, in VSB, will have a more efficient filtering capacity (EPA, 2000). Interception and adhesion of suspended solids to the aufwuchs, however, may be a significant removal mechanism in both VSB and FWS CW. Resuspension of the suspended solids in FWS CW may be induced by increased water velocities, wind and bioturbation, including fish, mammals and birds. Gases generated by algae and other microorganisms in the aufwuchs may also cause flotation of particulates (EPA, 2000).

3.6.2 Organic matter

The organic matter in polluted waters is measured analytically in a number of ways (Summarised in Table 3.10, from Kadlec and Knight, 1996). Organic matter enters the constructed wetland in the influent; it is also generated internally by a number of decomposition processes (Figure 3.22) (Wetzel, 1993; Kadlec and Knight, 1996 and EPA, 2000). The mechanisms for organic matter removal in CW are physical separations and biological conversions. Physical separation of particulate organic matter follows the same mechanisms described for suspended solid removal. Where the settled fraction of the organic matter undergoes biological decomposition, soluble organic matter may also undergo separation processes, such as sorption and volatilisation (EPA, 2000).

Table 3.10: Different analytical techniques used to measure the organic material in polluted Waters, Wetzel (1993).

Analytical measurement	Explanation
BOD_X	Biochemical oxygen demand, is the measure of oxygen consumption of microorganisms in the oxidation of organic matter, over x amount of days.
COD	Chemical oxygen demand, is the amount of a chemical oxidant, required to oxidise the organic matter.
TOC	Total organic carbon is measured by chemical oxidation followed by analysis for carbon dioxide.

Biological conversions are the most important removal mechanism for the organics, physical separation does help to remove portion of the organic matter. However, resuspension of this fraction back into the water column can occur (EPA, 2000). Organisms that consume the organics to sustain life and to reproduce, drive its biological removal, producing organics through cell synthesis (EPA, 2000). The end products of the biological reactions depend on the terminal electron acceptors. If oxygen is the acceptor, the reaction is termed 'aerobic', and the end products are mineralised products, gases and new biomass. If nitrates, sulphates or carbonates are the terminal electron acceptors, the reaction is termed 'anoxic' and the end products are mineralised products, gases but less biomass per unit of substrate converted, as the reaction yields less energy than the aerobic reaction. Anaerobic reactions where organic matter is the electron acceptor and donor are the least efficient. The removal of organic matter will only take place under these conditions if hydrogen or methane is produced (EPA, 2000). The type of reaction that will take place within the constructed wetland depends on oxygen replenishment in the system (as described earlier oxygen is transferred via the rhizomes in VSB and via atmospheric reaeration in FWS wetlands). Both of these mechanisms of oxygenation have to compete with the rate of oxygen consumption due to aerobic biological reactions. If the rate of consumption is higher than replenishment, the water column will be predominantly anaerobic or anoxic (EPA, 2000).

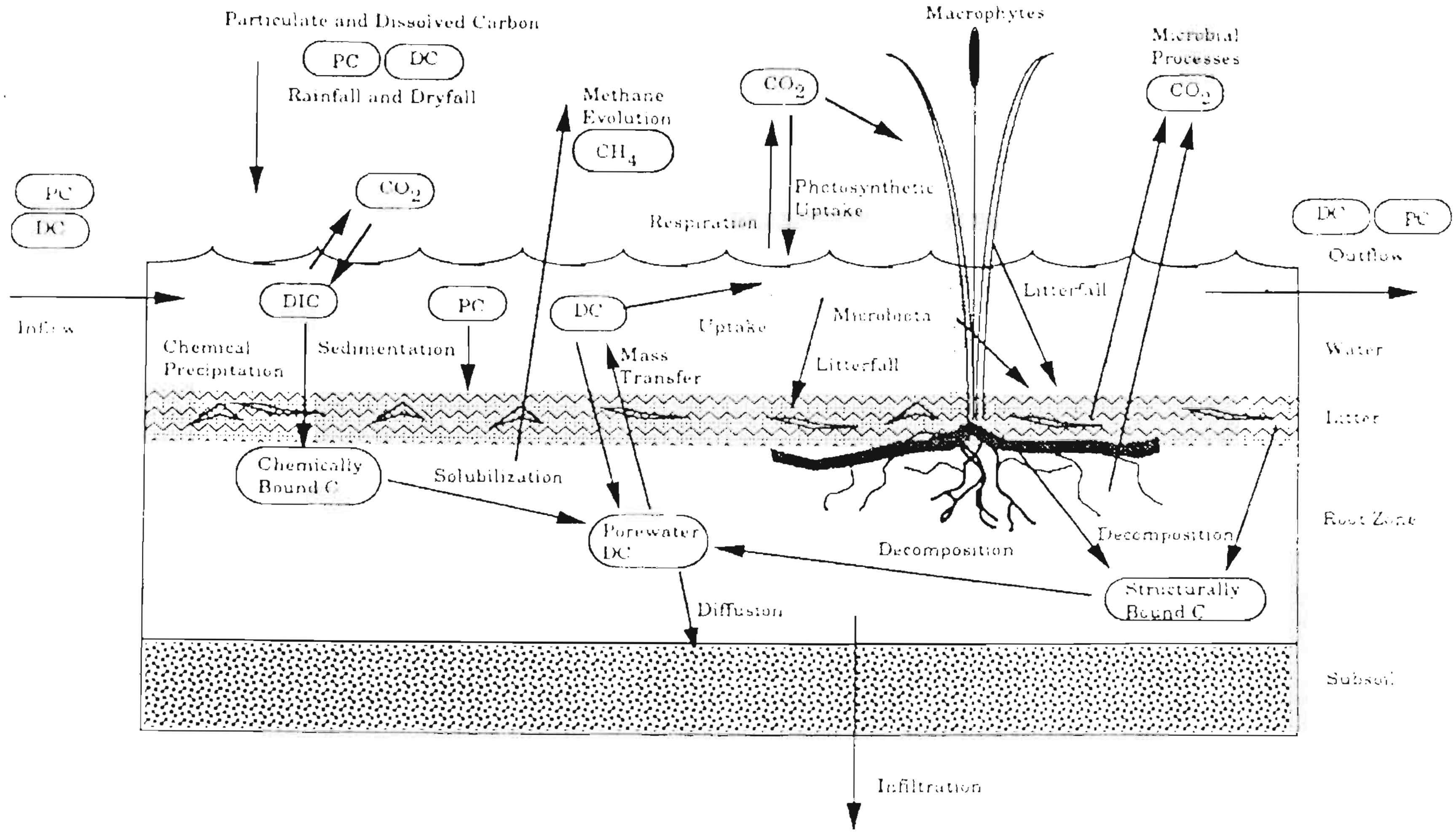


Figure 3.22: Carbon storages and transfers in CW, from Kadlec and Knight (1996). (DC = Dissolved Carbon, PC = Particulate Carbon, DIC = Dissolved Inorganic Carbon, DOC = Dissolved Organic Carbon, CH_4 = Methane and CO_2 = Carbon Dioxide. Biomass carbon consists of living and dead biomass and organic decomposition products).

3.6.3 Nitrogen

The most important forms of inorganic nitrogen in constructed wetlands are ammonia (NH_3^+), ammonium (NH_4^+), nitrite (NO_2^-), nitrate (NO_3^-), nitrous oxide (N_2O) and dissolved elemental nitrogen or dinitrogen gas (N_2). Nitrogen may also be in many organic forms: urea, amino acids, amines, purines and pyrimidines (Kadlec and Knight, 1996). The organic nitrogen which is associated with suspended solids may be removed by the same mechanisms previously discussed for suspended solids through physical separation followed by ammonification of the settled sediment (EPA, 2000). As mentioned, ion exchange of ammonium within the medium's matrix may play a short-term role in nitrogen removal, until the medium's ion exchange capacity has been depleted (EC/EWPCA, 1990; Reed et al, 1995 and EPA, 2000). The quantity of the relative species of aqueous ammonia in the water column is pH and temperature dependant and for a typical wetland system under average environmental conditions of 25^0C and a pH of 7, un-ionised ammonia is only 0.6% of the total ammonia present; at a pH of 9.5 and a temperature of 30^0C, the percentage of un-ionised ammonia increases to 72%. The volatility of un-ionised ammonia results in ammonia losses from the wetlands under high temperature and pH conditions, which may occur in CW during active photosynthesis (Kadlec and Knight, 1996 and EPA, 2000). The biologically mediated transformations of the nitrogen species are the most important mechanisms of nitrogen removal (Figure 3.23) (Reed et al, 1995).

If the influent content of organic nitrogen is high, the first microbial reaction will be ammonification. Ammonification will also take place during the break down of internally generated organic nitrogen. During this reaction organically combined nitrogen is transformed to ammoniacal nitrogen (EPA, 2000), thus adding to the influents ammoniacal nitrogen concentration. Under aerobic conditions the ammoniacal nitrogen is converted, through a two-step process, to nitrite and nitrate nitrogen. Approximately 4.3g of dissolved oxygen and 7.14g of alkalinity, as $CaCO_3$, is required to nitrify 1g of ammoniacal nitrogen. As discussed, due to the oxygen limitations in both VSB and FW CW, nitrification of high concentrations of ammoniacal nitrogen is limited and the use of hybrid systems is more appropriate (EC/EWPCA, 1990; Robinson et al, 1993; Van Oostrom and Russel, 1994; Reed et al, 1995; Kadlec and Knight, 1996; Green et al, 1997; Cooper, 1999; Cooper et al, 1999 and NADB, 2000).

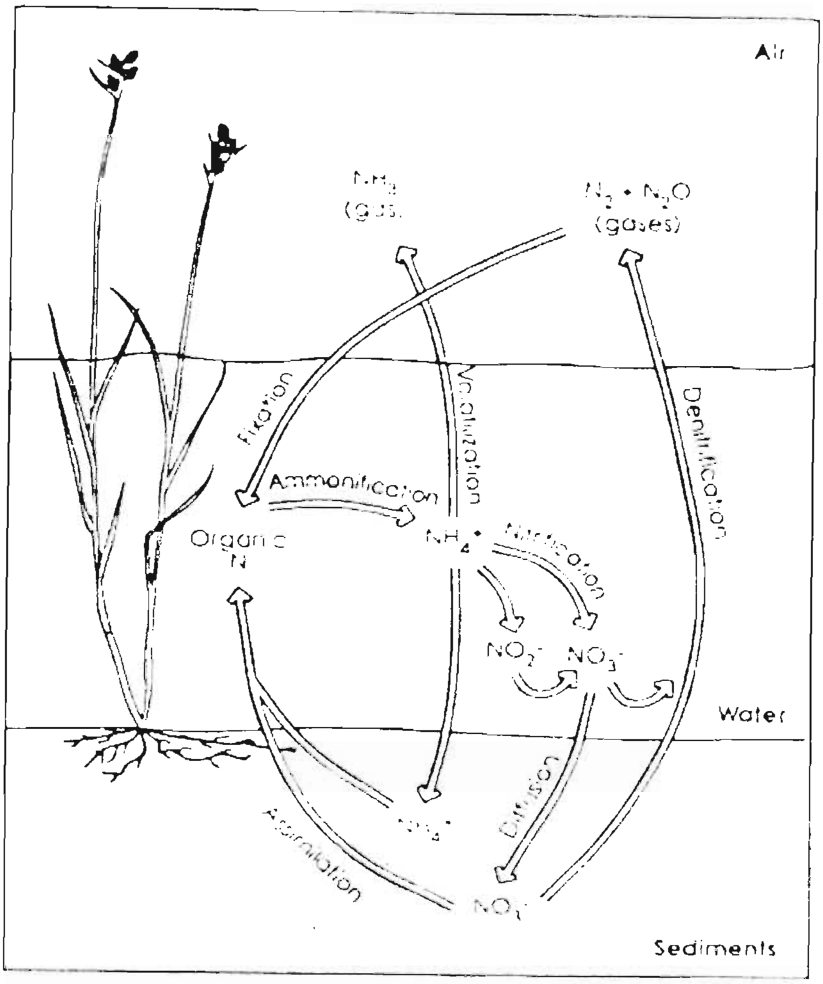


Figure 3.23: Constructed wetland biological nitrogen cycle, from Kadlec and Knight (1996).

Once the ammoniacal nitrogen has been nitrified, either due to pre-treatment or through sequential constructed wetland cells, and it is in the form of nitrate nitrogen, the final biological step in the removal process is denitrification. Denitrification takes place under anoxic conditions where nitrate is the electron acceptor and carbon is the electron donor; nitrate nitrogen is converted to N_2 and N_2O gasses that readily exit the wetland (Reed et al, 1995, Kadlec and Knight, 1996 and EPA, 2000). The nitrogen gas formed through denitrification and present in the water column through atmospheric reaeration may, however, be converted back to organic nitrogen through nitrogen fixation, mediated by specific bacteria and blue-green algae under aerobic and anaerobic conditions (EPA, 2000).

Plants may also have a significant potential to remove nitrogen from wastewaters. The removal of nitrogen by plant uptake must, however, be regarded as temporary removal as most of the nutrients taken up are returned back to the system once the plants die and decompose (Wetzel, 1993). Algae may also remove considerable amounts of nitrogen, but release all nutrients after death, since algae contain less structural refractory material (Rogers et al, 1985).

3.6.4 Pathogens

Pathogens associated with suspended solids in the influent may be separated from the water column by the same mechanisms discussed for suspended solids removal. Once separated they must compete with a consortium of other microorganisms and, being intestinal organisms, most will not survive. They will also be destroyed by predation or by ultra violet irradiation in open water areas. Temperature fluctuations will also not be favourable for pathogen survival (Khatiwada and Polprasert, 1999a and EPA, 2000). Macrophyte rhizomes have also been reported to have an antibiotic effect on pathogens (Brix, 1993a).

3.6.5 Other pollutants

CW have been shown to remove other pollutants such as metals, phosphorus and trace organics. These pollutants, however, are not of interest in this research and the reader is referred to the references for more information (Reed et al, 1995; IAWQ, 1994, 1995, 1997, 1999; Kadlec and Knight, 1996; Robinson et al, 1998a; Wood, 1999 and EPA, 2000).

3.7 Other water quality parameters of interest

3.7.1 Chlorides

Chloride concentrations may be used as a tracer in CW to estimate dilution and concentration effects (Kadlec and Knight, 1996). It may also be used to determine the amount of polluted water infiltration into the substrata and groundwater beneath the constructed wetland (Kadlec and Knight, 1996)

3.7.2 Electrical conductivity

Electrical conductivity of an aqueous solution is the reciprocal of the resistance between two platinum electrodes and is a function of the total quantity of ionised materials in a polluted water sample (Kadlec and Knight, 1996). It is proportional to the total dissolved solids or salinity, the use of electrical conductivity as an indicator for dilution and concentration effects in CW is inaccurate as ionic salts in CW are altered by biological and physical processes (Kadlec and Knight, 1996).

3.7.3 pH

The hydrogen ion concentration in CW is used as an environmental indicator, as chemical and biological reactions are often pH dependant; Denitrifiers require a pH range between 6.5 and 7.5 and nitrifiers a pH value of 7.2 and higher (Kadlec and Knight, 1996). The abundance of relative aqueous ammoniacal nitrogen species is also dependant upon pH, as discussed above, and certain metal precipitates are also pH dependant (Kadlec and Knight, 1996). Diurnal fluctuations in pH experienced in constructed wetlands (Figure 3.24) is influenced by macrophyte and algae photosynthesis, with a decrease in pH values experienced during the night when oxygen is consumed through respiration and carbon dioxide is released into the water column. During the day time the carbon dioxide is consumed and oxygen is released causing an increase in pH values (Kadlec and Knight, 1996 and EPA, 2000). Many CW have been shown to display a 'buffering' capacity to influent pH changes. However, long-term feeds of acidic influent have been reported to be followed by similar long-term trends in effluent pH (Kadlec and Knight, 1996). The 'buffering' capacity, or the capacity of the system to neutralise influent acidic waters is usually measured in terms of alkalinity as $CaCO_3$ (Tchobanoglous and Burton, 1991).

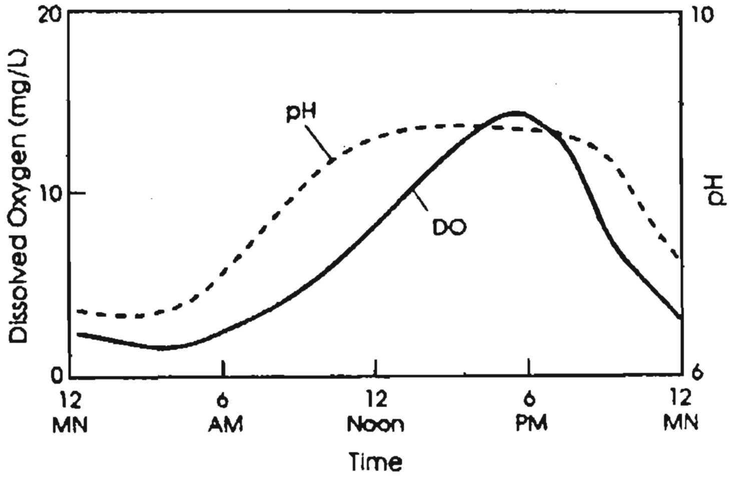


Figure 3.24: Typical diurnal pH fluctuations experienced in CW due to photosynthesis and respiration of macrophytes and algae, from Kadlec and Knight (1996).

3.7.4 Temperature

CW have exhibited a 'buffering' capacity with respect to influent temperatures, typically the effluent water temperature is around the mean daily air temperature, which represents a balance between influent water temperatures, energy gains and losses through solar radiation and evaporation respectively. The detritus layer has also been shown to provide an insulating layer during winter months, preventing the water column from freezing (Kadlec and Knight, 1996).

3.8 Overall treatment performance and expectations

Meteorological processes, such as evapotranspiration, temperature, precipitation and solar radiation, drive the performance of the constructed wetland. These processes are cyclic, both diurnally and annually, influencing the constructed wetland's overall performance to follow the same cyclic trends (Kadlec, 1999). During winter, air temperature and solar radiation are lower, affecting the temperature dependant microbial reactions within the water column; photosynthesis is also affected during this period, with repercussions on vegetative processes and on the overall cycling of carbon and nutrients. During these winter periods the overall performance is likely to be lower than during the warmer summer months, where microbial activity, carbon and nutrient cycling are more active (Kadlec, 1999). Macrophytes also follow a life-death cycle in CW and the build up of dead plant litter during winter undergoes decomposition during summer, releasing carbon and nutrients back into the water column (Wetzel, 1993 and Kadlec, 1999). The effects of evapotranspiration and precipitation also follow these cycles and can, in many ways, have a negative influence on the system's performance (Kadlec, 1999). The overall design of the constructed wetland system must take these cyclic processes into consideration, in order to consistently meet specified discharge standards throughout the design life of the system.

CHAPTER 4

4.1 Full and pilot scale treatment case studies

During the dissertation, site visits to municipal wastewater and landfill leachate treatment plants in South Africa and Great Britain respectively were arranged. EnvirosAspinwall (UK), an environmental management consulting company, arranged a 'Leachate Safari' which included site visits to landfill leachate treatment plants around the UK, including: Fiskerton Landfill (SBR); Whitehead Landfill (SBR); Trecatti Landfill (SBR); Buckden South Landfill (SBR, CW and Ozonation); Sundon Landfill (SBR and CW); Monument Hill Landfill (CW) and Judkins Landfill Site (CW) (Figure 4.1).

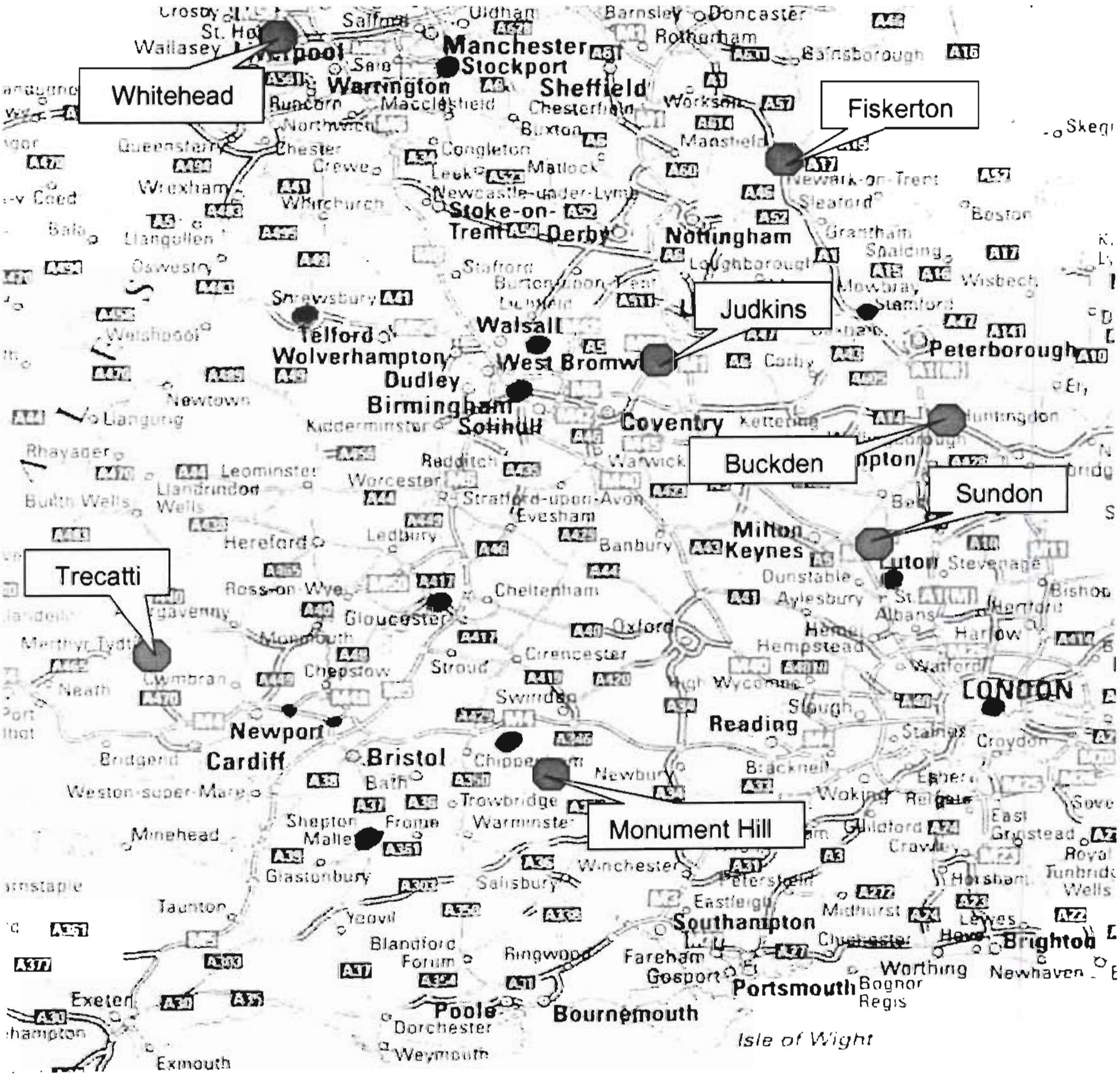


Figure 4.1: Location of the full scale, landfill leachate treatment, case studies visited in England.

Wood (1999) had reviewed the status of CW in South Africa, identifying 58 plants for the treatment of domestic, mining and industrial wastewaters, applied to stormwater and urban catchment management, riverine rehabilitation and protection, groundwater recharge and development of urban nature reserves. It was found during the initial 'desktop' study of this research that many of the South African sites had been decommissioned or were in the process of decommissioning. Among these sites were Mpophomeni in KwaZulu Natal and the CSIR campus wetlands. Further investigation into other sites showed that some of the responsible authorities had little or no knowledge of the existence of the CW. The four sites selected for the case studies range from the above mentioned cases to the largest site in South Africa and include previously commissioned sites that were not identified by Wood (1999). All the sites are involved in the treatment of municipal wastewater and they include: Kranskop; Bethlehem; Kwazamokuhle and Dullstroom (Figure 4.2).

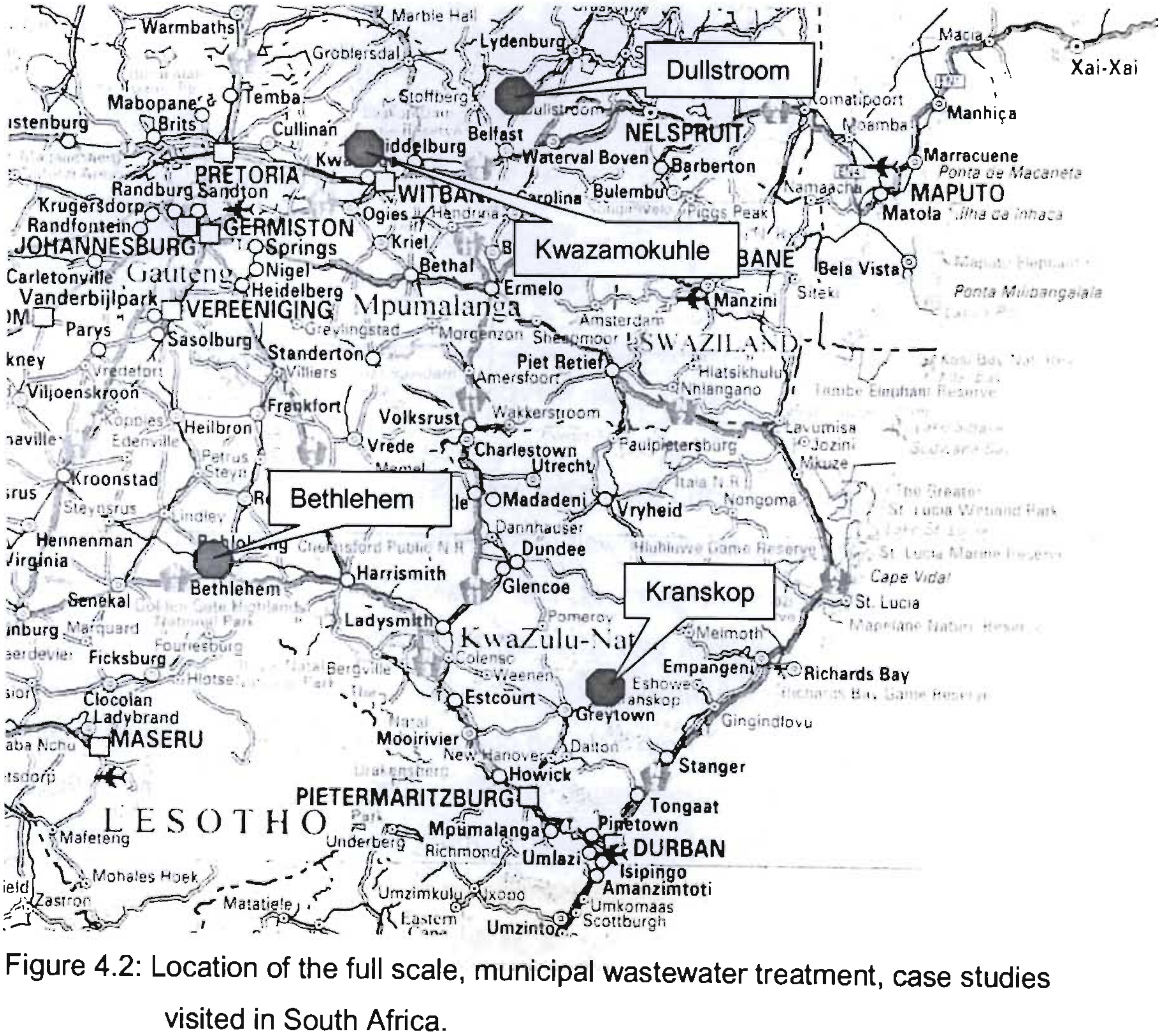


Figure 4.2: Location of the full scale, municipal wastewater treatment, case studies visited in South Africa.

Strachan (1999) gave reference to the La Mercy Landfill Site constructed wetland for the treatment of its leachate. However, during a brief site visit it was noticed that the constructed wetland is no longer in use and no analytical data are avaliable. Included in the following case studies are the pilot scale nitrification/denitrification SBR treatment trials for the treatment of landfill leachate in South Africa, conducted at the Bisasar Road Landfill site in Durban, KwaZulu Natal, South Africa (Strachan, 1999 and Olufsen, 1999) (Figure 4.3).

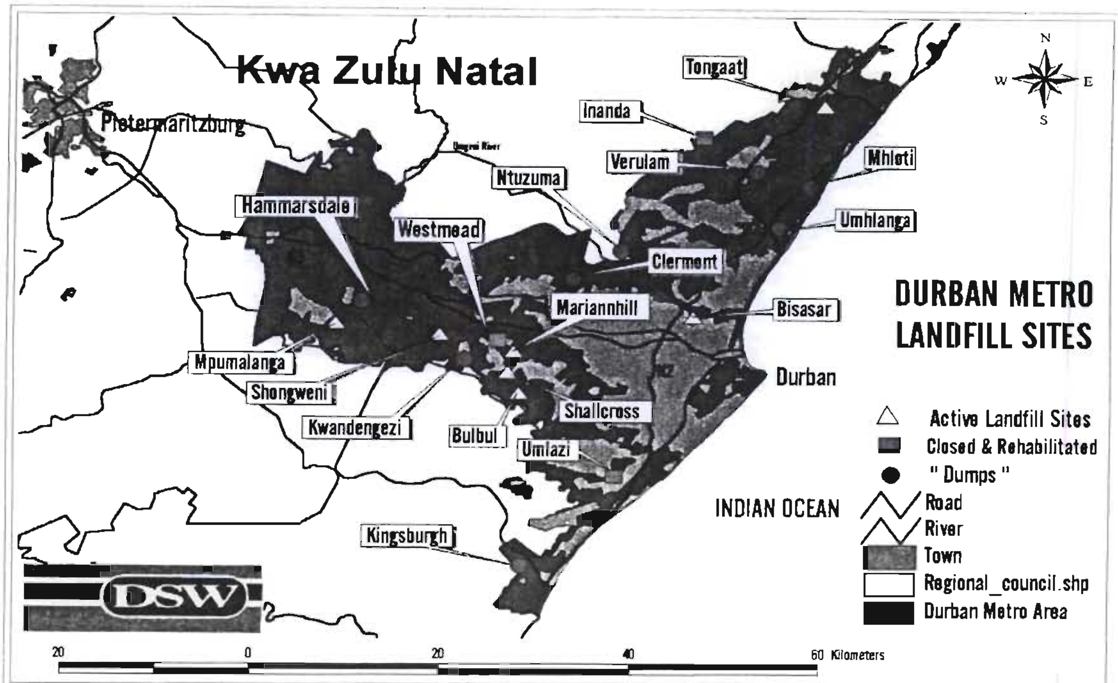


Figure 4.3: Landfill Sites in the Durban Metropolitan Area, courtesy of DSW.

4.1.1 Bisasar Road Landfill Site, pilot scale SBR, treatment plants

In collaboration with EnvirosAspinwall (UK) Durban Solid Waste established a pilot scale nitrification SBR treatment plant at the Bisasar Road Landfill site in October 1998 (Strachan, 1999). The plant was later modified, after the completion of the nitrification treatability trials, for denitrification (Olufsen, 1999). Two leachates were tested in the treatability trials: a leachate from a young operational landfill, the Mariannhill Landfill (Plate 4.1), which showed the characteristics of a leachate from both the acetogenic phase and the methanogenic phase of biodegradation, and a leachate from an old landfill, the Bisasar Road Landfill (Plate 4.2), in the methanogenic phase of biodegradation (Olufsen, 1999). Both landfills are operated by Durban Solid Waste of the Durban Metropolitan Council. The Mariannhill Landfill is located approximately 6 km from central Pinetown and it serves the Inner West City Council (Figure 4.3). The site was originally established in July 1997, it is regarded as a 'new generation' landfill with geomembrane liners and leachate drainage/collection blankets. Leachate is channeled to a collection chamber at the toe of the site, from where it is pumped to two collection tanks. The characteristics of the leachate used in the treatability trials are presented in Table 4.1. The landfill is classified as a GLB$^+$ landfill, implying that it is a large general waste landfill with significant leachate generation (DWAF, 1998).

Plate 4.1: Mariannhill Landfill Site

Plate 4.2: Bisasar Road Landfill Site

The Bisasar Road Landfill is a 210 million cubic metre capacity landfill situated some 10 km from the central business district of Durban (Figure 4.3). The site was established in May 1980, it comprises an existing unlined attenuation waste body over and around which a newly engineered landfill is being developed. It is also classified as a GLB$^+$ landfill. Leachate generated is collected in three ways: leachate generated in the 'core' attenuation section is partially extracted during methane gas extraction, through gas wells that extend to the full depth of the waste (±40m)(Leachate used in treatability trials, characteristics presented in Table 4.1); leachate which is not extracted in the wells is partially intercepted by a sub-soil drain at the base of the stability berm. Leachate generated in the new lined 'cells' of the landfill are collected by collection blankets under the waste body. The current treatment practice is to channel the leachate from the site into a nearby sewerline.

The nitrification/denitrification treatability trials were conducted in a pilot scale SBR system in order to assess the removal efficiency for ammoniacal nitrogen, nitrate nitrogen and COD. The pilot plant consisted of 2 modified 240-litre 'wheelie-bins' (Plate 4.3) operated in a complete nitrification/denitrification cycle over 24 hours, with a mixed liquor volume of 200 litres per SBR.

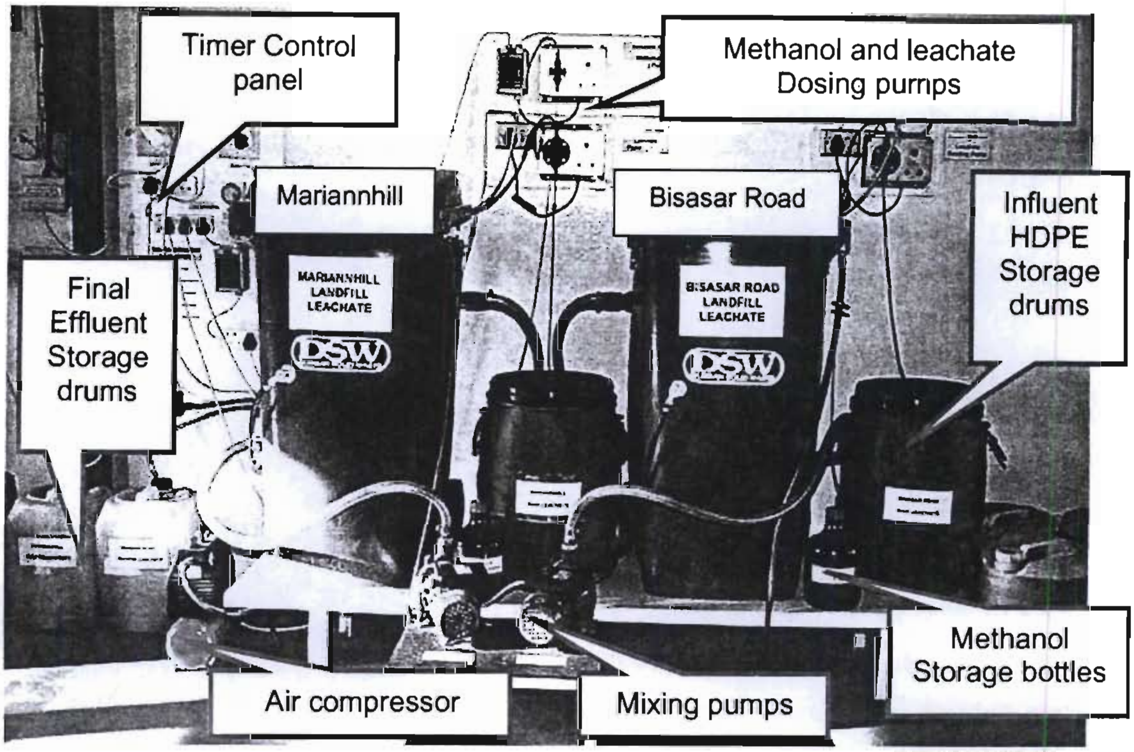


Plate 4.3: Sequence Batch Reactors used in the treatment trials, taken at Bisasar Road laboratory.

Raw leachate is dosed for 1-minute every 6 minutes for 12.5-hours during a 13.5-hour aeration period. At the end of nitrification a 30-minute stand period is allowed to assist in the reduction of dissolved oxygen concentrations to adequate levels for denitrification to take place. A subsequent 4-hour denitrification stage is applied using methanol as a carbon source. A 2-hour aeration phase is then necessary to oxidise any residual methanol and to facilitate the release of nitrogen gas. A final 3-hour settling stage, required to achieve a clear supernatant, is then followed by the effluent discharge.

The successful completion of the trials, proved that the full removal of nitrogen compounds can be easily achieved, under South African climatic conditions, in a single sludge SBR system, both for the young leachate from Mariannhill and the very old from Bisasar Road landfill sites (Robinson and Strachan, 1999; Strachan, 1999; Olufsen, 1999; Strachan et al, 2000a and Strachan et al, 2000b).

Table 4.1: Typical concentration of pollutants in the raw Bisasar Road and Mariannhill landfill leachates.

Parameter	Bisasar Road	Mariannhill
pH	8.1	8.1
Alkalinity (mg$CaCO_3$/l)	5724	4186
NH_4-N (mg/l)	1126	470
COD (mg/l)	2420	1166
BOD_5/COD	0.31	0.57
Chlorides (mg/l)	3110	1484
Sulphates (mg/l)	<1	30
Cadmium	<0.1	<0.1
Lead	0.2	0.3
Mercury	<0.005	<0.005
Sodium	1930	860
Magnesium	146	318
Potassium	1300	794
Calcium	64	331
Chromium	0.1	0.1
Manganese	0.3	3.4
Iron	4.9	33
Nickel	0.5	0.4
Copper	<0.1	<0.1
Zinc	0.1	0.3

The SBR proved to be simple to operate and required very low maintenance. The findings of the initial nitrification treatability trials (15 weeks) showed that the peak biological loadings of ammoniacal nitrogen for the Bisasar Road SBR was 0.032 kgNH_4-N/kgTSS/d and 0.019 kgNH_4-N/kgTSS/d for the Mariannhill SBR, without causing stress on the system (Strachan, 1999). During the combined nitrification/denitrification treatability trials (24 weeks), biological loading rates of 0.025 and 0.015 kgNH_4-N/kgTSS/d were experienced for the Bisasar Road and Mariannhill SBR respectively (Olufsen, 1999).

Ammoniacal nitrogen removal efficiencies of greater than 99 percent were consistently achieved in both trials (Strachan, 1999 and Olufsen, 1999). During the nitrification trials COD removal efficiencies of 60 and 70 percent for the Bisasar Road and Mariannhill SBR respectively were achieved with an average COD removal efficiency of 50 percent during the nitrification/denitrification trials (Strachan, 1999 and Olufsen, 1999). The alkalinity requirements during the nitrification trials compared with the theoretical value of 7.14mg of alkalinity (as $CaCO_3$) per 1mg of ammoniacal nitrogen nitrified, however, were found to be slightly higher (Strachan, 1999), during the nitrification/denitrification trials the average net loss of alkalinity was found to be 3.6 mg of alkalinity (as $CaCO_3$), compared to the theoretical value of 3.7 mg of alkalinity (as $CaCO_3$) (Olufsen, 1999). Methanol was used as the external carbon source for the denitrification trials and a ratio of 4 to 5 kgCH_3OH : 1kg nitrate nitrogen denitrified was determined in order to achieve and maintain treatment efficiencies of 99 percent (Olufsen, 1999). Operational problems experienced included, aerator and mixing pump problems and foaming due to possible biological stressing of the system (Strachan, 1999 and Olufsen, 1999). Further research into the use of molasses as the external carbon source for denitrification showed that high denitrification removal efficiencies could be achieved at ratios of 8.5kg of molasses : 1kg nitrate nitrogen denitrified; however, COD removal efficiencies dropped, ranging from 15 to 40 percent, probably due to the complex composition of the molasses and the effluent colour was of very low quality (Strachan et al, 2000).

4.1.2 Whitehead Landfill Site and full scale SBR treatment plant

(EnvirosAspinwall, 2001)

The Whitehead Landfill nitrification SBR treatment plant, situated some 10km west of Salford, was designed to pretreat the landfill's leachate before discharge to sewerline, it has a design flow of 150 m^3/d of raw leachate with the possibility of treating up to 200 m^3/d, provided that the strength is significantly reduced by this time. The effluent was required to meet the conditions of the Trade Effluents Discharge consent, outlined in Table 4.2, with a maximum trade effluent discharge of 400m^3/d and a rate that shall not exceed 6litres/sec. The treatment system (figure 4.4) comprises a raw leachate balancing lagoon from which leachate is dosed into the SBR (Figure 4.5 and Plate 4.4), the SBR is automated with online pH, temperature and dissolved oxygen sensors, aeration is achieved using 4 air-entraining 18.5Kw venturi aerators, positioned for optimum mixing and aeration.

Table 4.2: Trade effluent consent values for Whitehead landfill SBR effluent discharge.

Parameter	Consent value
Sulphides, hydrosulphides, polysulphides and substances producing hydrogen sulphide on acidification	1mg/l
Separable grease and oil	100mg/l
Sulphates as SO_4	1000mg/l
Toxic metals	5mg/l
Cyanides and cyanogen compounds which produce cyanide on acidification	1mg/l
COD	200kg per any 24 hours
Methane in solution	0.14mg/l
Ammoniacal nitrogen	35mg/l
pH	6 to 11
Temperature	Not greater than 43.3°C

An alkalinity dosing pump connected to the online pH sensor doses required alkalinity in the form of sodium hydroxide to maintain the pH between 7.8 and 8, phosphoric acid is added manually every month. A silicone antifoam emulsion, dosed only during the aeration period if required, is used to control excessive foaming. The effluent from the SBR is pumped to the effluent-balancing lagoon, from where it is pumped to the public sewer, in accordance with the flow rates specified by the Trade Effluents Discharge consent.

Plate 4.4: Whitehead Landfill SBR.

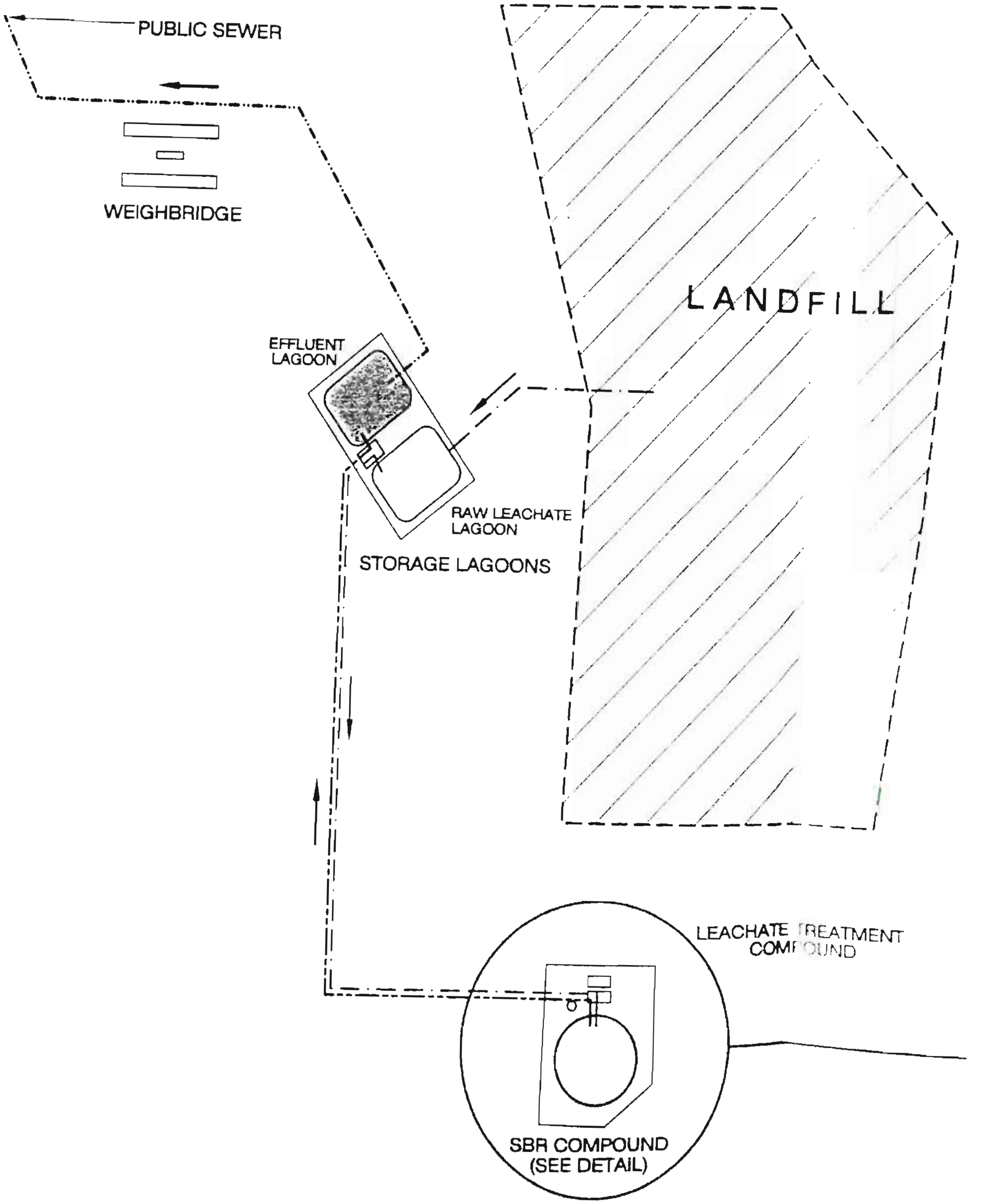


Figure 4.4: Schematic layout of the Whitehead Landfill treatment system, courtesy of EnvirosAspinwall (UK).

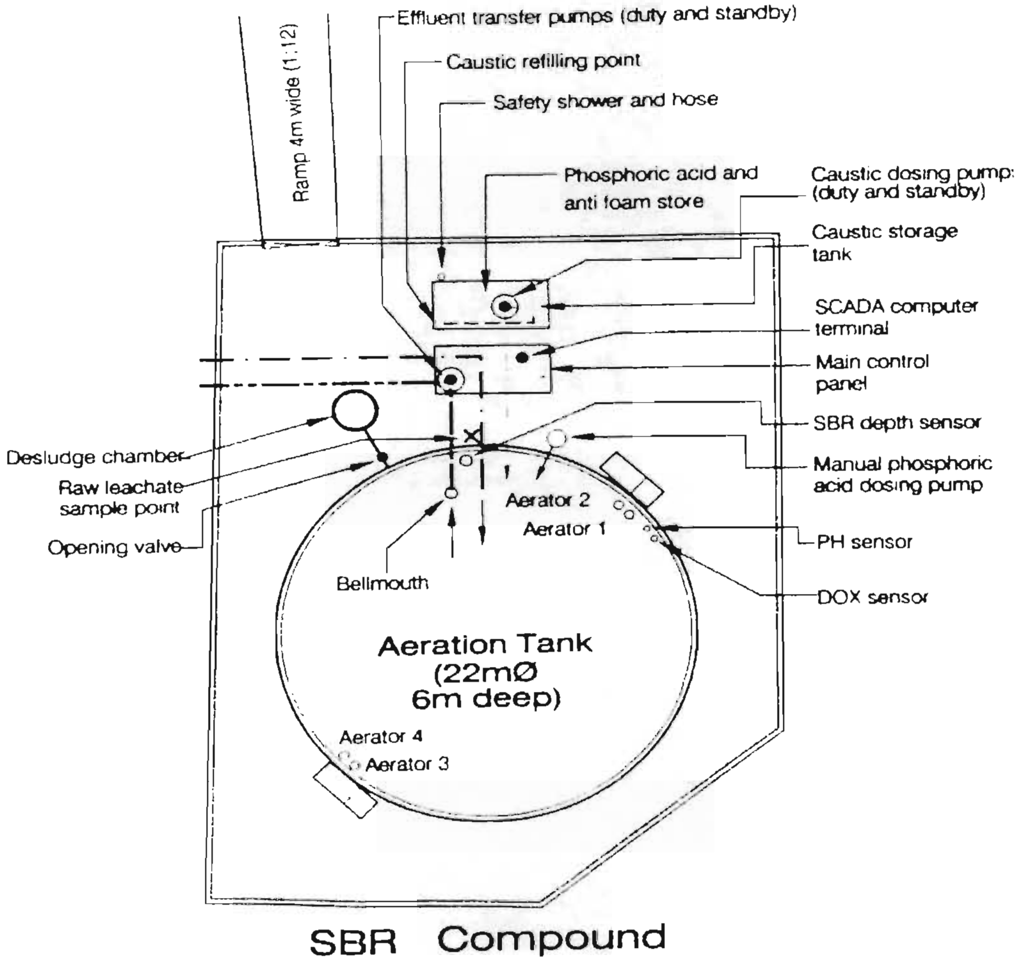


Figure 4.5: Schematic diagram of the Whitehead Landfill SBR, courtesy of EnvirosAspinwall (UK).

4.1.3 Trecatti Landfill Site and full scale SBR treatment plant

(Robinson and Harris, 2001)

Trecatti landfill, a 10Mm^3 lined landfill (Plate 4.5), in the United Kingdom, is located in a disused opencast coal working near to Merthyr Tydfil. The treatment plant is in an extremely exposed location subject to extreme weather conditions. It was designed to pretreat the methanogenic leachate before final discharge into the Merthyr Tydfil sewerage system, it comprises two covered nitrification SBR units, achieving aeration and maintaining adequate temperatures for nitrification to take place by using venturi aerators. The treatment system is fully automated, with dissolved oxygen, pH and temperature sensors, it has consistently maintained extremely high ammoniacal nitrogen removal efficiencies (Table 4.3), despite the climatic conditions.

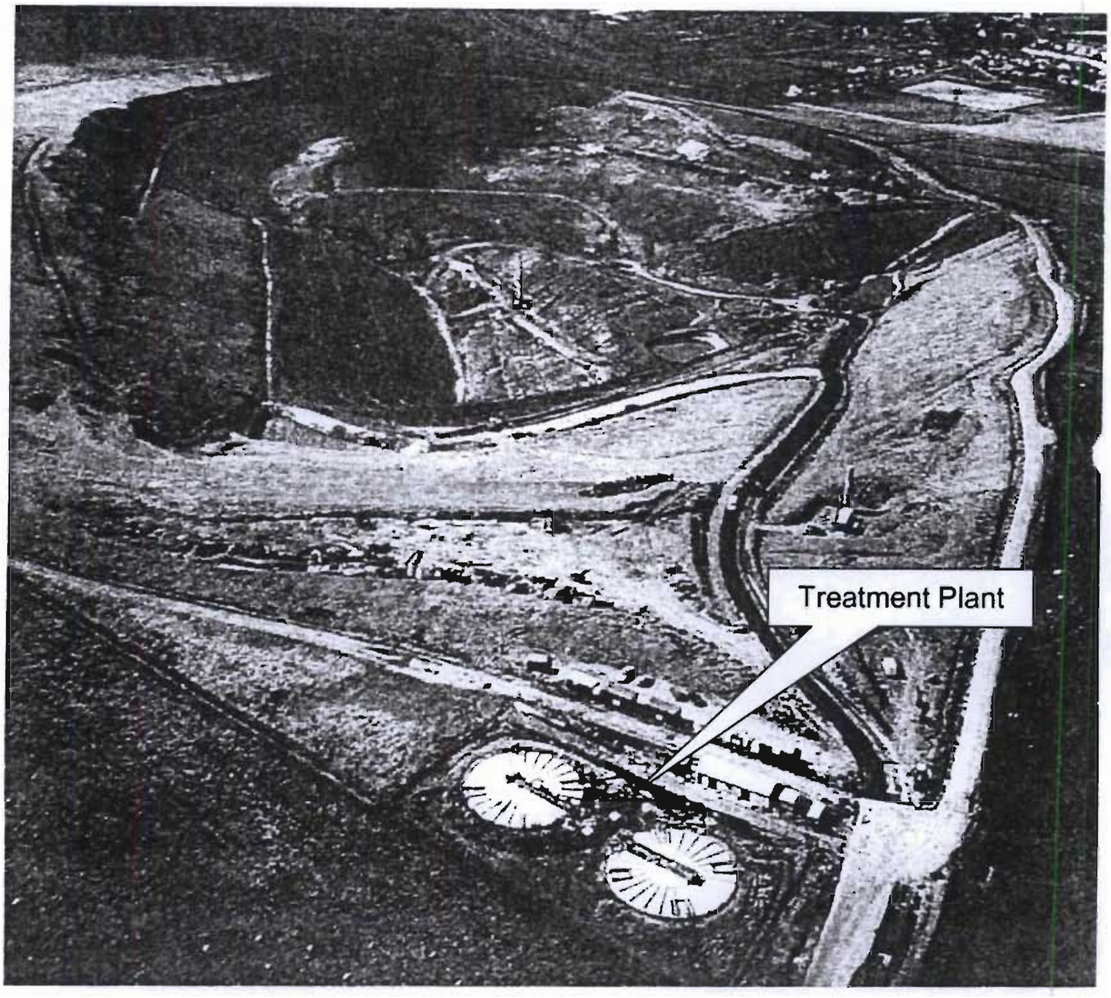


Plate 4.5: Trecatti Landfill site and treatment plant, courtesy of EnvirosAspinwall (UK).

Table 4.3: Typical Leachate and effluent quality for Trecatti Landfill, courtesy of EnvirosAspinwall (UK).

Parameter	Influent leachate	SBR effluent
pH	7.1	7.6
COD (mg/l)	1200	299
BOD_5 (mg/l)	210	3
TOC (mg/l)	299	115
NH_4-N (mg/l)	541	0.5
Chloride (mg/l)	1070	991
Alkalinity (as $CaCO_3$) (mg/l)	4540	608
Conductivity (mS/m)	10.34	8.31
NO_3-N (mg/l)	<0.1	616

NO_2-N (mg/l)	0.6	<0.1
Sodium (mg/l)	750	608
Magnesium (mg/l)	206	193
Potassium (mg/l)	469	379
Calcium (mg/l)	249	262
Iron (mg/l)	10.43	<0.6
Zinc (mg/l)	0.53	0.12

4.1.4 Fiskerton Landfill Site and full scale SBR treatment plant

(Robinson and Harris, 2001)

The Fiskerton Landfill Site, in the United Kingdom, is located some 4 km south west of Newark, in Nottinghamshire. It is a relatively small landfill that receives domestic and commercial wastes. The landfill is surrounded by a cutoff wall that is 'grooved' into the underlying clay substrata, designed to contain the generated leachate. A discontinuity in the cutoff wall coupled with the build up of leachate within the landfill causing a hydraulic gradient between the leachate table and the surrounding water table has led to the migration of the leachate into the surrounding environment. In order to reduce the leachate levels within the landfill and hence possibly reverse the hydraulic gradient from out-of to into the landfill, a tank-based nitrification SBR system (Plate 4.6) with a surface aerator (Plate 4.7) and a design flow of 30 m^3/day of leachate was constructed. The plant is automated with pumps that balance the loadings. The final effluent from the treatment plant is then discharged into a local sewerage treatment works through a 1km long pipeline.


Plate 4.6: Tank based SBR.


Plate 4.7: Surface aerator.

The typical influent and effluent data for the treatment plant is presented in Table 4.4.

Table 4.4: Typical influent and effluent data from the Fiskerton nitrification SBR treatment plant, courtesy of EnvirosAspinwall (UK).

Parameter	Influent leachate	Treated effluent
pH	7.3	8
COD (mg/l)	1640	378
BOD_5 (mg/l)	660	16
TOC (mg/l)	538	85
NH_4-N (mg/l)	414	0.08
Chloride (mg/l)	897	912
NO_3-N (mg/l)	<0.5	311
NO_2-N (mg/l)	0.2	0.06
Conductivity (mS/m)	9	5.8
Alkalinity (as $CaCO_3$)	4430	720
Sodium (mg/l)	620	582
Magnesium (mg/l)	236	141
Potassium (mg/l)	571	528
Calcium (mg/l)	180	153
Iron (mg/l)	19.6	3.57
Zinc (mg/l)	0.46	0.251

4.1.5 Buckden South Landfill Site and full scale landfill leachate treatment plant

(Robinson and Harris, 2001)

The Buckden South Landfill Site (Plate 4.8), in the United Kingdom, is located about 1km south west of Huntingdon in Cambridgeshire; it is a closed landfill adjacent to the River Great Ouse that flows within 500m of the site.

As part of the restoration works, a leachate treatment plant was designed and constructed in the flood plain of the river. The plant has a design flow of up to 200 m^3/d and comprises two nitrification SBRs in series with an ozonation plant and 2500 m^2 of constructed wetlands, capable of treating the leachate to discharge consent values, allowing the rainbow trout to live in the final effluent that is discharged into the adjacent river.

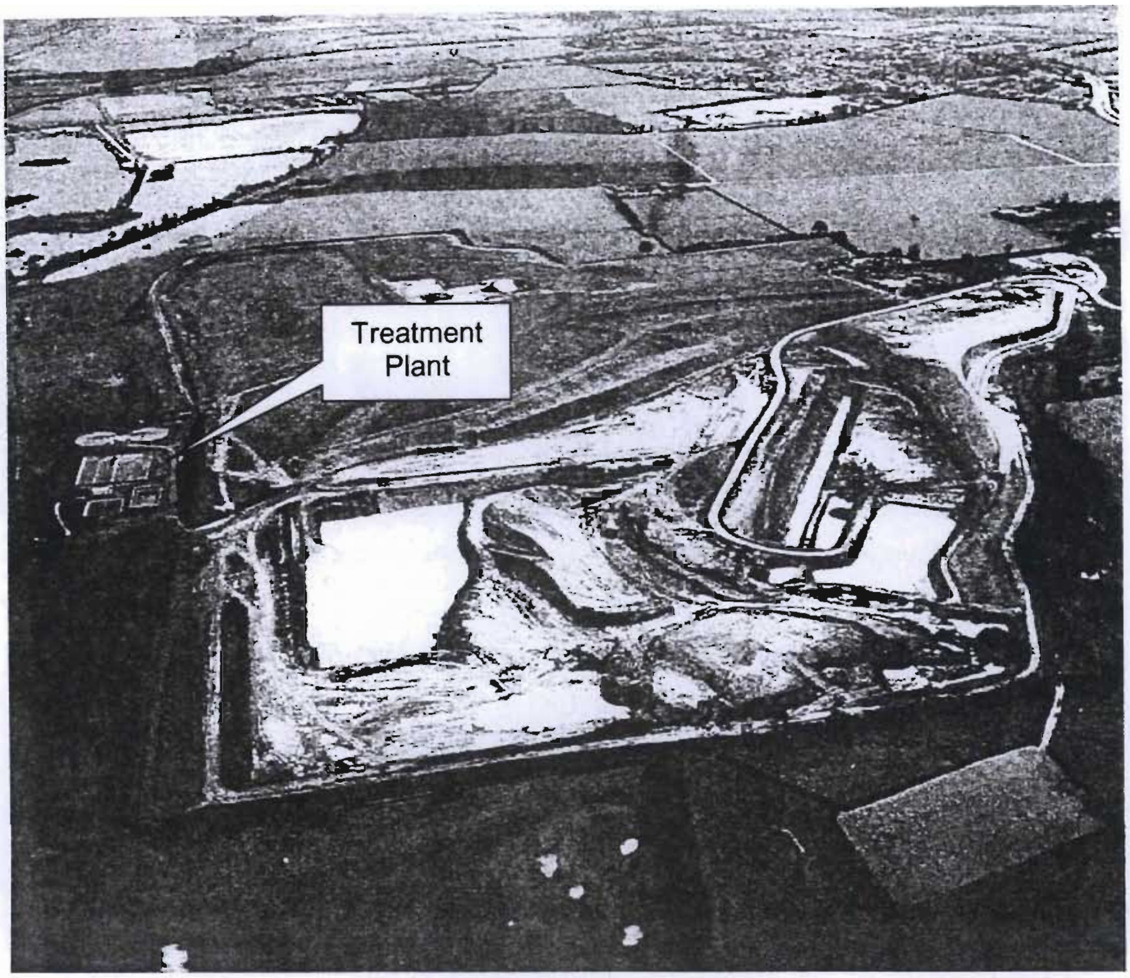


Plate 4.8: Arial view of Buckden South Landfill Site, courtesy of EnvirosAspinwall (UK).

A full characterisation of the leachate indicated the presence of the herbicides Mecoprop and Isoproturon at unacceptable levels. The leachate was also found to be indicative of the methanogenic phase of biodegradation with low BOD_5 to COD ratios. A nitrification SBR system was specifically designed for the removal of ammoniacal nitrogen, from typical concentrations of 300 to 400 mg/l to below a consent value of 10 mg/l. Denitrification was not required. Each SBR (Plate 4.9) was covered for protection from the harsh climate. Aeration was achieved using venturi aerators (Plate 4.10) that were also used to maintain the heat efficiency of the reactors. Typical influent and effluent values are presented in Table 4.5.

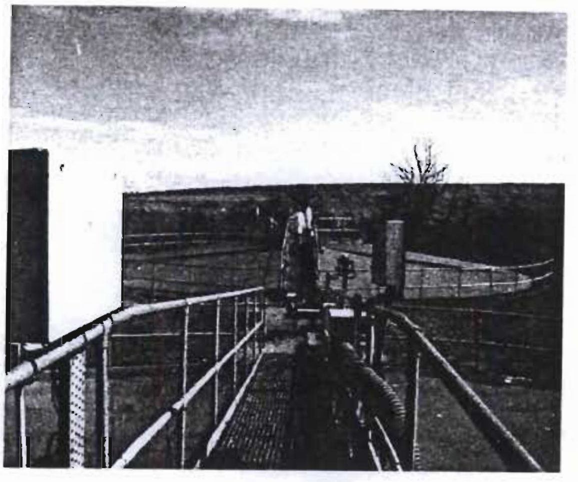


Plate 4.9: Covered twin-SBR system.

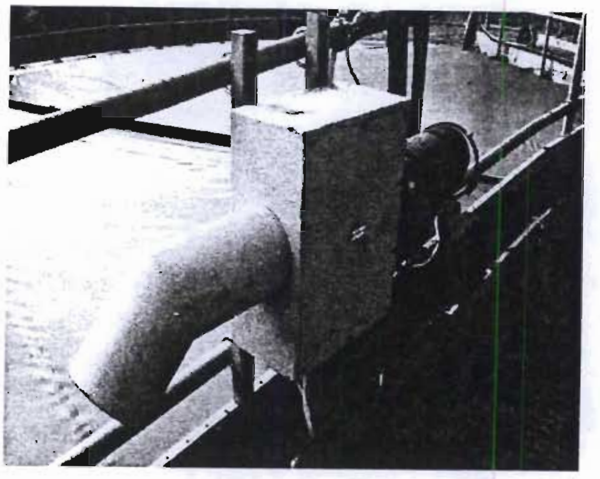


Plate 4.10: Venturi aerator.

Table 4.5: Typical influent and effluent values for the twin-SBR system, courtesy EnvirosAspinwall (UK).

Parameter	Influent Leachate	Effluent SBR1	Effluent SBR2
pH	7.2	8.4	7.9
SS (mg/l)	96	26	14
COD (mg/l)	600	424	399
BOD_5 (mg/l)	35	4	6
NH_4-N (mg/l)	405	1.6	<0.1
Chloride (mg/l)	1830	1700	1720
NO_3-N (mg/l)	0.5	396	409
NO_2-N (mg/l)	<0.05	0.1	0.07
Iron (mg/l)	17.4	<0.6	2.9
Zinc (mg/l)	<0.05	0.16	0.21

As previously mentioned the stringent discharge consent required that rainbow trout should be able to survive in the final effluent. A characterisation of the landfill's leachate also indicated the presence of the herbicides mecoprop and isoproturon in unacceptable levels, which would not be substantially removed by the twin SBR system. It was decided to follow the twin SBR system with 2000m^2 of VSB CW, planted with *Phragmites*, in order to polish the effluent and possibly aid in the removal of some of the more biodegradable mecoprop. The effluent from these reed beds is then passed through an ozonation plant, that is capable of dosing up to 150mg/l of ozone, in order to break down the residual herbicides into more degradable organics, which are then polished through a final 500m^2 of VSB CW, vegetated with *Phragmites* before being discharged into the River Ouse (Plate 4.11). The CW are lined with a synthetic liner. The inlet structure comprises a channel laid across the bed width with a central feeder. The outlet structure is a perforated pipe situated on the outlet end along the bottom of the bed. Water level controls are provided as well as a rodding hole in the outlet pipe for maintenance and cleaning. The typical treatment results from the CW and ozonation plant are presented in Table 4.6.

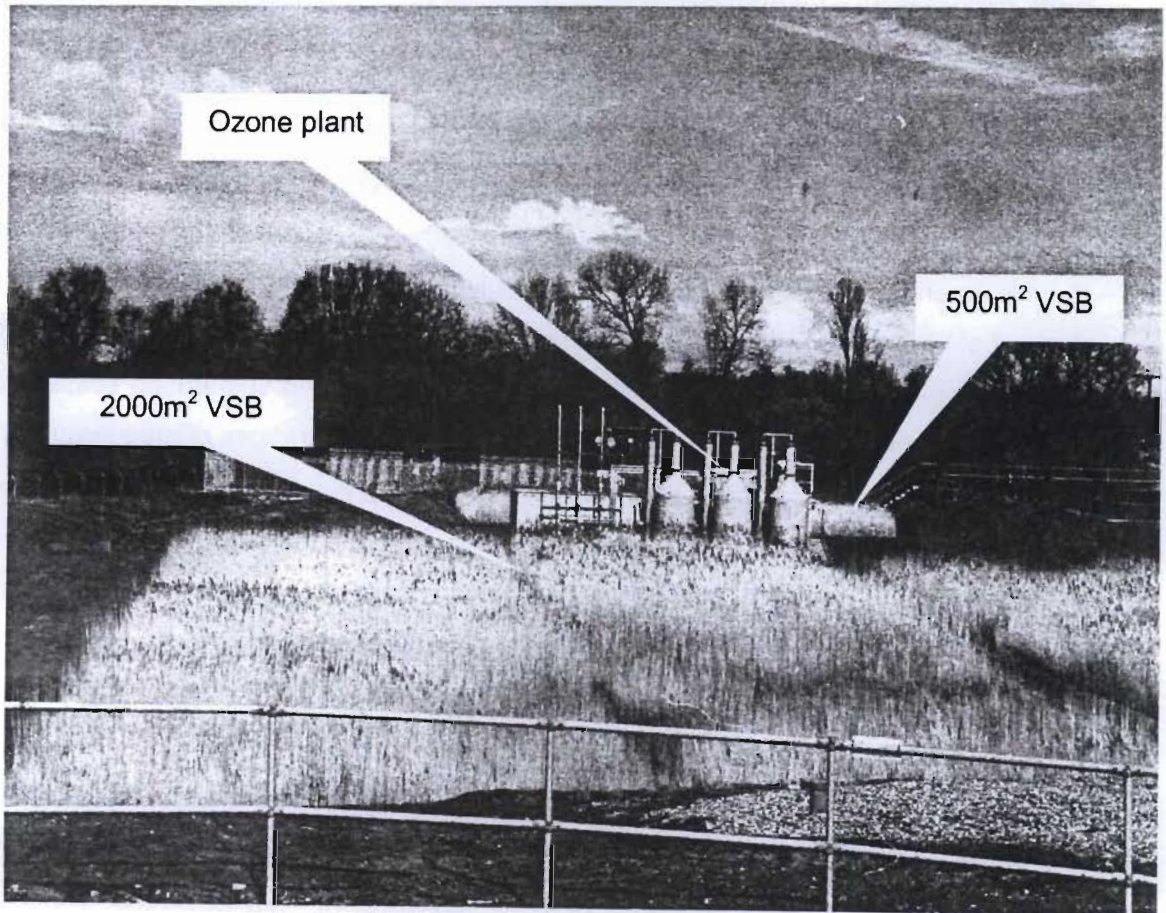


Plate 4.11: Buckden South constructed wetlands and ozonation plant.

Table 4.6: Typical influent and effluent values for the Buckden South CW and Ozonation plant, courtesy EnvirosAspinwall (UK).

Parameter	Influent, average SBR1 and SBR2	VSB1 effluent	Ozone plant effluent	VSB2 effluent
pH	8.2	7.9	7.9	7.5
SS (mg/l)	20	9	54	4
COD (mg/l)	412	390	395	350

BOD_5 (mg/l)	5	4	11	5
NH_4-N (mg/l)	0.8	0.7	0.8	1.0
Chloride (mg/l)	1710	1750	1740	1740
NO_3-N (mg/l)	403	394	388	382
NO_2-N (mg/l)	0.09	0.5	<0.1	0.4
Mecoprop (µg/l)	200	-	<0.05	<0.05
Isoproturon (µg/l)	300	-	3.0	1.9

4.1.6 Monument Hill Landfill Site and full scale VSB constructed wetland

(Robinson et al, 1998 and Robinson and Harris, 2001)

The Monument Hill Landfill Site is situated in a remote valley, east of Devizes in Wiltshire, Southern England. Part of the site is a wildlife reserve. It received domestic waste during the 1960s and 1970s and is unlined with a culverted stream beneath it, which has failed under the 10 to 15 m overburden of waste. The failure of the culverted stream resulted in the pollution of the Stert Watercourse, primarily from iron and suspended solids but also from ammoniacal nitrogen, typically 30 mg/l, and low concentrations of mecoprop, up to 19 µg/l. Due to the locality of the site, with no power supply and no avaliable sewer line for leachate disposal, a low cost, low maintenance treatment system was required. As part of the remedial works, in 1985, a new culvert was pipe jacked to direct the stream around the landfill. However, the old culvert remained and continued to pollute the stream. Further site investigation works in 1992 aimed at determining the leachate and stream flow rates and qualities, comparing them with seasonal and daily weather variations. The treatment option selected was an 1800 m^2 synthetically lined, gravel filled VSB, 600 mm deep and planted with pot-grown *Phragmites australis.* The VSB system was commissioned in July 1996. It consists of an initial iron settlement tank (Plate 4.12) to maintain the low maintenance requirements of the reed bed and aid in the prevention of the clogging of the bed with iron, as it was found that 14 percent of the iron in the leachate had already precipitated out of solution before it had left the old culvert. The inlet distribution channel which lead from the iron settlement tank was a channel laid across the width of the bed, which could easily be cleaned and maintained, the outlet system was similar to that used in the Buckden South CW presented earlier, with a water level control device. The typical operating concentrations and treatment performance of the VSB are presented in Table 4.7.

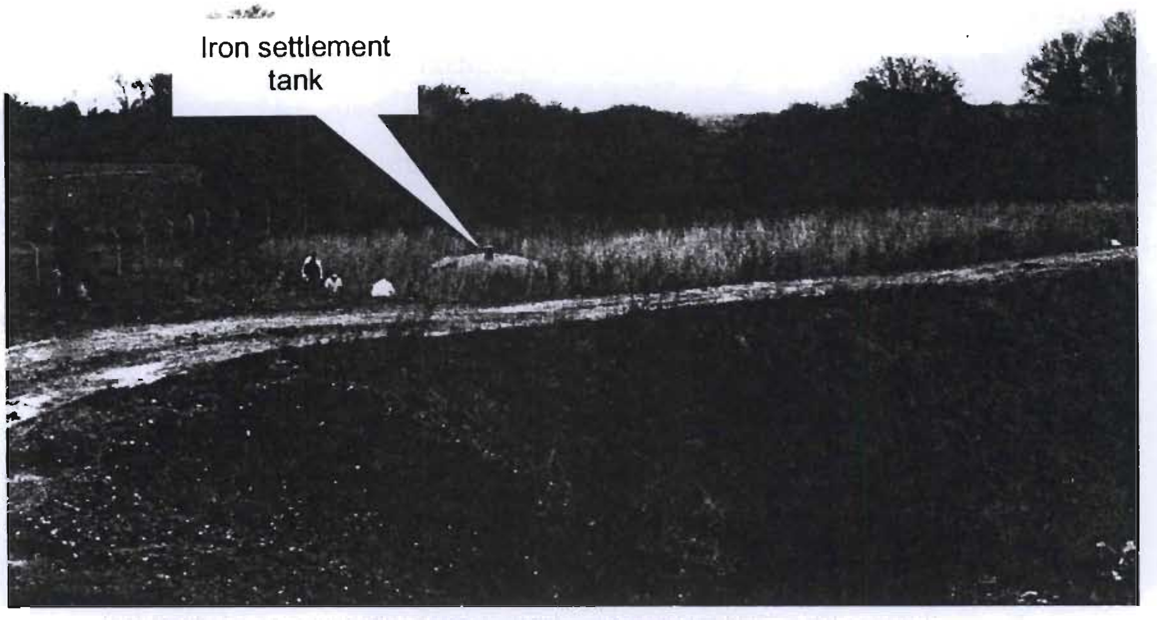


Plate 4.12: Monument Hill VSB constructed wetland and iron settling tank.

Table 4.7: Typical operating concentrations and treatment performance for the Monument Hill VSB, from Robinson et al (1998).

Parameter	Old culvert	After Iron settling tank	VSB effluent
pH	6.8	6.9	7.4
BOD_5 (mg/l)	<2	<2	<2
NH_4-N (mg/l)	19.4	19.4	11.8
Fe (mg/l)	16.9	12.2	<0.6
SS (mg/l	42	42	3
Chloride (mg/l)	78	77	76
Mecoprop (μg/l)	9.4	10.5	2.68

The above results show 28% removal of iron by the settling tank, and a further 95% removal occurring in the VSB.

4.1.7 Sundon Landfill Site and full scale landfill leachate treatment plant

(Robinson and Harris, 2001)

Sundon Landfill, in the United Kingdom, located east of Toddington Service station in Bedfordshire has received a mixture of wastes during its operational life; leachate generated by the landfill has accumulated within it and at the toe of the landfill where it has been diluted by rainfall. An extensive restoration program has led to the design and construction of a twin covered SBR treatment system, followed by 4000 m^2 of constructed wetlands (Plate 4.13). The nitrification SBR system, commissioned early 1997, has a design flow of up to 300 m^3/d. Aeration is achieved using venturi aerators. The strongly methanogenic leachate (Table 4.8) is pumped from bore holes, drilled to the base of the waste body, to a balancing lagoon where it is dosed into the twin-SBR system at rates of up to 250 m^3/d, the effluent (Table 4.8) of which is discharged for polishing through the constructed wetlands. The diluted leachate from the toe of the landfill is pumped directly to the reed beds at a rate of approximately 300 m^3/d. The final effluent from the constructed wetlands is discharged 2km to a local sewage treatment works.

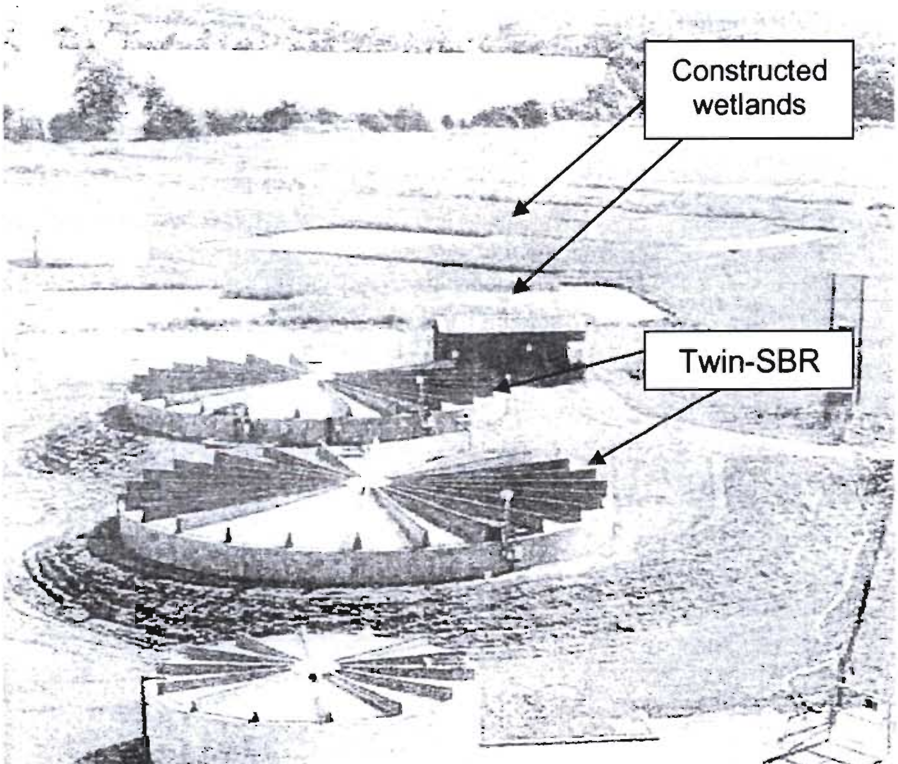


Plate 4.13: Sundon Landfill leachate treatment plant, courtesy of EnvirosAspinwall (UK).

Table 4.8: Typical influent and effluent values for the Sundon Landfill treatment system, courtesy of EnvirosAspinwall (UK).

Parameter	Influent Leachate	SBR Effluent
pH	8	7.5
COD (mg/l)	572	184
BOD_5 (mg/l)	25	3
NH_4-N (mg/l)	413	<0.3
NO_3-N (mg/l)	1.5	186
NO_2-N (mg/l)	<0.1	<0.1
Suspended solids (mg/l)	130	83
Zinc (mg/l)	<0.03	<0.03
Iron (mg/l)	3.7	<0.6

The 4000 m^2 of VSB constructed wetlands, designed for polishing to high standards before discharge to a local sewerage works, is lined with a synthetic liner system, and filled with a gravel medium to a design depth of 600mm and vegetated with *Phragmites*, it receives up to 250 m^3/day of effluent from the twin SBR treatment tanks and up to 300 m^3/day of diluted leachate pumped directly from lagoons at the toe of the landfill. The influent to the wetlands is fed from a holding tank (Plate 4.14), into an inlet distribution system (Plate 4.15), which comprises a channel laid across the width of the cell and a central feed point. The outlet arrangement allows for water level control within the wetland. Typical operating results from the Sundon CW are presented in Table 4.9.

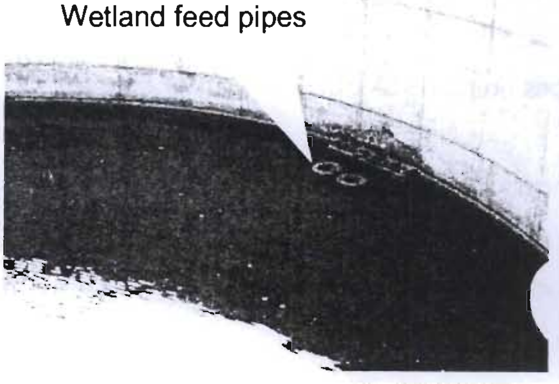


Plate 4.14: Wetland influent holding tank.

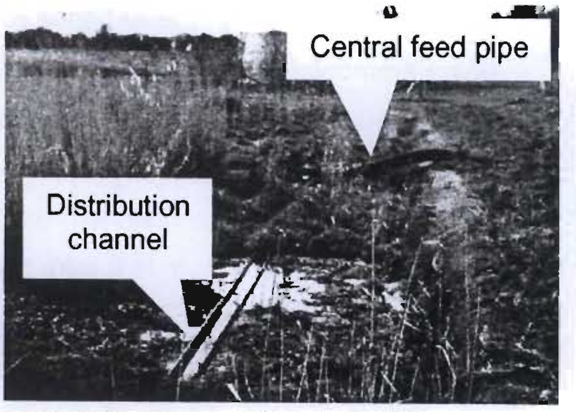


Plate 4.15: Inlet distribution channel.

Table 4.9: Typical operating results from the Sundon CW system, courtesy of EnvirosAspinwall (UK).

Parameter	SBR effluent	Diluted leachate	VSB effluent
pH	7.5	7.7	7.3
COD (mg/l)	184	146	93
BOD_5 (mg/l)	3	4	<2
NH_4-N (mg/l)	<0.3	2.9	<0.3
NO_3-N (mg/l)	186	25	31.3
NO_2-N (mg/l)	<0.1	5.8	0.6
Suspended solids (mg/l)	83	204	48
Zinc (mg/l)	<0.03	<0.03	<0.03
Iron (mg/l)	<0.06	1.4	<0.06

4.1.8 Judkins Landfill Site and full scale VSB constructed wetlands

(Robinson and Harris, 2001)

The Judkins Landfill Site is situated in a clay lined quarry. The generated leachate has, however, still managed to migrate into the ground water below the site. In an attempt to remediate the problem, the ground water is pumped out from the base of the site and passed through 2 in-parallel VSB constructed wetlands, with a surface area of 7800 m^2 and a design flow 1000 m^3/day. The wetland was vegetated with *Phragmites* seedlings (Plate 4.16). It was lined with a synthetic liner (Plate 4.17) and filled with a gravel-bedding medium to a depth of 600 mm. The influent leachate was very diluted and was thus placed directly through the wetlands without any pre-treatment, the main design parameter being ammoniacal nitrogen, with a concentration of approximately 20 mg/l is removed by the VSB to a consent value of 5mg/l, before being discharged to a water course.

The inlet structure is a channel, laid across the bed width, with a central feeder (Plate 4.18). The outlet structure is a perforated pipe situated on the outlet end along the bottom of the bed. Water level controls are provided as well as a rodding hole (Plate 4.19) in the outlet pipe for maintenance and cleaning.

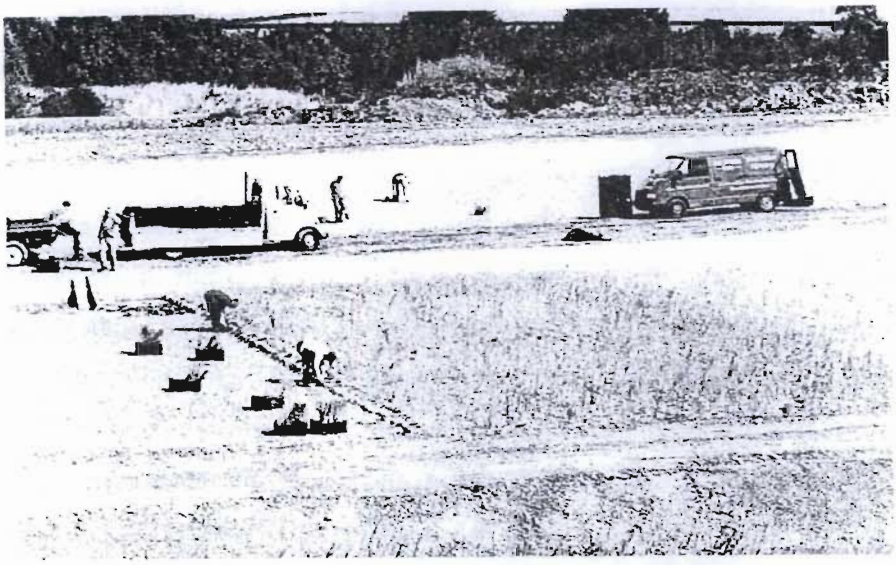


Plate 4.16: Planting of *Phragmites* seedlings, courtesy of EnvirosAspinwall (UK).

Plate 4.17: Earth works and placing of synthetic liner during construction, courtesy EnvirosAspinwall (UK).

Plate 4.18: Inlet channel with central feed.

Plate 4.19: Outlet rodding hole.

4.1.9 Kranskop full scale, municipal wastewater, treatment plant

The Kranskop, 2 unit parallel VSB constructed wetland, was initially designed according to the EC Constructed Wetland Design Guidelines (EC/EWPCA, 1990) to treat septic tank effluent (Wood, 1999). The 'cells' were described by Wood (1999) as each having a length of 40 m and width of 55 m, filled with a fine gravel to a depth of about 600 mm and planted with *Frogmouths Spp.*. During the site visit it was found that the local authority had little knowledge of the constructed wetlands, believing them to be "soak away pits". The site visit pointed out that the two cells actually received two different types of waste streams. One cell received a secondary settled effluent from a primary sewerage aeration tank (Plate 4.20) and the other received a mixed sewerage-runoff influent from a bus station. The cell that received influent from the aeration tank was 1.5 m wide with an indeterminable length, as it disappeared into the surrounding vegetation. Its bed gradients were found to be too large, favouring overland flow. It was vegetated with mixed weeds and there were no visible inlet and outlet structures. The other cell was (Plate 4.21) about 3 m wide also with an indeterminable length. The inlet structure (Plate 4.22) was a pit where the influent was allowed to pond and soak downstream, giving rise to unpleasant odours and mosquitoes. There was no visible outlet structure. During the site visit the vegetation, which was referred to as "Bugweed", was being cut back in order to "dry the solids". The effluent from both cells, which was not monitored, was believed to flow into a nearby stream, located at the bottom of the valley.

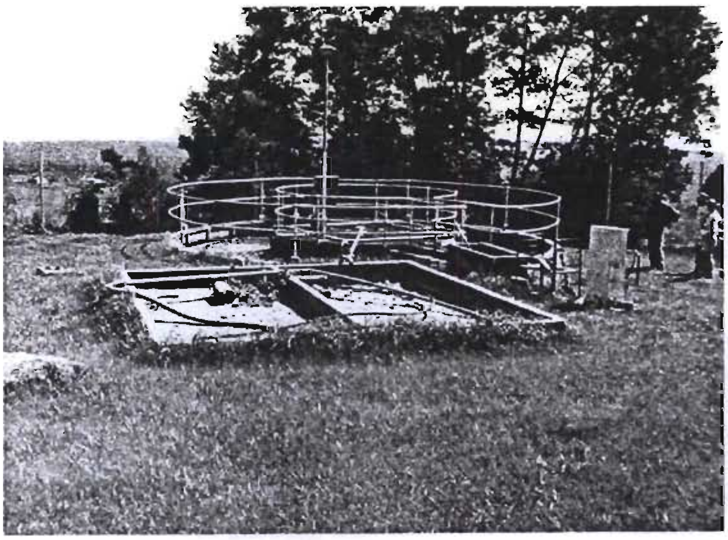


Plate 4.20: Kranskop, primary sewerage aeration tank.

The system was found to be totally uncontrolled. However, there were no visual detrimental effects on the surrounding environment, possibly due to the low magnitude of the overall wastewater flows, which allowed for natural purification processes to occur.

Plate 4.21: Cell receiving influent from nearby bus station.

Plate 4.22: Inlet structure.

4.1.10 Bethlehem full scale, municipal wastewater, treatment plant

The Bethlehem municipal wastewater treatment plant is located in Bethlehem, a small rural town in the Northern Free State. It consists of an array of treatment processes designed to accommodate flows of up to 4500 m^3/day, including a degritting chamber, a balancing lagoon, sedimentation tanks, 600 m^3 and 2000 m^3 SBR systems, a biological trickling filter, chlorination channels and 18200 m^2 of constructed wetlands. The plant was being upgraded during the visit, to include an activated sludge system. The VSB system (Figure 4.6) consists of five existing maturation ponds converted into CW, designed to polish the effluent from the primary treatment processes before final chlorination and discharge to a nearby stream. The cells have an average design depth of 650 mm with an inlet depth of 400 mm and an outlet depth of 900 mm and are planted with *Phragmites* and *Frogmouths Spp.*. They were filled with a rock medium, from a decommissioned biological filter, and railway ash, the latter with the intention to reduce phosphate levels in the effluent. Flows from the primary treatment processes are combined before discharge into the CW. Each cell's flow is regulated by a manual T-section valve feeding off the main peripheral influent pipeline that is connected to each inlet. There were no flow meters measuring the actual flow into each bed and the hydraulic loading to individual beds was found to be erratic, from a few hundred cubic metres to 2000 m^3/d through bed 5 alone (Wood, 1999). The influent flow to each bed was distributed across the bed width through a perforated pipe, which was buried within the gravel. The effluent was collected by a similar perforated pipe situated at the bottom of the bed and connected to an outlet sump capable of controlling the level within the bed (Plate 4.23). Dye tracer tests conducted by

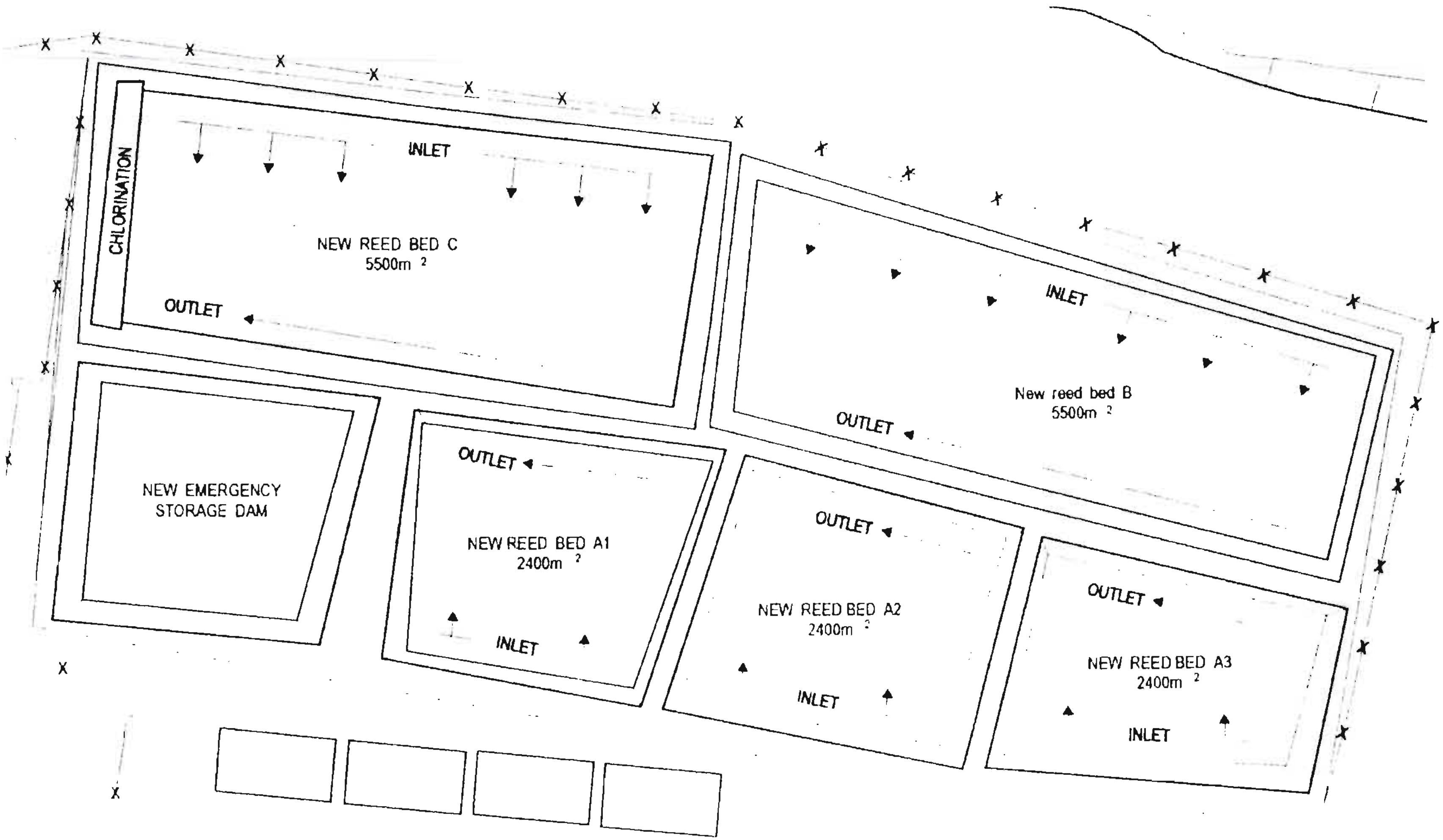


Figure 4.6: Schematic layout of Bethlehem constructed wetland system, from Wood (1999).

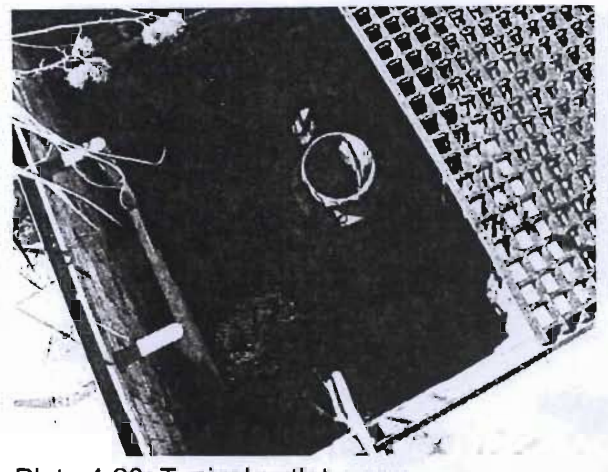


Plate 4.23: Typical outlet sump.

Wood (1999) showed the tendency for short circuiting between the inlet and outlet, with the predominant flow path being between the inlet and outlet sump as the collection point was restricted to one corner of each bed. Wood (1999) recommended that encroaching vegetation in the inlet zone should be cleared away on a regular basis, in order to avoid the clogging of inlet nozzles with rhizomes. However, during the site visit it was noted that both the inlet and outlet zones where covered by overgrown vegetation (Plate 4.24 and 4.25). Cells one to three had very dense stands of vegetation (Plate 4.26), with very few 'dead' areas, which where largely present in cells five and four (Plate 4.27). The performance of each cell was not monitored and only the final water quality after chlorination was avaliable, typical values of the influent and final effluent from September 1992 to July1994 are presented in Table 4.10 below.

Table 4.10: Performance of Bethlehem Constructed Wetlands, courtesy of Laubscher Human and Lombard consulting engineers.

Parameter	Influent concentration	Effluent concentration
TSS (mg/l)	16	6
COD (mg/l)	50	30
NH_4-N (mg/l)	3.5	2.1
NO_3-N (mg/l)	7.5	4.9
pH	7.5	7.6
Phosphates (mg/l)	2.0	1.9
Chlorides (mg/l)	78	87

Most of the influent concentrations were already below the discharge standards, except for ammoniacal nitrogen and phosphates that required slight 'polishing'. Initially the phosphate removal was high. However, the removal capacity was quickly reached within the first year of operation, as expected (Wood, 1999), leading to poor removal efficiencies as expressed in Table 4.10. The limited COD removal was attributed to the residual nonreadily biodegradable organics remaining after the primary treatment processes, while limited ammoniacal nitrogen removal was related to the oxygenation capacity of the system and low influent concentrations. The poor nitrate nitrogen removal efficiency was due to the availability of a useable carbon source for denitrification, tests on the microbiological quality of the effluent showed up to 99 percent reduction in E. coli and 99.9 percent removal of Total Coliforms (Wood, 1999).

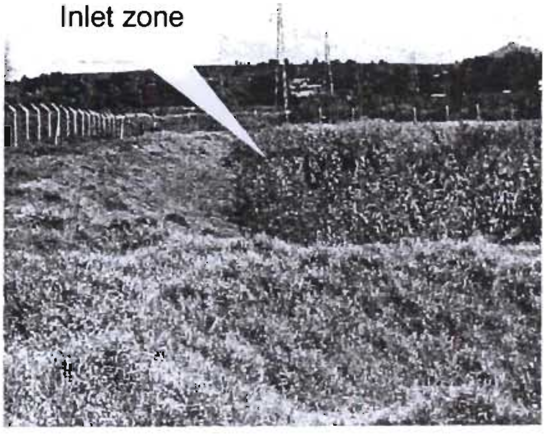


Plate 4.24: Overgrown inlet zone.

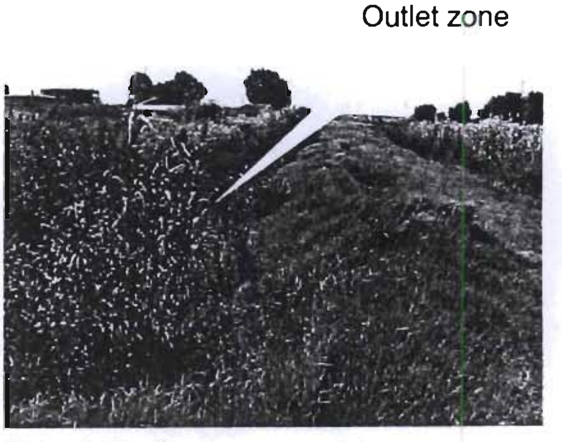


Plate 4.25: Overgrown outlet zone.


Plate 4.26: Dense vegetation stands in cells 1 to 3.

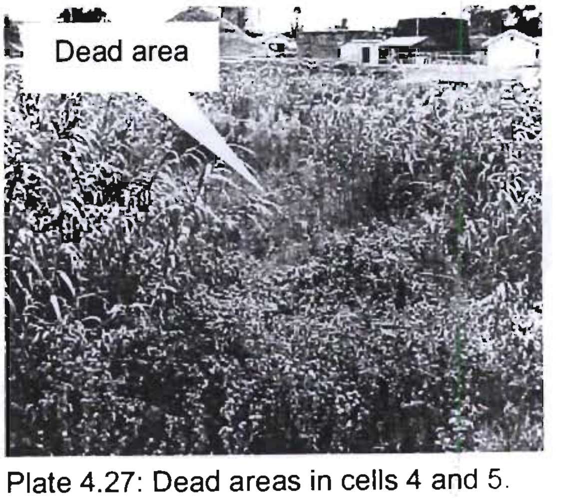


Plate 4.27: Dead areas in cells 4 and 5.

4.1.11 Kwazamokuhle full scale, municipal wastewater, treatment plant

The Kwazamokuhle sewerage treatment plant (Plate 4.28) is situated in Hendrina, Gauteng. It serves a small rural community and consists of a primary degritting chamber followed by two primary anaerobic ponds and a secondary anaerobic pond, from which portion of the effluent is sent to a biological trickling filter and the rest to an oxidation pond.

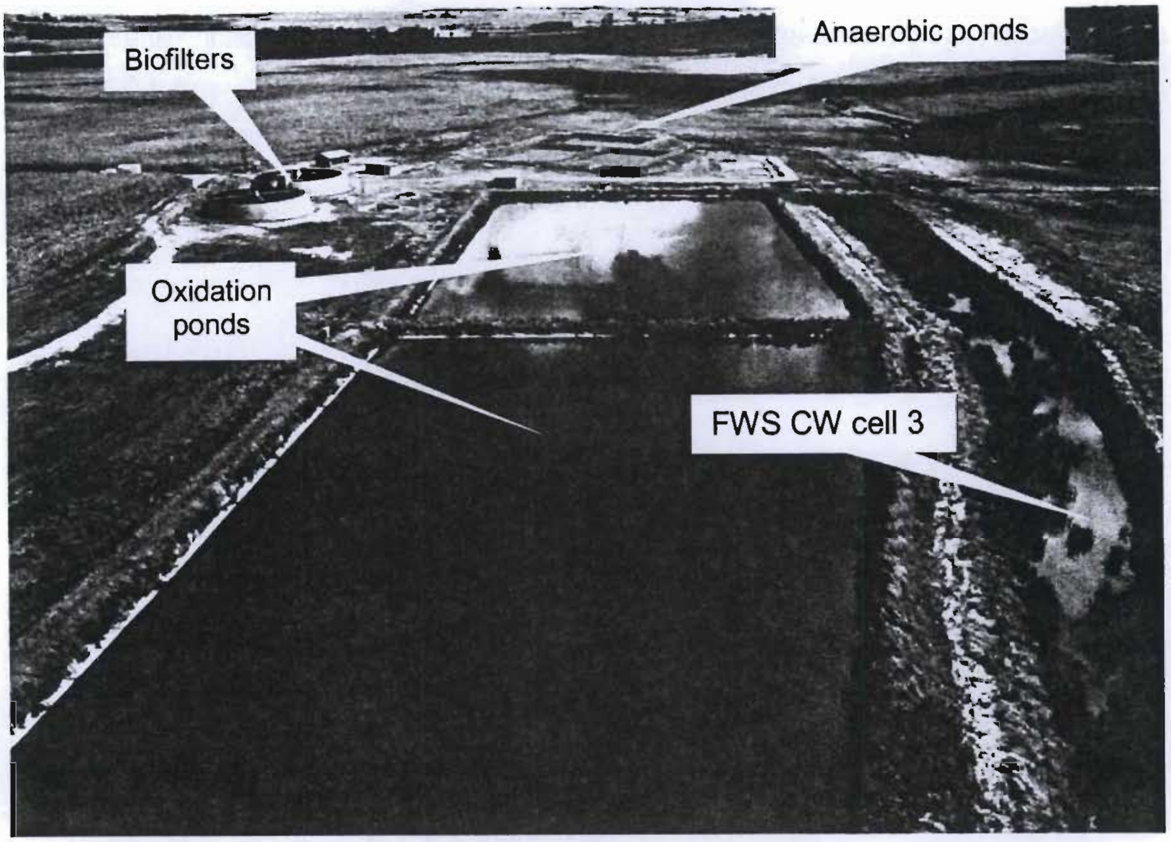


Plate 4.28: Arial view of the Kwazamokuhle sewerage treatment plant, courtesy of Wates, Meiring and Barnard consulting engineers.

The effluent from the oxidation pond is then recycled back to the primary anaerobic ponds, while a portion of the effluent from the biofilter is sent to an 11300 m^2 FWS constructed wetland, divided into three in-series cells, and the rest is recycled back to the primary anaerobic ponds. The plant was commissioned in April 1999. It has a design flow of 3.8 mega litres per day, with an actual flow of approximately 2.5 mega litres per day. The FWS constructed wetland was designed for polishing and removal of Faecal Coliforms in order to reduce the chlorine demand during final chlorination before discharge into the Klien Olifants River. As a secondary benefit the wetland was placed directly down gradient of the oxidation pond so that it would intercept any seepage from the pond. The CW area was excavated and topsoil was placed as a rooting medium. No synthetic liner system was used. It was vegetated by the rhizome method with *Typha Spp*. (Plate 4.29). During the site visit it was noted that fluctuations in the design water depth of 300 mm, forming deeper sections (up to 500 mm), inhibited emergent vegetation growth, allowing duckweed to cover these open water areas (Plate 4.30) and causing possible short-circuiting. The inlet and outlet structures did not allow for level adjustments as they comprised of long concrete weirs, susceptible to clogging from overgrown vegetation, affecting the even distribution of influent over the entire wetland width (Plates 4.31 and 4.32). The water quality of the influent and effluent from cell to cell was not monitored and only the initial influent and final effluent characteristics were avaliable as presented in Table 4.11 below.

Plate 4.29: Establishment of constructed wetlands, courtesy of Wates, Meiring and Barnard consulting engineers.

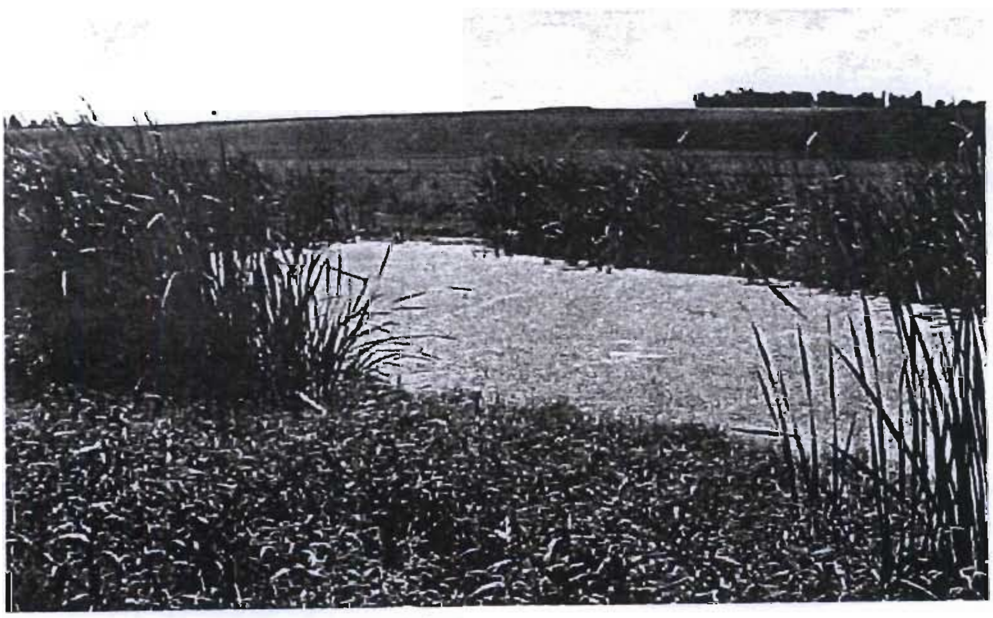


Plate 4.30: Open water areas, inhabited by Duckweed.

Table 4.11: Typical influent and effluent concentrations for the Kwazamokuhle Constructed wetland, courtesy of Wates, Meiring and Barnard consulting engineers.

Parameter	Influent concentration	Final Effluent Concentration
pH	6.05	6.59
TSS (mg/l)	24.13	18.40
COD (mg/l)	90.15	55.38
NH_4-N (mg/l)	7.30	4.56
NO_3-N (mg/l)	26.20	17.50

Note: Results on Faecal Coliforms removal were not avaliable.

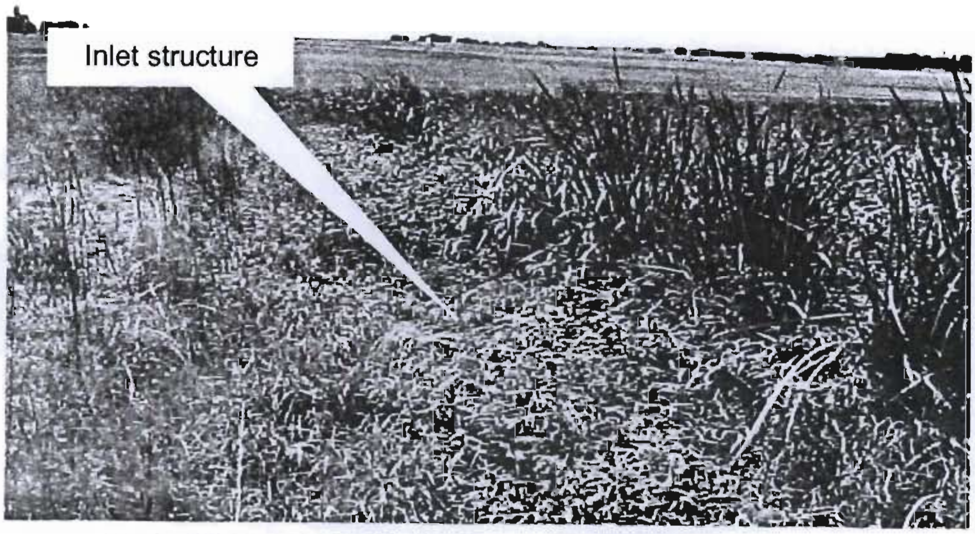


Plate 4.31: Overgrown inlet structure.

Plate 4.32: Concrete outlet weir, susceptible to organic clogging.

4.1.12 Dullstroom full scale, municipal wastewater, treatment plant

The 1 mega litre per day Dullstroom sewerage treatment plant (Plate 4.33) is situated in a valley, surrounded by a prestige area for farming, wildlife and waters used for trout fishing, the effluent from the site is regulated by special discharge standards. The plant was commissioned in September 1999 and consists of inlet degritting works, followed by anaerobic/anoxic mixing proceeded by aeration, with a portion of the effluent from the aeration chamber being recycled back to the anaerobic/anoxic mixing chamber and the rest is discharged to a final clarifier, effluent from the clarifier is proportioned between a series of storage ponds and 3600 m^2 of FWS constructed wetland, divided into two in-series cells and vegetated by the rhizome method, with *Typha Spp.*. The effluent from the storage ponds is used for irrigation on a nearby farmland, while the effluent from the constructed wetlands passes through a final chlorination chamber before being discharged to a small stream.

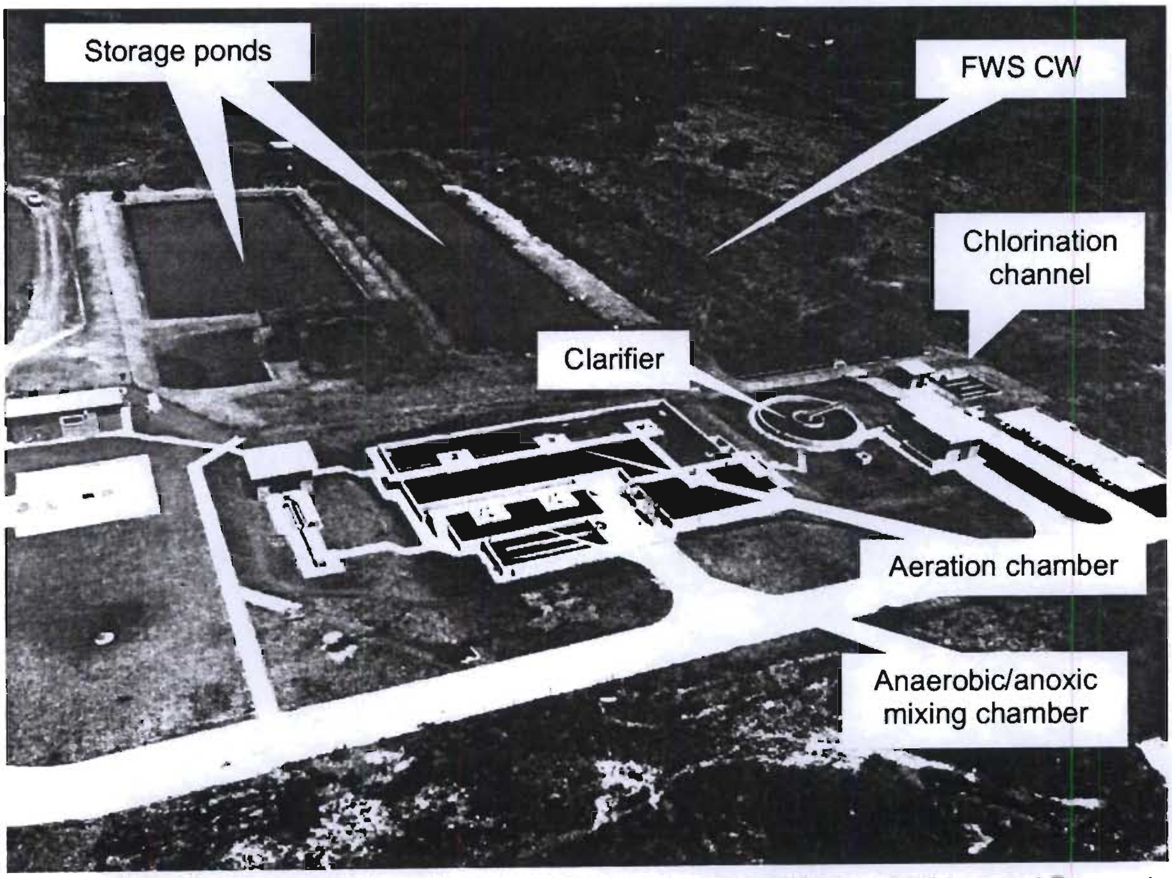


Plate 4.33: Dullstroom sewerage treatment plant, courtesy of Wates, Meiring and Barnard consulting engineers.

The FWS constructed wetland (Plate 4.34) was designed similarly to the Kwazamokuhle constructed wetland, with the aim to reduce final chlorine demand and for final effluent polishing. No synthetic liner system was used.

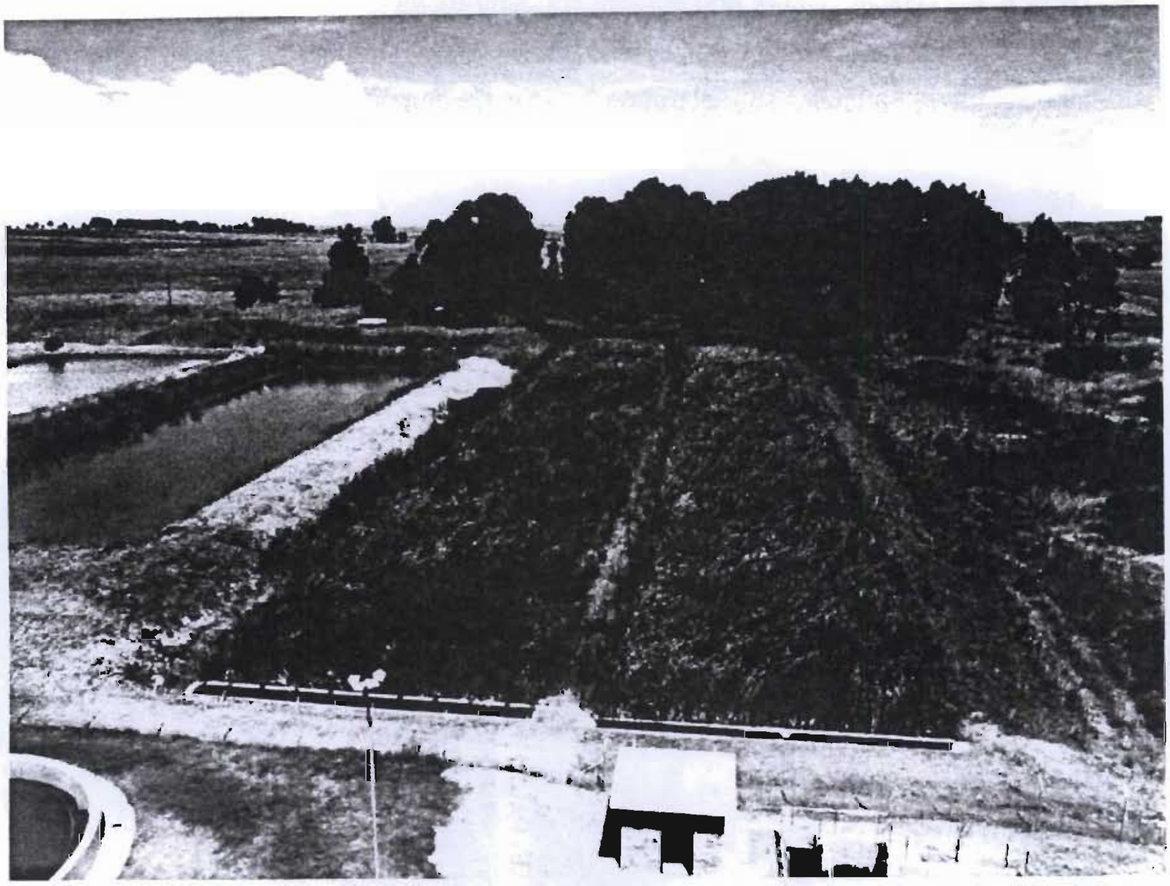


Plate 4.34: Dullstroom FWS constructed wetland, courtesy of Wates, Meiring and Barnard consulting engineers.

During the site visit it was noted that due to the improved grading of the wetland bed during construction in order to maintaining the design depth of 300 mm along the bed length, the vegetation had established much better than the Kwazamokuhle vegetation, with very few open water areas. Other noticeable improvements were the inlet structure (Plate 4.35), which had been very effectively modified, from the Kwazamokuhle inlet, with a concrete apron being provided on the upstream side to prevent the encroachment of vegetation into the inlet structure. There was still no water level control mechanism and the build up of biological solids on the outlet (Plate 4.36) required attention in order to maintain an even collection of the effluent from the entire width of the cell. As with the Kwazamokuhle wetlands no actual measurement of the flow into, between and from the wetlands was carried out. Water quality monitoring was also only performed for the inlet and final effluent, as presented in Table 4.12 below.

Table 4.12: Typical influent and effluent concentrations for the Dullstroom Constructed wetland, courtesy of Wates, Meiring and Barnard consulting engineers.

Parameter	Influent concentration	Final Effluent Concentration
pH	5.34	5.95
TSS (mg/l)	18.46	7.15
COD (mg/l)	56.27	36.54
NH_4-N (mg/l)	2.02	1.31
NO_3-N (mg/l)	11.38	8.06

Note: Results on Faecal Coliforms removal were not avaliable.

Plate 4.35: Improved inlet structure.

Plate 4.36: Biological buildup on outlet Structure.

4.2 Summary of case studies

The investigation of the practical application of CW during the site visits provided the opportunity to view different construction methods of full scale CW and typically used vegetation. The site visits also provided the opportunity to view issues described in the literature. Such as the encroachment of vegetation into the inlet structure of the CW, preventing even distribution of the influent across the width of the bed, as shown in Plates 4.24 and 4.31. Short circuiting was also evident in many of the FWS CW investigated, as shown in Plates 4.28 and 4.30.

After reviewing the data collected during the site visits it is clear that CW are only suitable for consistent polishing treatment of low concentrations of BOD, TSS, ammoniacal nitrogen and iron.

CHAPTER 5

5 PILOT SCALE STUDY ON VEGETATED SUBMERGED BED CONSTRUCTED WETLANDS

5.1 Introduction

Constructed wetlands have been used internationally as a cost-effective and integral part of many wastewater treatment plants to meet increasingly restrictive discharge requirements (Reed et al, 1995; Kadlec and Knight, 1996 and Lehman and Rodgers, 2000). Research into the use of pilot scale sequencing batch reactors for the full biological nitrification and denitrification of the Bisasar Road Landfill leachate was carried out by Strachan (1999) and Olufsen (1999). The treatment trials, described in Chapter 4, demonstrated that the full removal of nitrogen compounds could easily be achieved in a single sludge sequencing batch reactor system, under South African climatic conditions, for the Bisasar Road Landfill leachate. The system was found to be simple to operate and required low maintenance. However, the final effluent needed further 'polishing' of certain contaminants, especially residual COD prior its discharge into natural watercourses (e.g. Umgeni River) (Table 2.2). Being South Africa, a 'low gross income' country, it was necessary to consider an appropriate, cost effective and technically feasible 'polishing' treatment system. It was decided that pilot scale treatment trials using vegetated submerged bed constructed wetlands needed to be undertaken to assess the applicability and feasibility of such a system for the 'polishing' of the effluent from the pilot scale sequencing batch reactors.

5.2 Design of the polishing treatment system

5.2.1 Influent characterisation and flow rates

To simulate a future full-scale scenario, raw leachates from four collection points located on the Bisasar Road Landfill were mixed according to the predicted discharge percentages related to the total leachate flow from the site. This mixed leachate (Table 5.1) was then fed into the two pilot scale sequencing batch reactors, which were kept operational since the completion of the nitrification/denitrification treatability trials. The fully nitrified/denitrified supernatant (Table 5.2) was then monitored and characterised. Characterisation of the Umgeni river water quality was also conducted, upstream of the possible discharge stormwater outlet for the Bisasar Road Landfill, downstream of the stormwater outlet and at the stormwater outlet (Table 5.3). Durban Metro, Department of Wastewater Management, supplied the raw analytical data for the Umgeni River.

For the purpose of this research the leachate dosing pumps and timers where manipulated to allow for an average influent volume of 20 litres per reactor per day (giving a total average of 40 litres/day). This was the maximum allowable volume of influent which could be dosed due to the total volume of the 'wheelie bin' reactors. An amount of free-board was also required to account for foaming. In order to operate the pilot scale constructed wetlands on a daily basis, without the influence of any operational problems which might occur in the sequencing batch reactors, limiting the amount of effluent avaliable as influent to the wetlands, it was decided that a storage facility between the sequencing batch reactors and the wetlands, with a 'buffering' storage capacity, would be required.

Table 5.1: Characteristics of the Bisasar Road Landfill mixed raw leachate.

Parameter	# of Samples	Min.	Max.	Median	Mean	Std. Deviation
pH	9	7.80	8.49	8.30	8.23	0.2
COD	9	890	1183	1090	1075.67	99.85
NH_3-N	9	447	541	494	499.56	31.74
Alkalinity (as $CaCO_3$)	9	3095	3785	3625	3557.22	247.02
Chlorides	9	1659	1999	1859	1821.22	118.72
Conductivity (mS/m)	9	827	1063	1030	1012.78	74.33
TSS	9	19	75	50	48.44	18.95
E. Coli (cfu/100ml)	1	-	-	-	0	-
Fecal coliforms (cfu/100ml)	1	-	-	-	0	-
Total coliforms (cfu/100ml)	1	-	-	-	2	-
Microtox	1	-	-	-	*Nontoxic	-

Note: 1. Results in mg/l, except pH, conductivity, coliforms and Microtox.
2. Coliforms and Microtox tests conducted by Durban Metro, Department of Wastewater Management.
3. *Influence from stimulants in the leachate are expected, due to expected toxicity effects of ammoniacal nitrogen.

Table 5.2: Characteristics of the effluent from the Bisasar Road Landfill pilot scale nitrification/denitrification sequencing batch reactors.

Parameter	# of Samples	Min.	Max.	Median	Mean	Std. Deviation
pH	24	8.15	8.70	8.43	8.44	0.14
COD	24	476	612	511	526.5	40.1
BOD_5	8	5.7	18	11.7	11.93	4.82
BOD_{20}	8	18.2	31.6	24.8	25.15	5.19
NH_3-N	6	1.1	2.6	1.45	1.63	0.57
Alkalinity (as $CaCO_3$)	24	1070	1520	1242.5	1290.2	143.17
Chlorides	24	1729	1879	1811	1803.8	40.44
Conductivity (mS/m)	15	705	739	727	723.2	11.05
TSS	6	50	121	111	97.3	27.55
E. Coli (cfu/100ml)	1	-	-	-	0	-
Fecal coliforms (cfu/100ml)	2	0	4	-	2	-
Total coliforms (cfu/100ml)	2	0	2	-	1	-
Microtox	2	-	-	-	Nontoxic	-

Note: 1. Results in mg/l, except pH, conductivity, coliforms and Microtox.

2. Coliforms and Microtox tests conducted by Durban Metro, Department of Wastewater Management.

Table 5.3: Characterisation of sample points along the Umgeni River, conducted at the Durban Metro laboratories (Department of Wastewater Management).

Parameter	Upstream of Stormwater Outlet	Downstream of Stormwater Outlet	Stormwater Outlet
pH	7.5 (7.2;8.2)	7.4(7.2;7.5)	-
Conductivity (mS/m)	25.8(21;39)	42.3(26;90)	-
COD (mg/l)	25.5(10;44)	29(12;52)	-
NH_4-N (mg/l)	0.88(0.5;3.2)	1.14(0.5;3.4)	-
Total coliforms (cfu/100ml)	24542(3000;131000)	73385(4000;960000)	5000
E. coli (cfu/100ml)	4289(500;40000)	17068(500;260000)	0
Microtox	Nontoxic	-	Nontoxic

Note: 1. Format of results; mean(Min.;Max.).

2. Only 1 sample was taken at the stormwater outlet.

The characterisation of the effluent from the sequencing batch reactors confirmed that residual COD and TSS would require further 'polishing' in order to meet the general discharge standards. However, with ratios of 0.05 and 0.02 for BOD_{20} and BOD_5 to COD respectively it would be difficult to reach the specified general standard of 75mg/l. The characterisation of the Umgeni River pointed out the further need for the reduction of conductivity, with an average value between the upstream and downstream sampling points of 34.05 mS/m setting the general discharge standard at a maximum of 184.05 mS/m (Table 2.2). The results of the toxicity tests were of concern as the Microtox test found the mixed raw leachate to be nontoxic, even though the ammoniacal nitrogen concentrations were above the toxic limit. This anomaly was believed to be due to stimulants in the mixed raw leachate suppressing the toxic effects on the *photobacterium phosphorium* microorganisms used in the Microtox test (Jackson, 2001). This was of some concern when evaluating the other Microtox results, as it was not possible to asses if the same 'masking' effects were prevalent.

5.2.2 Local climatic conditions and modeling

Due to the scale and the expected climatic sensitivity of the pilot scale constructed wetland system, a local climatic study of the area was undertaken. Data for the Class-A-Pan evaporation, over a thirty two-year period, and other climatic data, taken over a twenty-year period, were collected from the South African Weather Bureau. Other climatic data included: wind speed, temperature, relative humidity and possible percentage sunshine, and are presented in Figure 5.1 (Raw data in Table B2, appendix B). The potential evapotranspiration was modeled using three techniques: the 0.7 and 0.8 times the Class-A-Pan evaporation method, the Pan Factor method proposed by Kadlec and Knight (1996) and the estimated water budget equations proposed by Bavor et al. (1988). Ideally the precipitation data for the local area were required due to the variability of precipitation in space and time. However, these data were not available and an overall monthly average for the Durban area was used. The monthly totals of precipitation and evapotranspiration were compared and the monthly water budget is presented in Table 5.4 and Figure 5.2. Evapotranspiration was further refined into average daily rates per month and the results are represented in Figure 5.3.

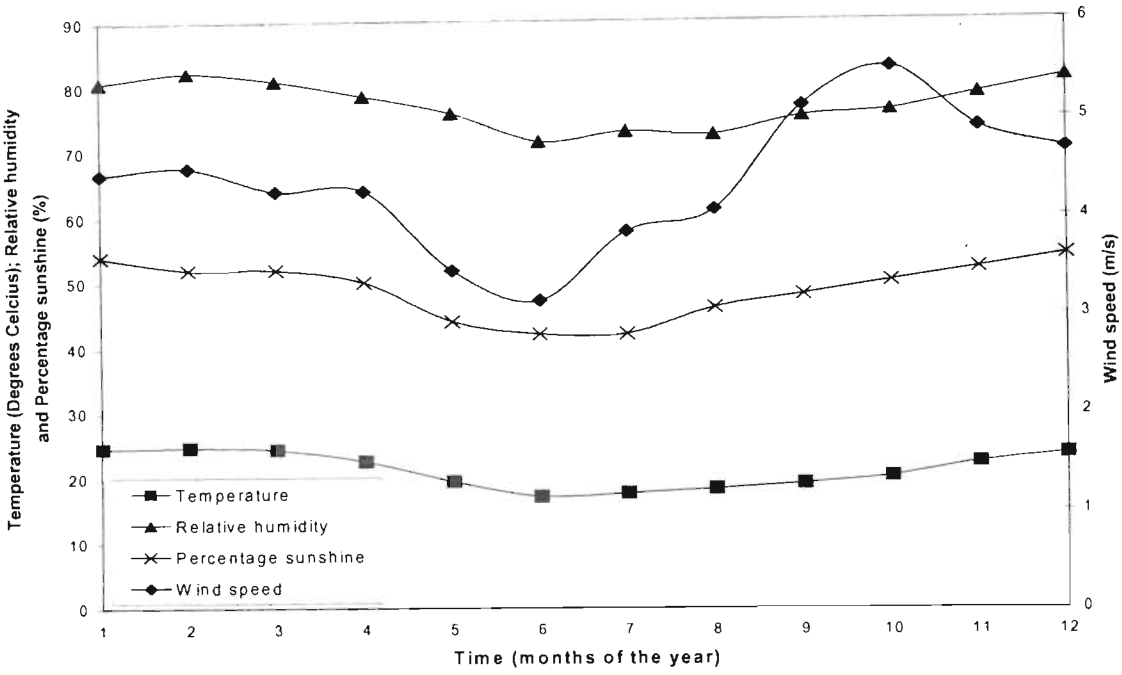


Figure 5.1: Average monthly climatic conditions for the Durban area, courtesy of the South African Weather Bureau.

Figure 5.1 indicates the different seasons and small climatic variations experienced in the Durban area, with a visible 'low' for all of the climatic conditions during the winter period, lasting of the order of four months, from May to August and a distinct decline and increase during autumn (March and April) and spring (September and October) respectively. An increase in wind speeds, above the annual average, starting in September and following through into early summer (November) may also be seen. The climatic conditions peak during the summer months, lasting from November to February. The climatic variations presented in Figure 5.1 indicate that the evapotranspiration would be at a maximum during summer. However, it only gives qualitative evidence and further quantitative analysis was required. The overall cyclic nature of the climatic conditions, which has an impact on the overall performance of the VSB CW, is also evident. However, low standard deviations for the respective climatic conditions show relatively small variations from the yearly averages, indicating minor differences between the seasons for the Durban area. The extent of the impact which these variations may have on overall vegetative growth and nutrient recycling would have to be evaluated during the pilot scale study. The average yearly temperature of 21°C with a standard deviation of 2.8 does, however, indicate that the annual temperature conditions are favorable for microbial growth.

Table 5.4: Total average monthly precipitation and predicted evaporation/evapotranspiration values for the Durban area.

Months of the year	Rainfall	(0.7) A-Pan	Pan Factor method	(0.8) A-Pan	Bulrush	A-Pan	Cattail
Jan	134	142	150	162	193	203	231
Feb	113	128	136	146	173	182	208
Mar	120	128	136	147	174	184	209
Apr	73	97	100	111	131	139	159
May	59	77	78	89	105	111	127
Jun	28	64	63	73	87	92	106
Jul	39	72	70	82	97	103	118
Aug	62	90	88	103	122	129	147
Sep	73	97	98	111	132	139	159
Oct	98	116	119	132	157	165	189
Nov	108	122	127	140	166	175	199
Dec	102	147	155	168	199	210	239

Note: 1. All results in mm/month

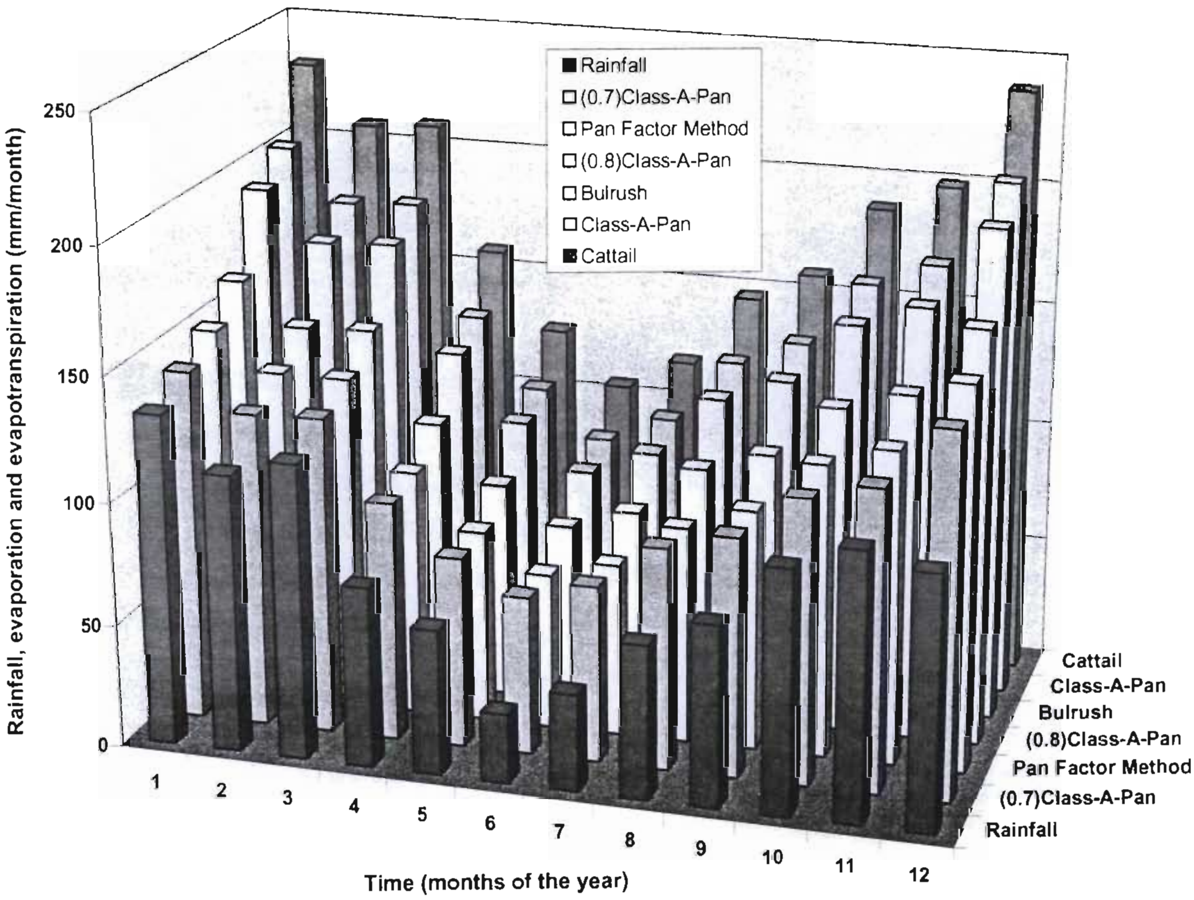


Figure 5.2: Total average monthly precipitation and predicted evaporation/evapotranspiration values for the Durban area.

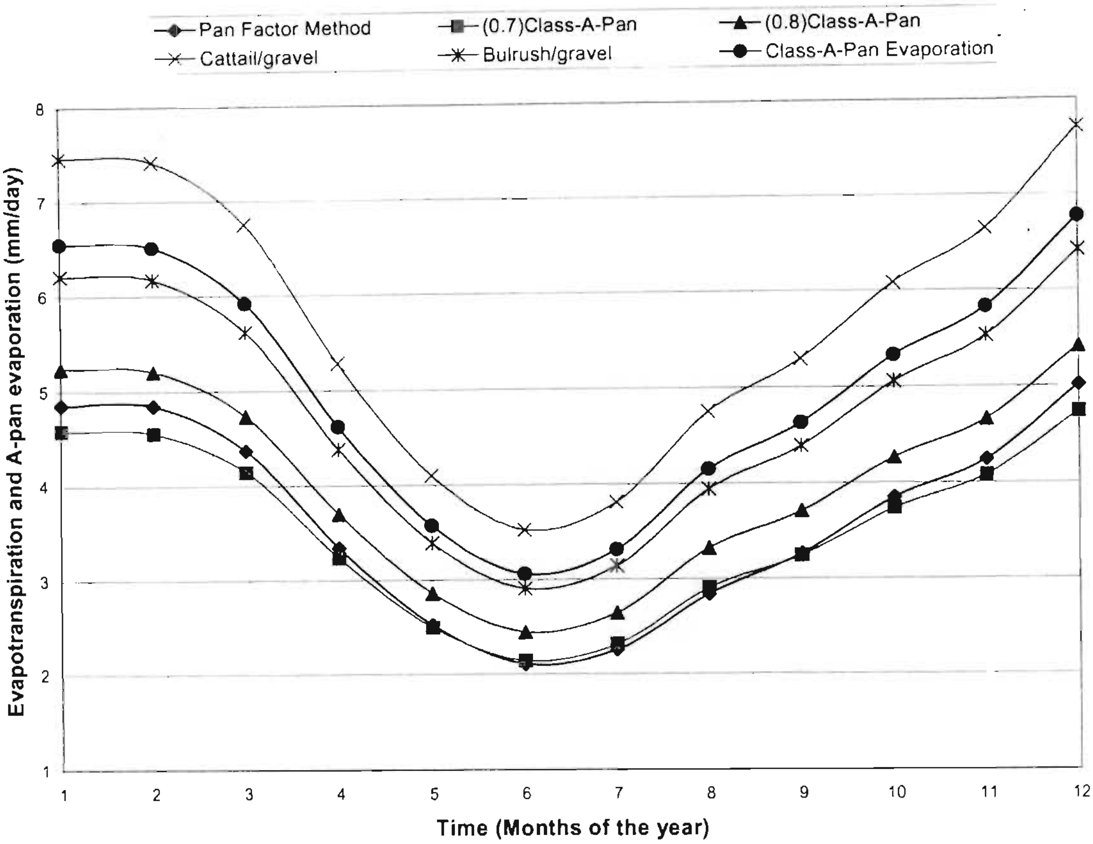


Figure 5.3: Average daily evaporation/evapotranspiration rates per month for the Durban area.

When comparing the total average monthly precipitation and predicted evaporation/evapotranspiration values for the Durban area, presented in Figure 5.2, it was confirmed that the monthly evaporation/evapotranspiration followed the expected trend, with maxima during the summer months. Figure 5.2 also highlights that Durban is a water deficient area when calculating the potential evapotranspiration using the respective methods. This finding differs from the classification, by DWAF, of Durban as a water surplus area; however, the climatic water balance used by DWAF is calculated using the wettest years on record and not the averages (DWAF, 1998). When comparing the different evapotranspiration calculations it was indicated that the gravel based systems planted with *Cattail* and *Bulrushes* had the highest potential evapotranspiration losses, with cattail losses exceeding the Class-A-Pan evaporation. When considering the data presented in Figures 5.2 and 5.3, it was pointed out that the evapotranspiration losses would be substantial and would have to be accounted for in the design of the pilot scale systems. For design purposes an evapotranspiration rate was estimated from Figure 5.3

as 4 mm/day, which was the annual average rate of evapotranspiration when multiplying the Class-A-Pan evaporation, by a factor of 0.8.

5.2.3 VSB sizing, media selection and geometric design

The VSB CW were sized using the rational surface area loading approach (Chapter 3 section 3.2.6). It was decided that four parallel VSB cells, three planted cells and one unplanted cell to provide a control, followed in series by simulated fresh water natural receivers would be selected for the research. Each cell would thus have an influent flow of 10 litres of supernatant from the SBR per day; which gave a COD loading rate of 5.3 g/day, a BOD_5 loading rate of 0.12 g/day and a BOD_{20} loading rate of 0.25 g/day. Typically a BOD_5 SLR of 6 $g/m^2/day$ has been used for VSB CW (Table 3.3). The polluting parameter of concern, however, was COD, which arguably comprised refractory organics that would be difficult to remove in the VSB CW and a large HRT with a low SLR would be necessary. An initial COD SLR of half the typical BOD_5 areal loading rate was selected for the design of the VSB treatment zone (3 $g/m^2/day$). The inlet and outlet zones were set at 0.5 m lengths. The design SLR gave a treatment surface area of 1.8 m^2. By using the design evapotranspiration rate of 4 mm/day, it was found that over the treatment area alone there would be a loss of 7.2 litres/day (almost ¾ of the influent). In order to counteract the losses due to evapotranspiration it was decided to add 10 litres of water from a spring situated on site. The spring water was analysed and its characteristics are presented in Table 5.5.

Table 5.5: Characteristics of the natural spring water, from the Bisasar Road Landfill site.

Parameter	# of Samples	Min.	Max.	Median	Mean	Std. Deviation
pH	9	7.36	8.41	7.71	7.80	0.31
COD	9	5	23	190	194	7
NH_4-N	9	0	0	0	0	0
Alkalinity (as $CaCO_3$)	9	220	270	255	256	15.09
Chlorides	9	110	260	190	194	44.60
Conductivity (mS/m)	9	105	118	109	109	3.97
TSS	9	0	9	2	3.11	3.44

Note: 1. Results in mg/l, except pH and conductivity.

Through mass balance calculations and using the mean values, the combined influent to the constructed wetland was estimated and its characteristics are presented in Table 5.6.

Table 5.6: Characteristics of the mixed pilot scale VSB CW influent.

Parameter	SBR supernatant	Natural Spring	Estimated influent mix
pH	8.44	7.80	-
COD	526.5	194	360
NH_4-N	1.63	0	0.82
Alkalinity (as $CaCO_3$)	1290.2	256	773
Chlorides	1803.8	194	998
Conductivity (mS/m)	723.2	109	416
TSS	97.3	3.11	50

Note: 1. Results in mg/l, except pH and conductivity.

The COD loading rate of the mixed influent was 7.2 g/day at a flow rate of 20 litres/day, which gave a treatment surface area of 2.4 m^2 for a COD SLR of 3 g/m^2/day and a design evapotranspiration loss of 9.6 litres/day. A width of 0.8 m and a length of 3 m was selected for the treatment zone in order to obtain a large length to width ratio of 3.75 to 1 which would help to avoid short circuiting of the flow through the small treatment area. With the selected inlet and outlet lengths the overall length of the VSB CW was 4 m with a width of 0.8 m; this gave a design evapotranspiration loss of 12.8 litres/day over the whole wetland area and an effluent flow of 7.2 litres/day. The overall bed depth of 700 mm was selected for unrestricted root growth. Due to preferential flow paths and the possibility of short-circuiting below the root zone, the VSB depth was also divided into two zones; the treatment zone where the major flow path would be encouraged to flow and the rooting medium zone (Figure 5.4). The treatment zone comprised of a 200 mm deep, 13.2 mm stone layer in which the vegetation was planted. The function of the 500 mm rooting medium layer, comprised of 50% topsoil and 50% 13.2 mm stone (Figure 5.5), was to encourage root growth throughout the upper gravel layer where the majority of the flow would take place, due to the higher hydraulic conductivity of the stone. This insured contact between the influent and the root zone. A synthetic geofabric filter was placed between the subsurface outlet system and the rooting medium to reduce the transportation of fines from the rooting medium into the outlet system.

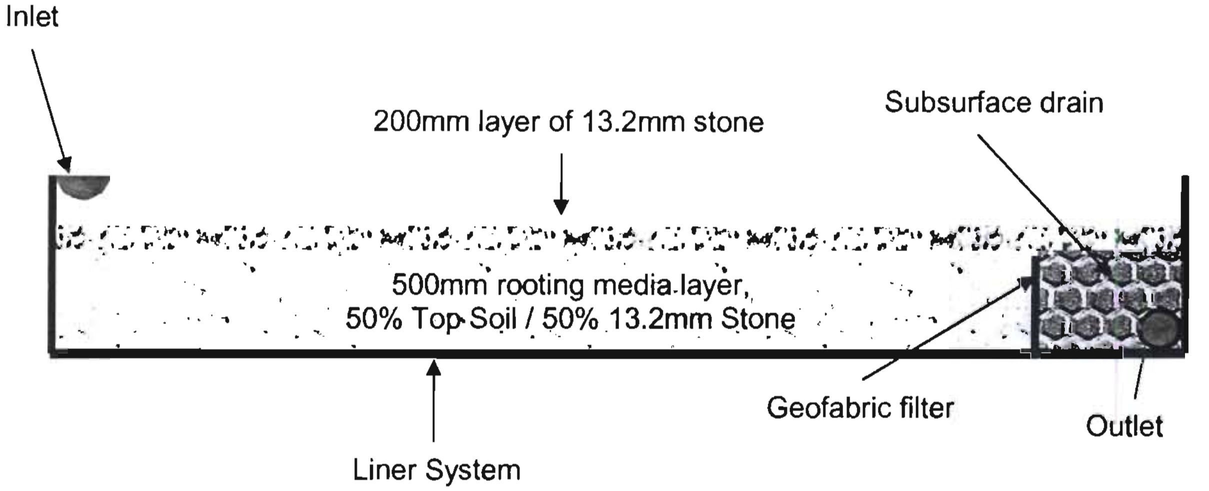


Figure 5.4: Schematic cross section of the VSB CW cell.

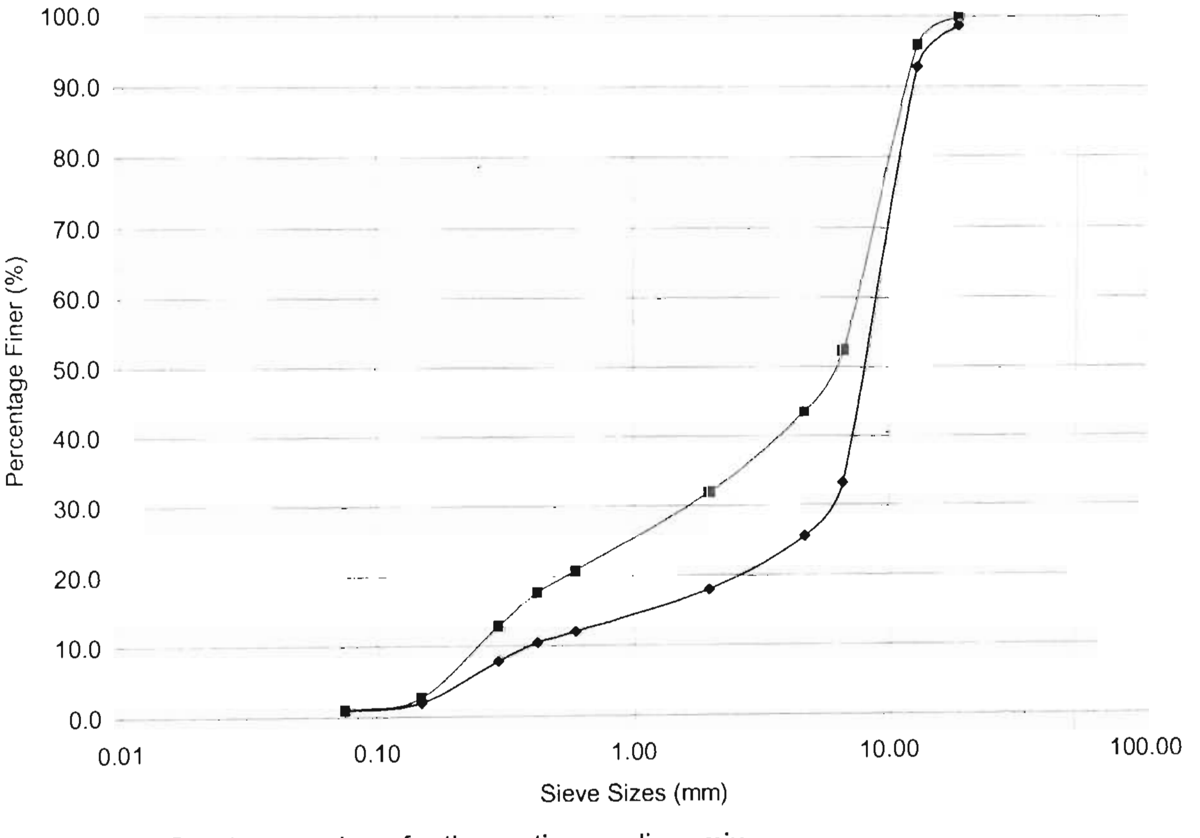


Figure 5.5: Grading envelope for the rooting medium mix.

The hydraulic losses through the system were checked using equations 3.4, 3.8, 3.19 and 3.20:

Q	= 0.0136 m^3/day (20 liters/day influent and 7.2 liters/day effluent)
A_c	= 0.16 m^2 (the cross sectional area of the vertical treatment zone)
u	= 0.085 m/day
D	= 0.0132 m
Re	= 0.013 at 20°C (<10)
k	= 15000 m/day (10% of clean k used = 150 m/day) (EPA, 2000)
x	= 4 m

Head loss = 2.3 mm

Due to the scale of the system the head loss was regarded as negligible.

5.2.4 Inlet and outlet design

The function of the inlet was to distribute the influent evenly across the bed width. Initially a simple overflow weir system was selected, however, laboratory testing showed this type of system to be inefficient for the scale of the project. The low flows required a level weir edge, with an aluminum 'lip' to reduce surface tension effects which favored 'short circuiting' of the influent over one section of the weir. Further laboratory testing pointed out that a half section of 110 mm uPVC pipe, perforated at the bottom along the long axis, was the most suitable inlet system. The perforated holes were sized in order for the influent to 'pool up' in the half section, allowing for an even distribution of flow across the section, however, large enough not to allow an over flow. 'Clogging' of the perforations was of concern. However, due to the low loadings of solids to the system it was not perceived to pose a large maintenance and operational problem.

The function of the outlet system was to evenly collect the effluent over the width of the bed. It was decided to use a subsurface system comprising a perforated 110 mm uPVC pipe, surrounded by large hand placed stone to form a sub surface drain. 'Clogging' of the outlet system with fines from the rooting medium was of. A geomembrane filter was selected to be placed between the rooting medium and the outlet system to prevent this from happening (Figure 5.4). The barrier extended from the base of the cell up to 500 mm and across the entire width of the bed. It did not extend into the treatment zone. The outlet manholes were designed to allow for any effluent overflows or spillages to be channeled off into the nearby sewerline. The outlet hoses were designed to be fitted to a level control system, situated in the manholes, which comprised a vertical slider rod with a level adjustment fitting (Figure 5.7). The daily effluent was collected in 25 litre drums and manually measured.

5.2.5 Selection of the vegetation

Two types of grasses were selected for the research, *Vetiveria zizanioides* and *Leersia hexandra.* The reed *Phragmites australis* was also investigated in the research. *Leersia Hexandra* (Figure 5.6), which is indigenous to South Africa, is a perennial with long stout branched rhizomes; it grows in water either by rooting in the medium or floating on the surface of the water, it tends to establish its roots at water depths below 200 to 300 mm (KwaZulu Natal Herbarium, 2001). It has also been found growing near the inlet of many municipal wastewater CW in South Africa, as well as in uncontrolled leachate ponds on landfill sites (Lombard, 2000). It was selected due to its very dense rhizome system, which would be capable of spreading throughout the 200 mm deep treatment zone. *Vetiver zizanioides* (Plate 5.1) is a perennial grass, which was once thought to be confined to wetlands, however, it is now known that it thrives over a range of ecological conditions. It grows in dense hedges and has an extensive root system that can extend up to 3 m in depth (Plate 5.2). It has small rhizomes, which fold back on themselves and it is a sod-forming grass. Its clumps grow out and when they intersect with neighbouring ones they intertwine and form a sod giving the hedge its tight and compact characteristics. *Vetiver* is not indigenous to South Africa; however, the species

Figure 5.6: *Leersia hexandra*

studied are sterile and can only replicate by vegetative propagation. It is not seriously affected by pests or diseases and, in fact, has the ability to repel insects with an oil that it produces. It can grow in both highly acidic and alkaline soils with a pH ranging from less than 4 to 11. It grows well in damp sites such as swamps and bogs with a mean temperature range from 18 to 25°C; it also has high tolerances to aluminum and manganese at concentrations up to 550 mg/l.

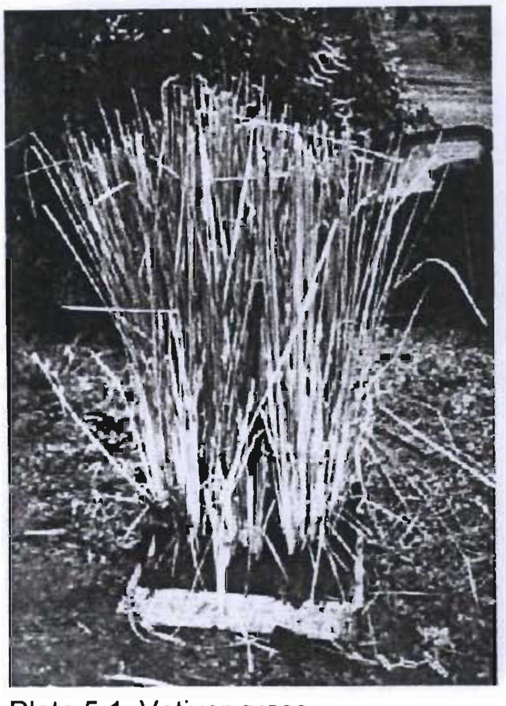


Plate 5.1: Vetiver grass.

Plate 5.2: Extensive rhizome system.

The organic oil that is emitted occurs primarily in the roots and has a sweet and pleasant odour, however, it may serve as an influent source of unknown organics into the wetland system that may reduce the efficiency of the system. The oil does not decompose in an alkaline medium and is extremely complex, containing more than 60 compounds including bicyclic and tricyclic sesquiterpenes hydrocarbons, alcohols and carboxylic acids (Borlaug et al, 1993; Vetiver Newsletter, 1996 and http://www.vetiver.org/TVN_about.htm). *Vetiver* has been used for rehabilitating contaminated land, sludge removal and stabilisation in CW and evapotranspiration beds for latrines. Research has also been conducted on the ability of Vetiver to purify landfill leachate and it was found that it was highly tolerant to landfill leachate, obtaining excellent removal rates for both ammoniacal nitrogen and phosphates (Vetiver Newsletter, 1996 and http://www.vetiver.org).

Phragmites australis has been used extensively in treatment constructed wetlands. It is a tall, emergent herbaceous perennial reed better known as the 'common reed' (Plate 5.3). It spreads laterally via vegetative growth of the rhizomes, which may penetrate up to 0.6 m in depth. Dense cover is possible within one year after planting at 0.6 m spacings. It has an optimum growth pH range from 2 to 8 and may tolerate moderate salinity of less than 45 g/l. It has a hydroperiod of 70 to 100% with a maximum water depth during flooding of 0.5 to 1 m, while it has also been found to be resistant to drought (Reed et al, 1995 and Kadlec and Knight, 1996).

Plate 5.3: Fully grown *Phragmites australis.*

5.2.6 Integrating the pilot scale SBR and VSB CW

In order to operate the VSB CW on a daily basis, a continual supply of influent was required. For this to occur without any influence from the SBR during periods of maintenance and operational problems, a 'buffer' storage volume was needed. It was decided to use two 2.5 m^3 tanks, for the storage of the effluent from the SBR, which were filled to capacity during the construction phase of the VSB CW. Four 50 litre high density polyethylene drums, used individually as header tanks for each cell, were selected, to allow for mixing and controlled volumetric dosing. It was decided to batch feed the CW every morning, to simulate the batch discharge of the SBR directly into the CW.

5.2.7 Design summary and overall layout

A summary of the design parameters is presented in Table 5.7 below:

Table 5.7: Design parameters.

Parameter	Design Value	Comment
COD SLR	3 $g/m^2/d$	For treatment zone
Total cell length	4 m	-
Inlet zone length	0.5 m	-
Outlet zone length	0.5 m	-
Treatment zone length	3 m	-
Cell width	0.8 m	-
Total surface area	3.2 m^2	-
Treatment surface area	2.4 m^2	-
Treatment zone depth	0.2 m	-
Rooting medium depth	0.5 m	-
Total cell depth	0.7 m	-
Length to width ratio	3.75 : 1	For treatment zone
Design freeboard	10 mm	From top of wetland surface
Design ET rate	4 mm/d	-
Design porosity	30 %	Figure 3.9
Design hydraulic conductivity	150 m/d	For treatment zone
Design head loss	2.3 mm	Over total cell length
Design influent flow rate	0.02 m^3/d	-
Design effluent flow rate	0.0072 m^3/d	-
Average daily flow rate	0.0136 m^3/d	-
Design superficial velocity	0.085 m/d	-
Design HLR	8.3 mm	For treatment zone
Design HRT	10.6 days	For treatment zone
Treatment zone stone size	13.2 mm	-
Rooting medium mix	50% top soil/50% 13.2 mm stone	
Vegetation	Phragmites australis	-
	Leersia hexandra	-
	Vetiver zizanioides	-

A cross section through a typical manhole is presented in Figure 5.7. The VSB CW were constructed within a brick shell, lined with a flexible geomembrane; a cross section through a typical cell is presented in Figure 5.8. The effluent from each VSB cell was discharged into a pond constructed to simulate the natural receiving environment of this landfill site. The ponds were planted with aquatic plants, indigenous to the area, and fish species were also introduced to assess whether the residual COD in the effluent from the VSB cells had any detrimental affect on the receiving environment.

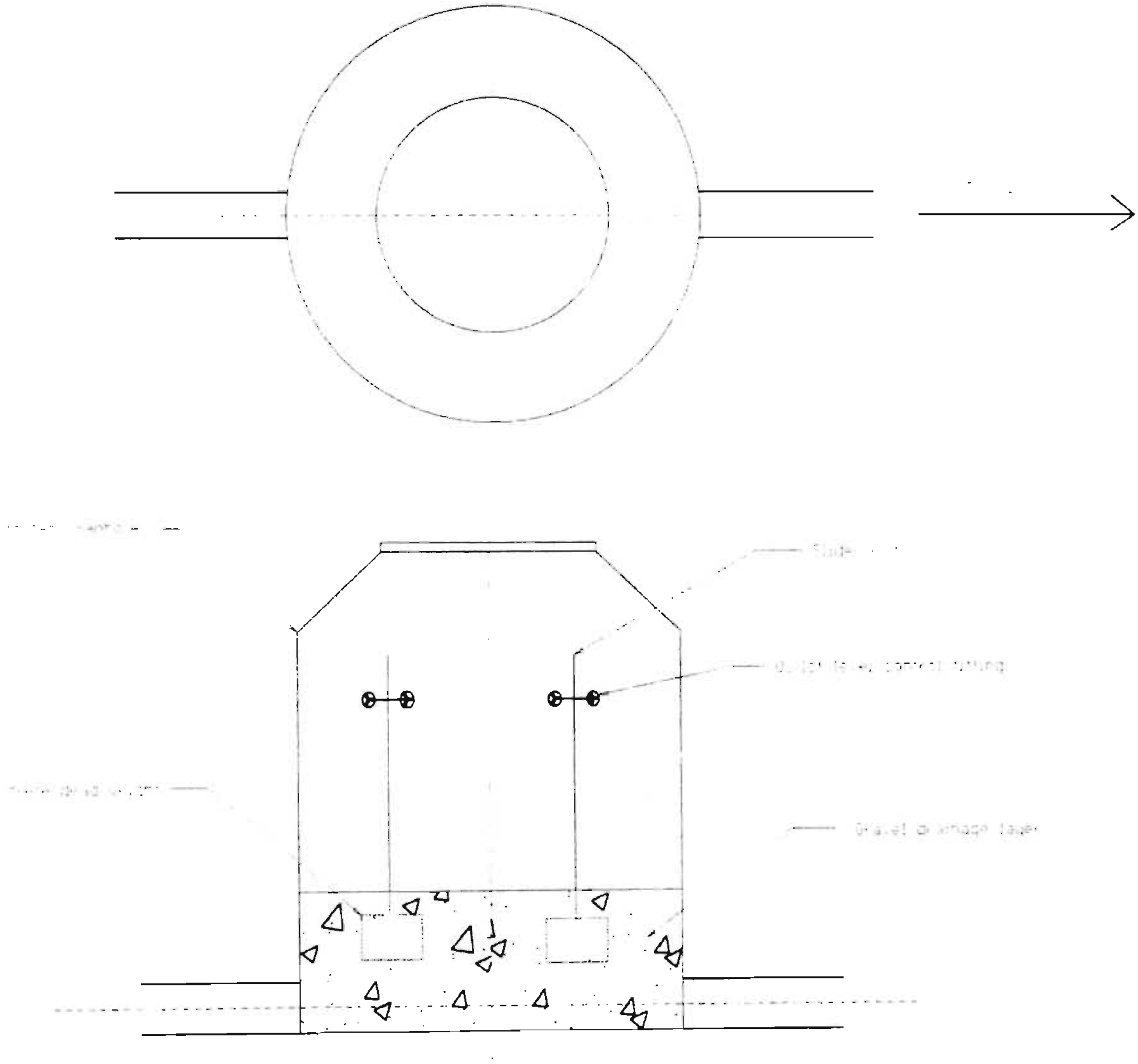


Figure 5.7: Cross section through a typical manhole.

The overall schematic layout of the system is presented in Figure 5.9 and the pipe layout and flow directions are presented in Figure 5.10.

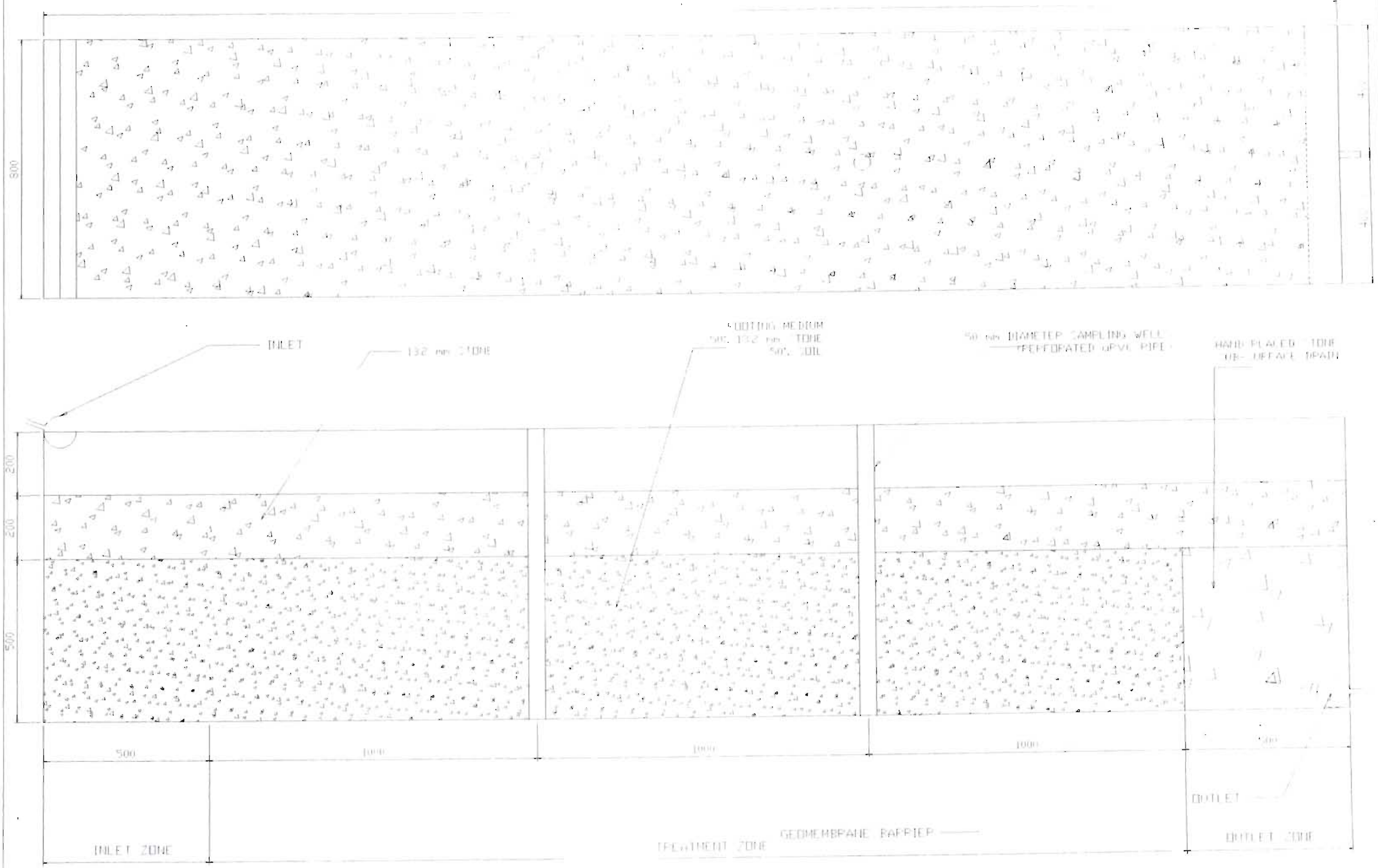


Figure 5.8: Plan and cross sectional view of a pilot scale VSB

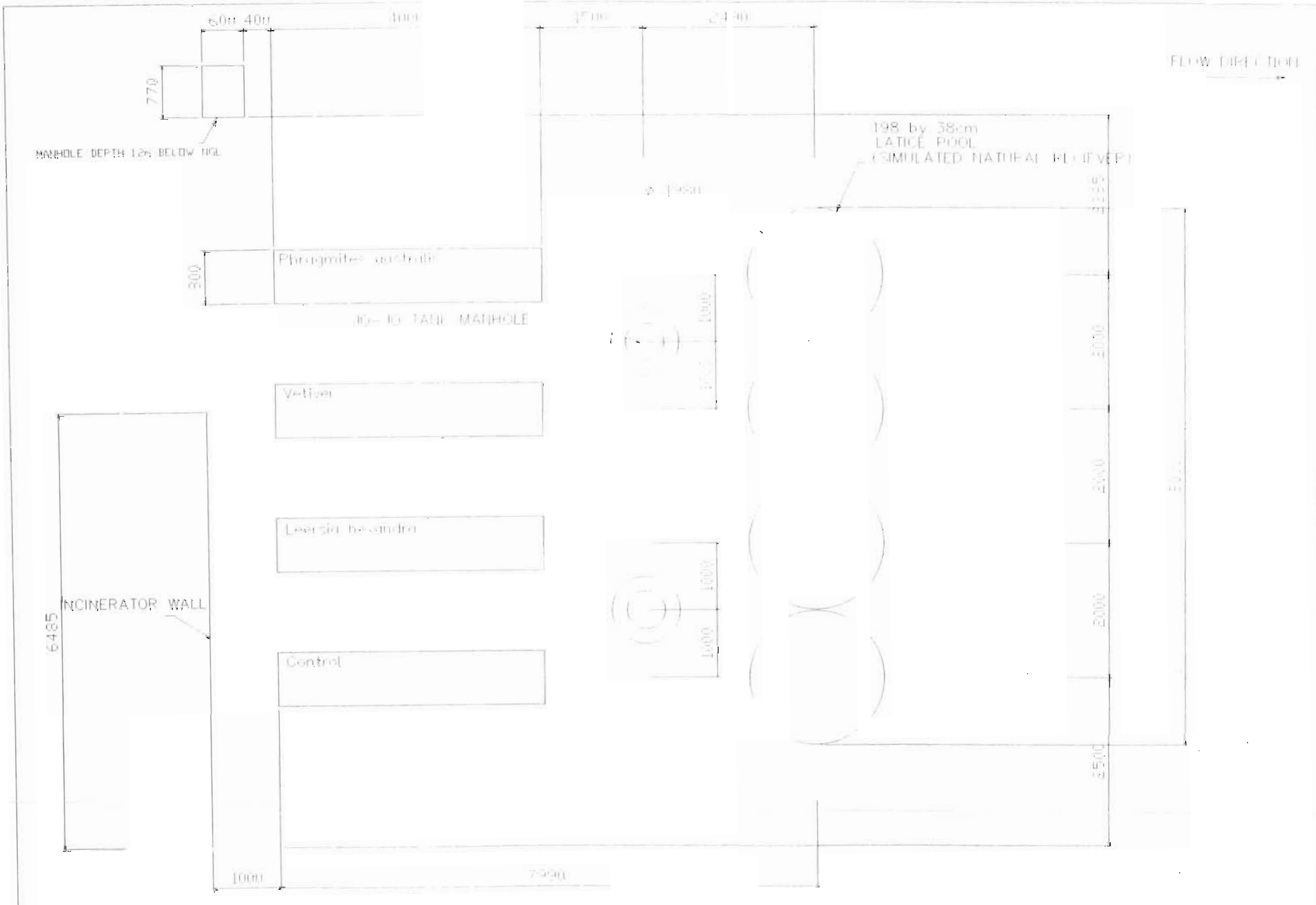


Figure 5.9: Pilot scale VSB system layout

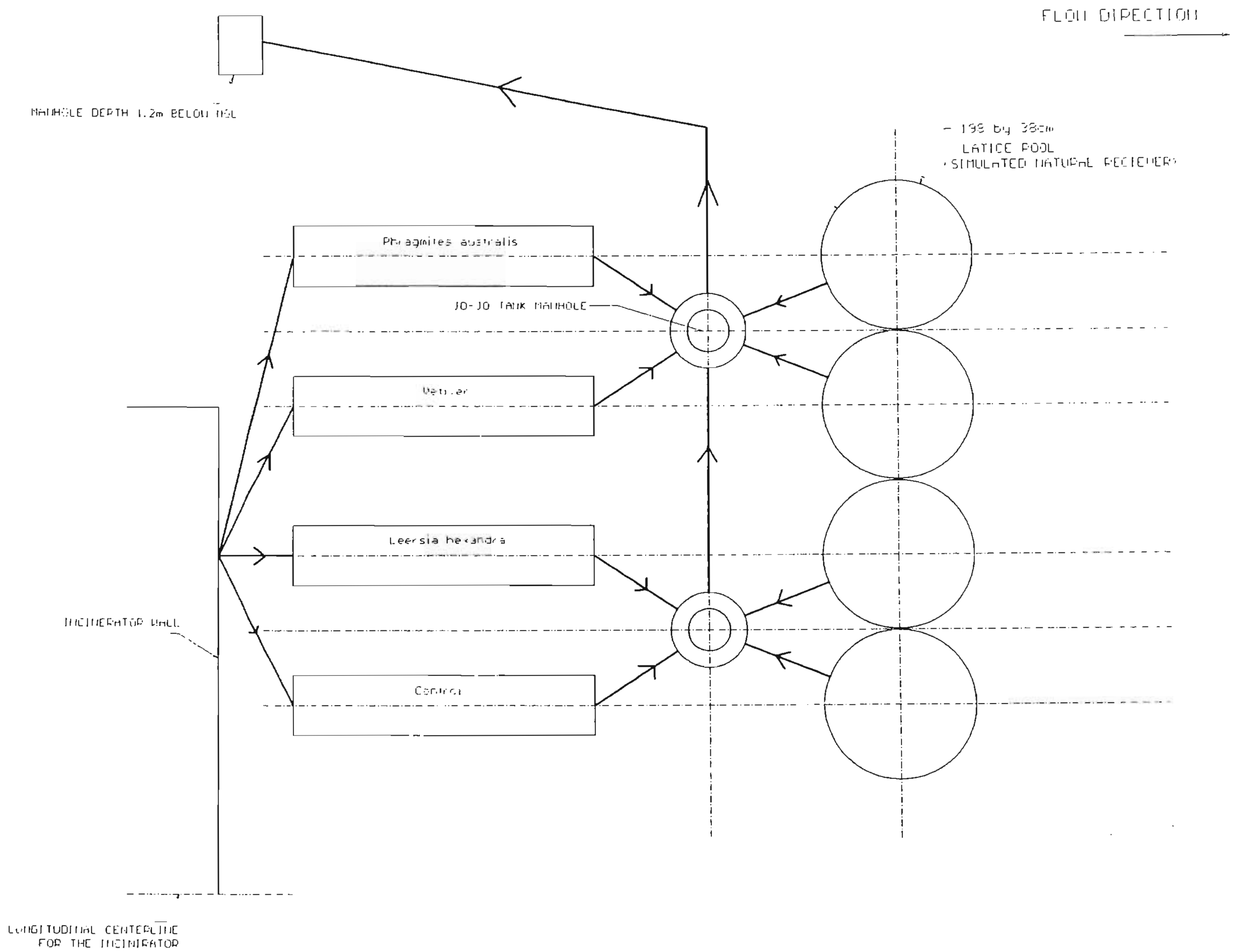


Figure 4.10: Pilot scale USB system pipe layout.

5.3 Construction, planting and establishment

The area selected for the pilot scale VSB CW was initially surveyed and the elements of the system were set out on the ground. Minor earth works and excavation were required; initially to find existing electrical cables and water pipes and then for wetland cells, manholes, lattice pools and pipe work (Plate 5.4). During excavation de-watering was required as a nearby water connection was faulty and had to be repaired before construction could proceed (Plate 5.5).

Plate 5.4: Setting out and excavation of the VSB CW.

Plate 5.5: Faulty water connection causing excavated areas to flood.

After the area had been prepared and excavated a 50 mm blinding layer was poured into the base of the excavated wetland cells to serve as a working platform for the construction of the brick 'shells'. The manholes and lattice pools were placed ready for pipe work and connections (Plate 5.6). A strip of high-density polyethylene (HDPE) was placed around the lattice pools to protect them from damage during grass cutting or by accidental scuffing (Plate 5.7). Each cell was lined with a prefabricated, 5 mm thick HDPE box which was lowered into the individual brick shells (Plate 5.8). Once the HDPE boxes were in place the outlet connections and pipe work

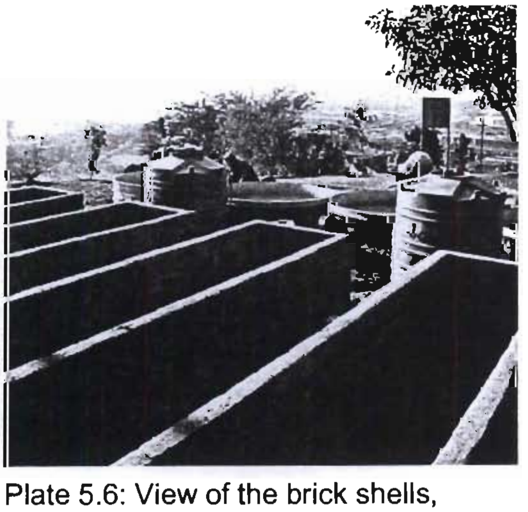


Plate 5.6: View of the brick shells, manholes and lattice pools.

was completed (Plates 5.9, 5.10, 5.11 and 5.12).

Plate 5.7: HDPE protective strip and outlet connection.

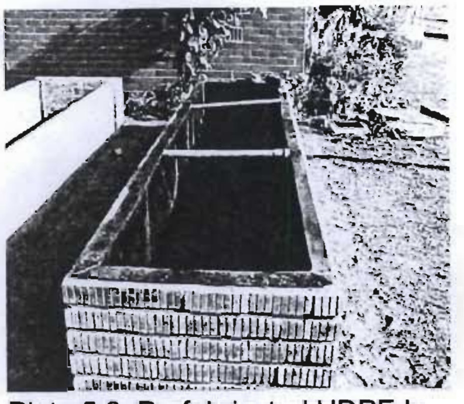


Plate 5.8: Prefabricated HDPE box placed in brick 'shell'.

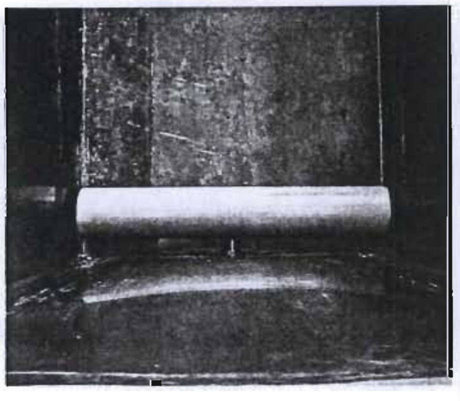


Plate 5.9: Outlet collector pipe.

Plate 5.10: Typical outlet connection

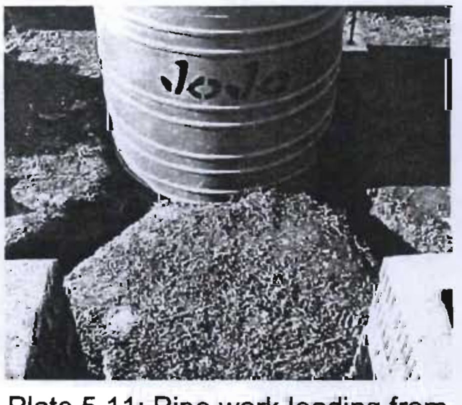


Plate 5.11: Pipe work leading from VSB CW to the manhole.

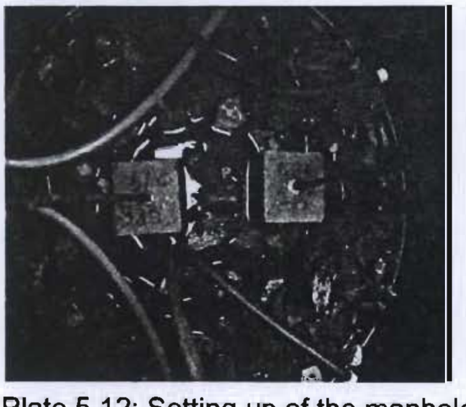


Plate 5.12: Setting up of the manhole and level control system.

Once the pipe work had been completed the VSB CW were filled with the rooting medium and the subsurface drain was simultaneously constructed (Plates 5.13 and 5.14). The 0.2 m, 13.2 mm stone layer was then placed on top of the rooting medium layer and subsurface drain.

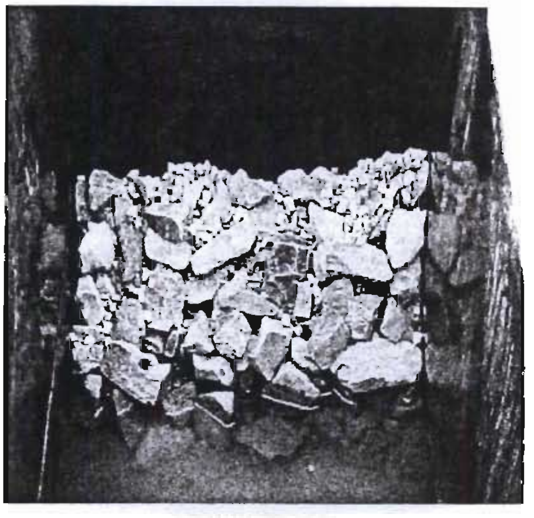


Plate 5.13: Hand placed stone around the outlet collection pipe.

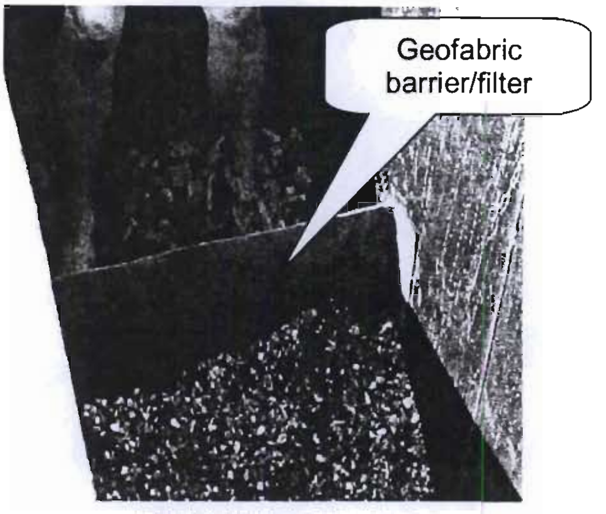


Plate 5.14: Placing the rooting medium and simultaneously constructing the subsurface drain.

Once the wetland cells had been completely filled with the perforated sample pipes in place they were filled to level (10 mm below the top surface) with spring water and the inlet systems were constructed and calibrated (Plates 5.15 and 5.16).

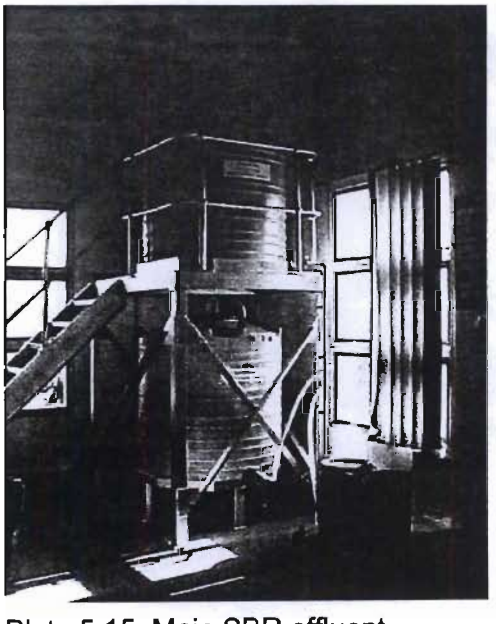


Plate 5.15: Main SBR effluent storage tanks.

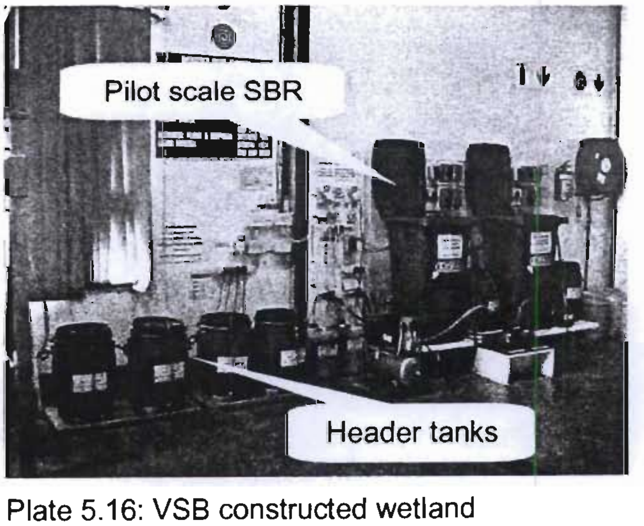


Plate 5.16: VSB constructed wetland header tanks.

Construction of the wetlands was completed by the 7th July 2000, after which the system was operated with spring water to simulate normal operating conditions and to find any initial problems that may arise.

Collection and planting of the vegetation species was done during early July (mid-winter). However, there was no effect on the establishment on the plants due to the small variations between seasons in the Durban area. The *Leersia hexandra* was collected from a nearby municipal pond. Sections of the plant, including rhizomes and above ground parts, were extracted and transported to the site for planting. The extracted sections were kept moist until planting to avoid drying out. The sections were planted directly into the upper stone layer using the rhizome method, at small spacings to aid in rapid establishment. The *Vetiver zizanioides* was extracted from the Bisasar Road Landfill site stability berm where it was being used to stabilise the bank slopes. Sections of the above ground parts including the top portion of the 'sod' were extracted and planted in tightly spaced hedges perpendicular to the direction of flow, directly into the upper stone layer. The *Phragmites australis* were collected from a viaduct situated near the Durban harbor and transported to the site where they were kept moist until planting. The rhizome sections were planted directly into the upper layer of stone at very small centers to aid in rapid establishment.

After planting and during vegetation establishment the wetlands were again operated using spring water. During this period there was rapid establishment of the plant species. However, an operational problem causing draw down of the water level within the wetland cells was encountered during days of heavy rainfall, this was found to be due to the overflowing of the effluent collection drums; the outlet pipes were placed into the drums and when the drums overflowed the pipe ends became submerged causing a siphoning effect between the wetland cell and effluent drum. Attaching half sections of uPVC pipe to the ends on each outlet hose and placing that section into the drum easily overcame this and allowed for the effluent entering the half section to remain at atmospheric pressure, thus eliminating the siphoning effect.

Feeding of the influent mix commenced, on the 11th of September 2000, after the vegetation had been established. Initially the effluent volumes were as expected. However, after a few weeks they dropped, for two of the cells, and it was expected that a leak had formed somewhere in the liner system or in one of the outlet connections. Inspection holes near the outlet connection were excavated; the connections were found to be working with no detectable leak so the inspection holes were left open over night, the following day it was found that the holes had formed shallow pools of water; samples of the water were analysed for chlorides and the results confirmed elevated concentrations, proving that a leak had formed somewhere in the liner system. The contents of the two cells were removed and the prefabricated HDPE boxes were taken out of the brick shells for inspection. After inspection it was found that fine cracks had formed along the fillet welds. The boxes were then sent for repairs. The cracks were believed to be caused by the expansion and contraction of the HDPE as well as the weight of the stone and water. Once the boxes were returned they were again placed into the brick shells, a further more flexible geomembrane protected by a geofabric was also placed inside the HDPE boxes to act as a further liner system. The contents of the cells were carefully replaced and reset for operation. Flow measurements taken after this incident confirmed that the leaks had been successfully fixed. Operation of the two unchanged cells ceased during repairs, however, they were not drained and contained mixed influent. The two repaired cells were drained and refilled with stream water. The fully established cells after repairs are presented in Plates 5.17, 5.18, 5.19 and 5.20.

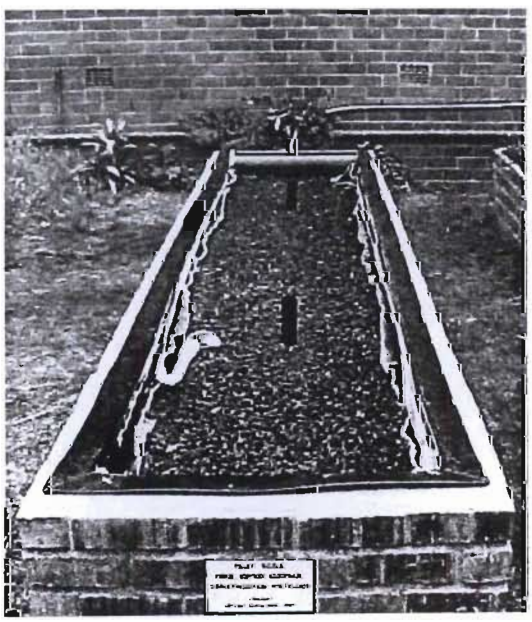


Plate 5.17: Control cell (Brown treatment line).

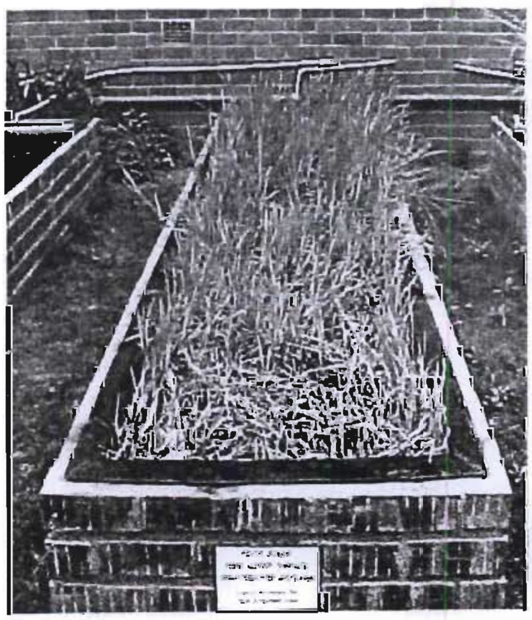


Plate 5.18: *Leersia hexandra* (Red treatment line).

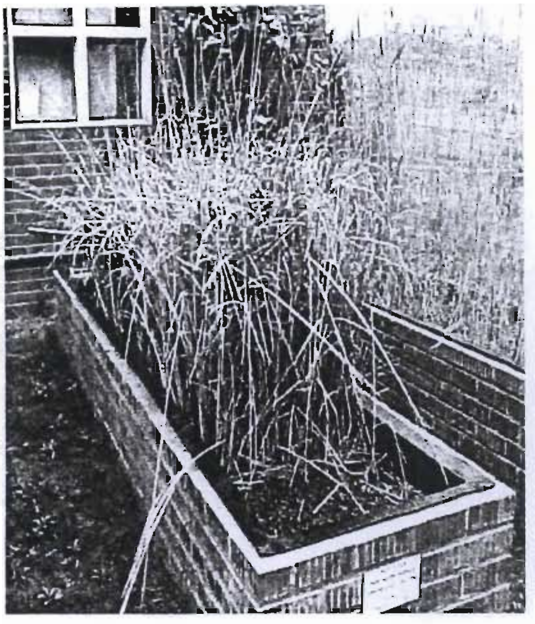


Plate 5.19: *Vetiver zizanioides* (Blue treatment line).

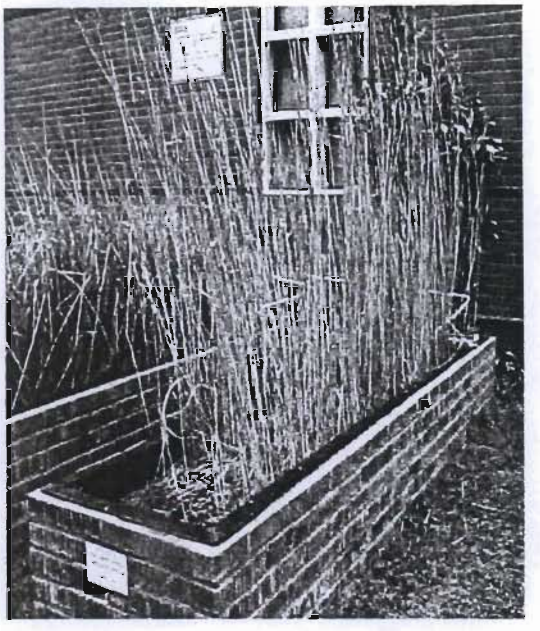


Plate 5.20: *Phragmites australis* (Orange treatment line).

The simulated natural receivers (Plate 5.21) were initially filled with spring water and planted with *Juncukrausii, Zandeschia aethiopica, Papyrus and Nymphae alba*. The fresh water fish species *Tilapia* and *Guppy* were also introduced.

Plate 5.21: Established simulated natural receivers.

CHAPTER 6

6 EXPERIMENTAL PROCEDURES

6.1 Operations and maintenance

The treatment trials commenced on the 4th of January 2001 after an initial acclimatisation period. Batch dosing of the influent mix took place every morning after mixing of the spring water and stored SBR effluent in the header tanks. The influent mix was then fed from the respective header tank, after a timed solenoid valve opened, into the inlet distribution system where it was evenly distributed across the width of the wetland cell (Plate 6.1). The daily effluent, which was collected in 25 litre drums in the manholes (Plate 6.2), was measured manually using a 2 litre measuring cylinder and poured into the respective simulated natural receiver, the effluent of which was not measured and was allowed to discharge into the nearby sewer line. The treatment lines were allocated a

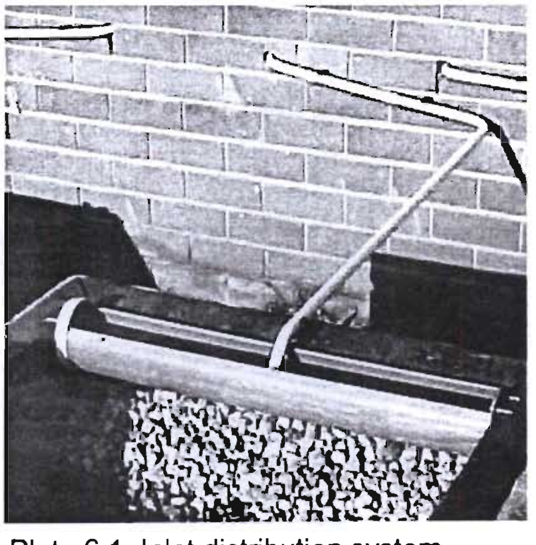


Plate 6.1: Inlet distribution system.

Plate 6.2: Effluent collection drums.

colour for simplicity; the control line was referred to as the Brown treatment line, the treatment line planted with *Leersia hexandra* was referred to as the Red treatment line, while the treatment line planted with *Vetiver zizanioides* was referred to as the Blue treatment line and the *Phragmites australis* planted treatment line was referred to as the Orange treatment line.

Inspection of the inlet system, vegetation, outlet hoses and simulated receivers was carried out on a daily bases in order to maintain the system in optimum operational conditions. The inlet system was found to be easy to maintain. However there were times when the inlet distribution pipes (Plate 6.1) had to be cleaned and the perforations at the bottom of the pipe along the long axis had to be reopened. It was found that this was not due to the influent but due to wind driven objects such as leaves, dust and seeds. After the outlet pipes had been fitted with the half section uPVC pipe sections they posed no other maintenance problem during the time of the treatability trials. The *Leersia hexandra* and *Vetiver zizanioides* were easily maintained. However, the *Phragmites australis* required a lot more attention. Starting at the beginning of spring in September and following through summer up to the end of April (end Autumn), the *Phragmites* was severely attacked by aphids, which destroyed the foliage and hindered the further growth of the *Phragmites,* this occurrence was possibly due to the scale of the system. In order to avoid additions of unwanted chemicals, which may have an effect on microbial growth, an 'organic cocktail' was mixed comprising of: 4 table spoons of salt and 3 teaspoons of crushed garlic allowed to soak in 25 liters of water over night, sieved and placed in spray bottles (Olufsen, 2001). The mixture was sparingly sprayed in a fine mist over the leaves of the *Phragmites;* It was found to work well in maintaining the population down to a level where the *Phragmites* could grow unrestricted by the aphids. However, it was not able to eliminate them completely and continual monitoring and spraying was required up to the end of autumn when the aphids disappeared. The *Phragmites* were also used by the local weaver birds for nest building. During spring the *Phragmites* leaves were totally stripped by the birds. This had a large effect on the evapotranspiration rates which sharply decreased during this period. No attempt to prevent the birds from stripping the leaves was considered and the effect which they had on the hydraulics of the system was probably due to the scale of the project. Although the *Leersia hexandra* did not pose any maintenance problems during the treatability trials it was found to encroach into the outlet and inlet zones. This was not seen to pose a problem for the outlet zone as the rhizomes of the plant would not be able to reach the outlet pipe and clog it. However, in the long term, it could have an affect on the inlet distribution system and clearing of the overgrown inlet zone would be required. Resuspension of the biological solids in the natural receivers did occur during the pouring of the wetland effluent into the ponds; this was overcome by hand placing large stones in the form of a stilling ledge over which the wetland effluent was poured. The outlet pipe of the ponds was occasionally blocked by organic solids and had to be cleared.

The overall system was found to be easily maintainable during the treatability trials and most of the maintenance and operational problems should be related to the scale of the system. During the site visits to the full-scale constructed wetlands it was noticed that the encroachment of vegetation into the inlet and outlet zones did pose a maintenance problem and clearing of vegetation from these zones was necessary (Chapter 4). Continual monitoring and maintenance of the inlet and outlet systems would also be required to maintain the system at optimal operational conditions.

6.2 Sampling and testing

6.2.1 Standard sampling points and testing

Initially the cells were filled to final level with spring water. Thus an increase in effluent concentrations was expected after initial feeding had started. Sampling during this period was done twice a week. After the initial period, sampling frequency was reduced to once a week and then to once every two weeks. Sampling of the spring water, raw leachate influent mix and simulated natural receivers was also carried out on a monthly basis. A summary of the samples taken and sampling frequency is presented in Table 6.1.

Table 6.1: Summary of sampling during the treatability trials.

Sample point	Comment
Raw leachate mix (Influent to SBR)	Once a month
Stored SBR effluent	Initially twice a week followed by once a week and then every 2 weeks
Spring water	Once a month
Constructed wetland influent mix	Initially twice a week followed by once a week and then every 2 weeks
Constructed wetland sample pipe 1 (1.3m from the inlet)	Initially twice a week followed by once a week and then every 2 weeks
Constructed wetland sample pipe 2 (2.6m from the inlet)	Initially twice a week followed by once a week and then every 2 weeks
Constructed wetland effluent	Initially twice a week followed by once a week and then every 2 weeks
Simulated natural receiver	Once a month and continual visual monitoring of the fish population and the growth of the aquatic vegetation

The analytical testing was done, at the School of Civil Engineering, Surveying and Construction of the University of Natal Durban (UND), in accordance to the *Standard Methods for the examination of water and wastewater* (Clesceri et al, 1989). The analytical tests carried out are presented in Table 6.2.

Table 6.2: Analytical tests carried out at the UND.

Parameter	Unit	Reference
pH	-	(Clesceri et al, 1989)
COD	mgO_2/l	(Clesceri et al, 1989)
BOD_5 and BOD_{20}	mgO_2/l	See Appendix C1
Chlorides	$mgCl^-/l$	(Clesceri et al, 1989)
Electrical conductivity	mS/m	(Clesceri et al, 1989)
NH_3-N	$mgNH_3$-N/l	(Clesceri et al, 1989)
Alkalinity	$mgCaCO_3/l$	(Clesceri et al, 1989)
TSS	mgTSS/l	(Clesceri et al, 1989)

6.2.2 Other sampling points and testing

The Durban Metro Water Services bacteriological laboratory carried out faecal and total coliform tests (Clesceri et al, 1989) as well as Microtox tests (Appendix C2). These tests were carried out once on the influent mixed raw leachate, twice on the stored SBR effluent, once on the constructed wetland mixed influent, once on the storm water outlet to the Umgeni River, twice up stream of the stormwater outlet and three times on the respective VSB constructed wetland effluents. Further analytical results for the sample points up and down stream of the stormwater outlet were obtained from Durban Metro Water Services.

6.2.3 Sampling methodology

The standard samples were collected in cleaned plastic sample bottles, which were sealed, clearly labeled and immediately taken to the University for testing, during which they were kept at 4°C, in the laboratory fridge, until used. The samples taken for microbiological and Microtox testing were collected in previously autoclaved glass bottles that were immediately sealed, clearly marked and placed on ice in a cooler box and transportation to the lab for testing. During sampling for the microbiological and Microtox tests all contact with the open sample bottle was kept at an absolute minimum to avoid sample contamination.

6.2.4 Presentation of the results

The wetland performance was analysed by reduction in pollutant concentrations and mass reduction. Concentration reduction is important for the requirements of meeting the South African general discharge limits and as a measure for potentially toxic materials; while, mass reduction may be regarded as important to the receiving environment. The two types of reduction may be described by equations 6.1 and 6.2 (Mulamoottil et al, 1998).

$$\% \text{ Concentration reduction} = 100(C_I - C_o)/C_I \qquad (6.1)$$

$$\% \text{ Mass reduction} = 100(Q_IC_I - Q_oC_o)/Q_IC_I \qquad (6.2)$$

6.2.5 Precision and Accuracy testing

6.2.5.1 Samples tested at the University of Natal Durban

Precision testing was carried out in two ways. Duplicate samples were done during testing and multiple analyses of a single sample were also conducted.

The standard deviations of the data from the duplicate samples were calculated using equation 6.3, taken from Clesceri et al (1989).

$$S_D = \frac{\overline{R}}{1.128} \qquad (6.3)$$

where

S_D = standard deviation for duplicates

and the average range $\overline{R}$ is:

$$\overline{R} = \frac{\sum |differences|}{n_o} \qquad (6.4)$$

where

n_o = number of observations.

The standard deviation of the data from the multiple analyses of a single sample was calculated using equation 6.5 (Roberson et al, 1995).

$$S_M = \sqrt{\frac{\sum_{Z=1}^{n}(C_Z - \overline{C})^2}{n_o - 1}} \qquad (6.5)$$

where

S_M = standard deviation for multiple analyses of a single sample

C_Z = concentration of sample z

and

$$\overline{C} = \frac{\sum_{Z=1}^{n} C_Z}{n_o} \qquad (6.6)$$

Accuracy testing was conducted for COD and chlorides. A standard COD and chloride solution of known concentration was prepared (Clesceri et al, 1989) and multiple analyses of the solutions were carried out.

COD

The results from the multiple analyses conducted on the synthetic solutions of known COD concentrations are presented in Table 6.3, while the results of the duplicate analyses conducted on the polluted water samples are presented in Table 6.4.

Table 6.3: Synthetic COD samples of known concentration.

Sample #	Synthetic Solution Batch 1	Synthetic Solution Batch 2
1	509	515
2	491	495
3	519	502
4	487	498
5	515	505
6	505	495
7	498	515
8	502	498
9	512	495
10	505	509
11	505	519
12	488	522
13	512	522
14	495	515

15	512	509
16	512	515
Actual COD	501	502
Mean	504	508
S_M	9.9	9.9
Maximum percentage deviation from actual COD	3.6	4.0

Note: Results are in mgO_2/l.

The results from the precision and accuracy testing on the synthetic COD samples showed very high precision and accuracy with a standard deviation below the specified 13 to 14 mgO_2/l given in by Clesceri et al (1989); which indicate a tight distribution about the means which are highly representative of the actual COD concentrations.

Table 6.4: Duplicate COD samples of polluted water.

Sample #	Sample 1	Sample 2
1	612	592
2	180	210
3	246	226
4	148	145
5	35	40
6	148	142
7	55	48
8	175	171
9	92	89
10	178	176
11	93	89
12	666	650
13	199	213
Average range	10.3	
S_D	9.1	

Note: Results are in mgO_2/l.

The precision tests conducted on the duplicate samples of polluted water showed very good precision with a standard deviation below that specified by the Clesceri et al (1989).

Chlorides

The results from the multiple analyses conducted on the synthetic solution of known chloride concentration (Clesceri et al, 1989) and on the polluted sample, as well as the results of the duplicate analyses conducted on the polluted samples are presented in Table 6.5 and 6.6 respectively.

Table 6.5: Multiple chloride analyses conducted on a synthetic sample and a polluted water sample.

Sample #	Synthetic sample	Polluted sample
1	490	1740
2	481	1702
3	481	1721
4	485	1740
5	490	1731
6	485	1750
7	481	1731
8	481	1731
9	481	1721
10	481	1721
11	481	1759
12	481	1731
13	481	1740
14	495	1711
15	481	1759
16	485	1731
Actual chlorides concentration	499	-
Mean	484	1733
S_M	4.4	15.7
Maximum percentage deviation from actual COD	3.6	-

Note: Results are in mgCl⁻/l.

The accuracy and precision tests conducted on the synthetic sample showed high precision with a relatively low standard deviation indicating a tight distribution about the mean, which was found to be a relatively good indication of the actual chloride concentration. The average bias (lower than the true value) was found to be 15 mgCl⁻/l. The multiple analyses conducted on the polluted water sample gave a higher standard deviation than that of the synthetic sample, which may arguably be due to interference from certain substances present in the polluted water. However, the standard deviation was still relatively low and within the order of accuracy for the treatability trials.

Table 6.6: Duplicate chloride samples of polluted water.

Sample #	Sample 1	Sample 2
1	1874	1874
2	815	705
3	835	835
4	655	645
5	245	255
6	745	735
7	345	355
8	1854	1824
9	655	655
10	715	685
11	415	405
12	645	655
13	355	385
Average range	20	
S_D	17.7	

Note: Results are in $mgCl^-/l$.

The standard deviation of the duplicate tests conducted on the polluted samples was found to correspond well with the standard deviation calculated for the multiple analyses conducted on the polluted sample.

Alkalinity

Results from the alkalinity precision tests conducted on polluted water samples are presented in Table 6.7 below.

Table 6.7: Multiple and duplicate alkalinity analyses of polluted water samples.

Sample #	Multiple analyses	Duplicate data set 1	Duplicate data set 2
1	1495	710	705
2	1490	775	775
3	1495	595	595
4	1490	310	315
5	1485	650	650
6	1485	370	375
7	1495	1575	1570
8	1490	670	675
9	-	625	625
10	-	435	435
11	-	760	770
12	-	485	485
Mean	1491	-	
Average range	-	2.9	

S_D	-	2.6
S_M	4.2	-

Note: 1. Results are in $mgCaCO_3/l$.
2. End point = pH 4.3.

The alkalinity tests showed very high precision; standard deviations of 1 $mgCaCO_3/l$ have been achieved when the alkalinity is due entirely to carbonates or bicarbonates, however, other laboratories have obtained standard deviations ranging from 5 to 40 $mgCaCO_3/l$ (Clesceri et al, 1989).

pH

Results from the pH-value precision tests conducted on polluted water samples are presented in Table 6.8. They show very high precision with a standard deviation below 0.13 which is specified by Clesceri et al (1989).

Table 6.8: Duplicate pH analyses on the polluted water samples.

Sample #	Duplicate data set 1	Duplicate data set 2
1	8.51	8.51
2	8.56	8.54
3	8.53	8.53
4	8.65	8.69
5	8.42	8.40
6	8.30	8.30
7	8.37	8.36
8	8.45	8.45
9	8.42	8.42
10	8.71	8.71
11	8.62	8.61
12	8.25	8.20
13	8.55	8.52
Average range	0.014	
S_D	0.012	

Note: Results are in pH units.

TSS

Results from the duplicate TSS analyses conducted on the polluted water samples are presented in Table 6.9.

Table 6.9: Duplicate TSS analyses conducted on the polluted water samples.

Sample #	Duplicate data set 1	Duplicate data set 2
1	78	80
2	43	36
3	44	47
4	47	64
5	45	39
6	16	21
7	11	14
8	14	8
9	12	14
10	16	23
11	15	25
Average range	6.2	
S_D	5.5	

Note: Results are in mgTSS/l.

The precision of the duplicate TSS tests was found to be within the range for the standard deviation, of 5.2 to 24 mgTSS/l for TSS concentrations of 15 to 242 mgTSS/l respectively (Clesceri et al, 1989).

BOD_5 and BOD_{20}

Results from the duplicate Oxy-top BOD_5 and BOD_{20} analyses conducted on the polluted water samples are presented in Table 6.10.

Table 6.10: Duplicate BOD_5 and BOD_{20} analyses conducted on the polluted water samples.

Sample #	Duplicate data set 1 (BOD_5)	Duplicate data set 2 (BOD_5)	Duplicate data set 1 (BOD_{20})	Duplicate data set 2 (BOD_{20})
1	13.6	13.3	27.8	28.6
2	5.3	5.3	12.1	12.1
3	1.8	1.2	4.4	4.1
4	2.4	3.8	7.1	8.9
5	2.4	1.8	4.1	4.1
6	2.7	2.7	7.1	8
7	3.5	4.4	20.1	17.1
8	0	0.6	12.7	8.9
9	6.5	5.3	2.4	3.2
10	4.1	2.7	3.8	3.2
11	0.3	0.3	6.2	6.8
12			0	0.9
13			12.7	13.6

14			0.3	1.8
15			1.8	1.2
16			4.7	5.9
17			1.5	1.2
Average range	0.54		0.61	
S_D	0.48		0.54	

Note: Results are in mgO_2/l.

Precession and bias tests conducted in a series of interlaboratory studies found no measurement for establishing bias of the BOD procedure and extreme variability in the test results. It was recommended that a control limit, for BOD_5, of 1 standard deviation be used for individual laboratories (Clesceri et al, 1989). Both the standard deviations calculated from the duplicate analyses for BOD_5 and BOD_{20} were below the control limit.

NH_3-N

Results from the multiple and duplicate NH_3-N analyses conducted on the polluted water are presented in Table 6.11.

Table 6.11: Multiple and duplicate NH_3-N analyses conducted on the polluted water samples.

Sample #	Multiple analyses data set 1	Multiple analyses data set 2	Duplicate data set 1	Duplicate data set 2
1	560	377	1.78	2.11
2	558	370	0.76	0.73
3	558	372	0.49	0.50
4	548	372	0.71	0.78
5	552	367	0.20	0.20
6	547	367	0.46	0.51
7	548	382	0.21	0.17
8	560	372	1.11	1.28
9	-	-	0.43	0.43
10	-	-	0.29	0.24
11	-	-	0.47	0.46
12	-	-	0.24	0.26
Mean	554	372	-	
Average range	-	-	0.07	
S_D	-	-	0.06	
S_M	5.7	5.0	-	

Note: Results are in $mgNH_3$-N/l.

A large discrepancy was found between the duplicate standard deviation and the multiple analyses standard deviation. This is due to the difference in order of magnitude between the two data sets. The relative standard deviations for the multiple analyses were found to be far below the specified 21.6% (Clesceri et al, 1989).

6.2.5.2 Samples tested at Durban Metro Water Services

The Total and Faecal coliform and Microtox tests were done at the accredited Durban Metro Water Services laboratory.

CHAPTER 7

7 RESULTS OF THE PILOT SCALE VSB CW TREATABILITY TRIALS

7.1 Hydraulic balance

Precipitation experienced at Bisasar Road Landfill site during the first eight months of the year 2001, calculated by averaging the daily rainfall measurements obtained from the Northern municipal wastewater treatment works and the Botanical Gardens weather station, is compared with the average monthly totals for the Durban area in Figure 7.1 (Raw data presented in Table D1, Appendix D1).

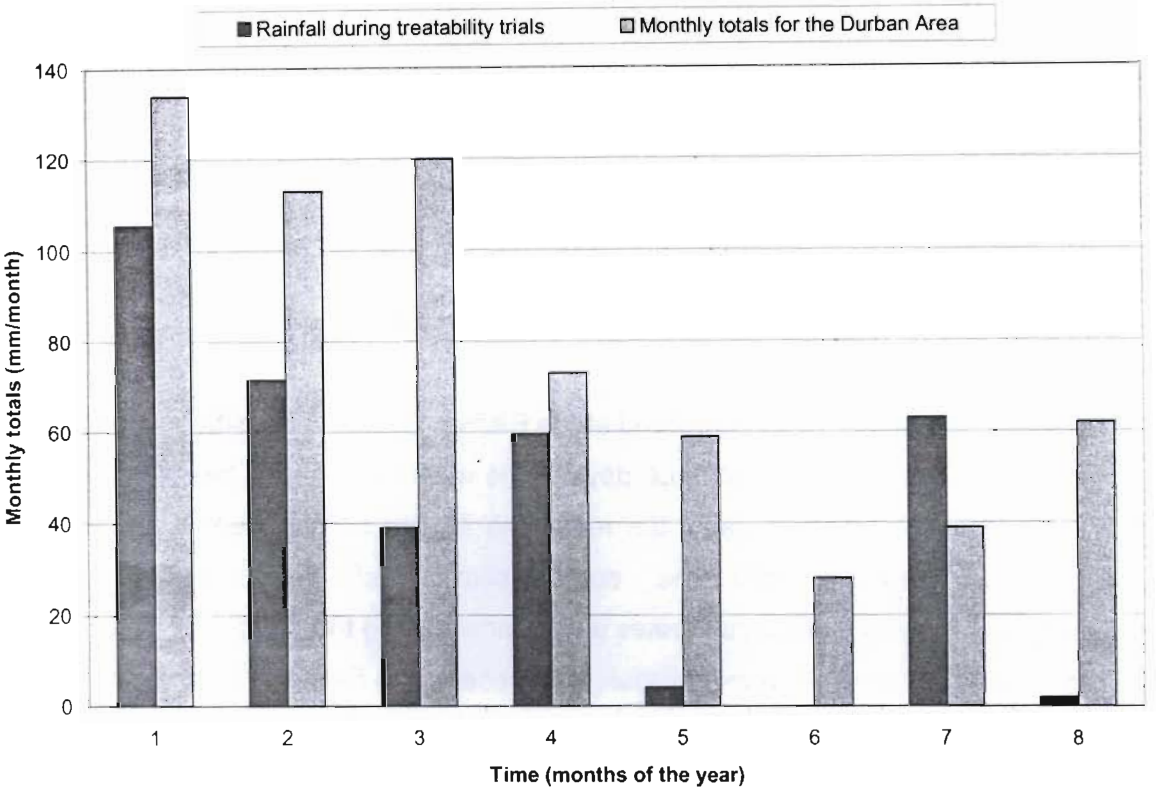


Figure 7.1: Monthly total precipitation experienced at Bisasar Road Landfill site.

The monthly precipitation experienced at the Bisasar Road Landfill site was generally found to be below the monthly averages for the Durban area. It has already been established that when taking monthly averages for precipitation and evapotranspiration into account the Durban area is water deficient. The findings presented in Figure 7.1 showed that concentrating effects, on the pollutants, from evapotranspiration would have a large influence on the effluent concentration due to the low HLR used for the treatment trials. The daily rainfall experienced during the treatability trials (starting 4 January 2001) is presented in Figure 7.2 (Raw data presented in Appendix D2).

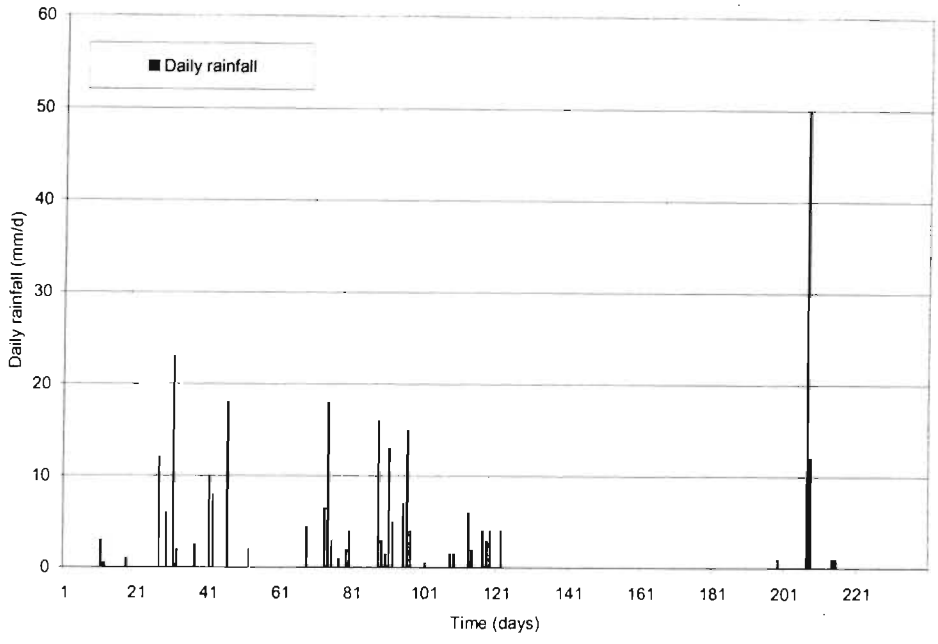


Figure 7.2: Daily rainfall experienced during the treatability trials.

The majority of the rain experienced at the Bisasar Road Landfill site during January 2001 (105mm) fell during the first three days of the month before the treatability trials started. Thus Figure 7.2 only presents the remaining 4.5mm experienced during the remaining days of January after the commencement of the trials. The measured evaporation/evapotranspiration rates experienced during the treatability trials, compared to the average Class-A-Pan evaporation, are presented in Figure 7.3 (Raw data presented in Appendix D2). Initially at the commencement of the trials the evapotranspiration rates for the four cells were similar with an average of 3 mm/day; as the trials continued the rates for the respective cells changed, due to seasonal variations and arguably due to further vegetation growth and establishment. The control cell (Brown treatment cell) showed a stable seasonal variation for the evaporation rates, with a linear relationship to the Class-A-Pan evaporation (Equation 7.1). A slightly less stable variation for the ET rates was experienced for the *Leersia hexandra* (Red treatment cell) and *Vetiver zizanioides* (Blue treatment cell); both cells showed higher losses than that of the control due to the transpiration component of the ET, however, the relationships between the Class-A-Pan evaporation was not as strongly linear as that of the control (Equations 7.2 and 7.3). The *Leersia hexandra* did not show any seasonal variations in growth and the foliage remained green through out treatability trials. The *Vetiver zizanioides* did, however, show signs of seasonal growth variations, which is indicated in Figure 7.3. The ET rates for the *Vetiver* decreased below the rates for the *Leersia*, starting in autumn and prolonging throughout the winter.

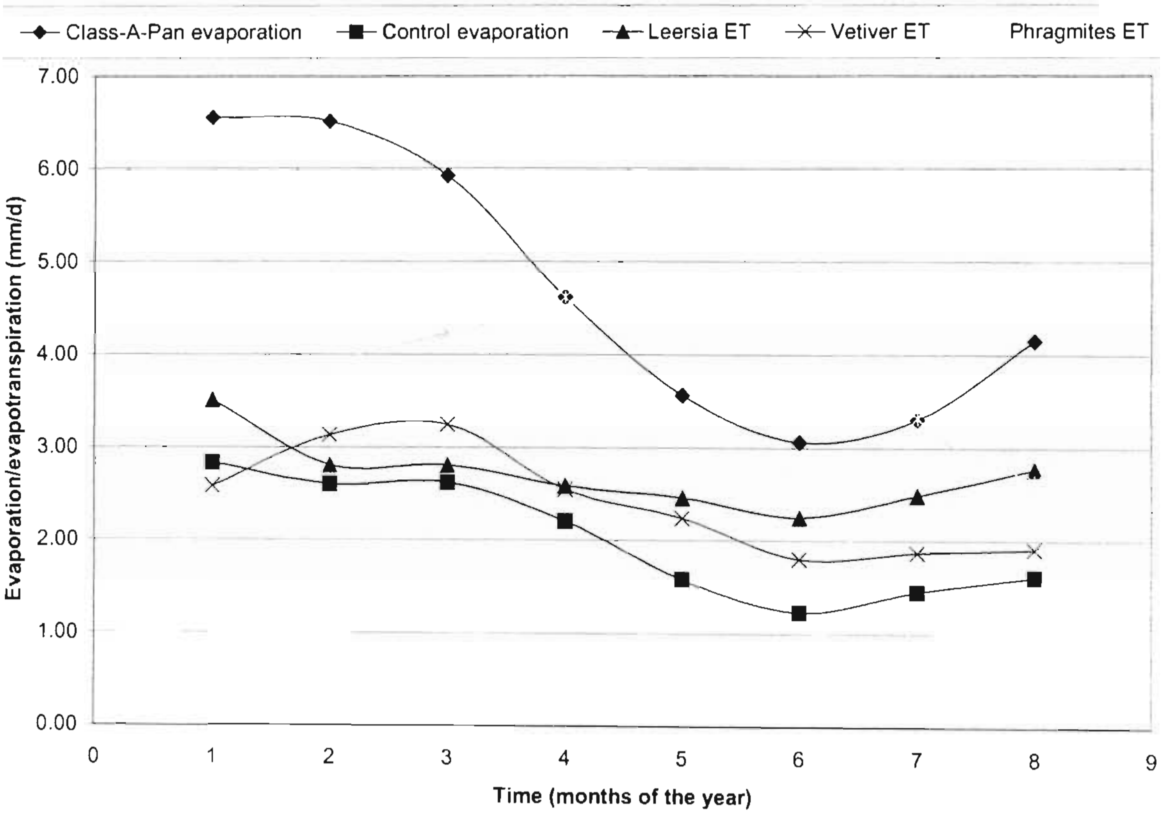


Figure 7.3: Evapotranspiration rates experienced during the treatability trials.

$$Evaporation = 0.4221(Class - A - Pan) - 0.0275 \quad (7.1)$$

$R^2 = 0.95$

Gravel/no plants

$$ET = 0.3369(Class - A - Pan) + 0.8291 \quad (7.2)$$

$R^2 = 0.74$

Gravel/*Vetiver*

$$ET = 0.2155(Class - A - Pan) + 1.6958 \quad (7.3)$$

$R^2 = 0.67$

Gravel/*Leersia*

The *Phragmites australis* showed erratic variations with no correlation to seasonal effects on ET. The gradual increase experienced during the first four months was probably due to the further establishment required for the vegetation, which stabilised during months 5 and 6. The *Phragmites* was stripped of all its foliage by weaverbirds during month 7, which is indicative of the sharp decrease in ET rates experienced in the following months. When evaluating the results from months 5 and 6 it was noted that the ET rates were higher than those estimated by Bavor et al. (1988) for *Cattails* and *Bulrushes*. The ET rates for the *Phragmites* were calculated to be about 1.7 times the Class-A-Pan evaporation.

From the results of the treatability trials, using equations 7.1 to 7.3 as well as 1.7 times the Class-A-Pan evaporation for the *Phragmites* ET rates, Figure 7.4 was constructed for the estimation of the potential ET rates experienced in the Durban area for the respective vegetation species (Raw data presented in Appendix D2). Due to the scale of the system and the low HLR used for the treatability trials there was a net concentrating effect on the system's pollutants.

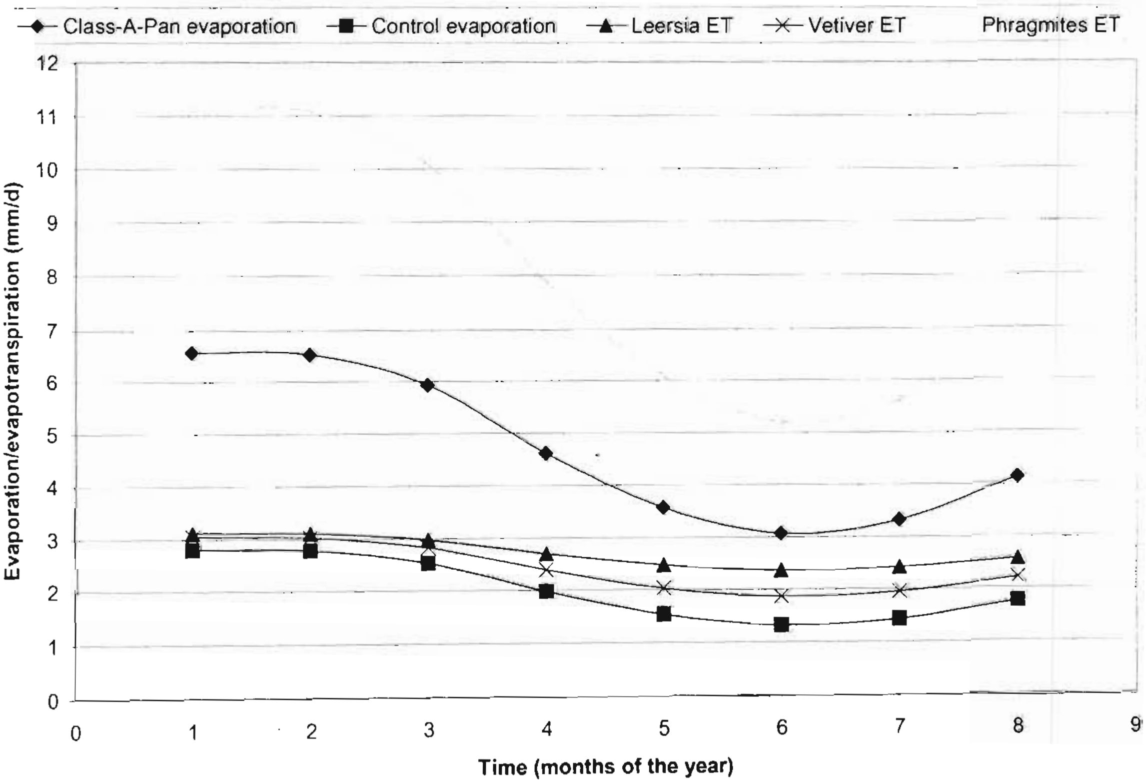


Figure 7.4: Potential evaporation/evapotranspiration rates for the respective vegetation species in the Durban area.

7.2 Characteristics of the influent mix

During the design of the pilot scale VSB CW the characteristics of the influent mix were estimated using mass balance calculations (Table 5.6). The actual characteristics, which were monitored during the treatability trials, are presented in Table 7.1 (Raw data presented in Table D7, appendix D3).

Table 7.1: Characteristics of the actual mixed pilot scale VSB CW influent.

Parameter	Influent mix (From Table 5.6)	# of Samples	Actual (mean)	Standard deviation
pH	-	24	8.5	0.16
COD	360	24	271	24.9
BOD_5	-	8	7.1	3.5
BOD_{20}	-	8	15.8	5.9
NH_4-N	0.82	6	0.85	0.14
Alkalinity (as $CaCO_3$)	773	24	776	70.3
Chlorides	998	24	1017	45.9
Conductivity (mS/m)	416	15	421	9.4
TSS	50	6	40	28.2

Note: 1. Results in mg/l, except pH and conductivity.

7.3 The use of chlorides as a mass balance tracer

In order to establish a point of stabilisation in the VSB CW, chlorides were used as a tracer. The use of chlorides as a tracer was selected, as it is present in the leachate and the influent mix at relatively high concentrations and the introduction of another tracer compound into the influent was not required.

The results of the tracer analyses are expressed in mass of chlorides into and out of the system, calculated using equation 7.4 (Kadlec and Knight, 1996).

$$C_I \cdot Q_i + C_P \cdot Q_R = C_O \cdot Q_O \qquad (7.4)$$

where:

Q_R = Flow rate of rainfall through the system, m^3/d

The rainfall term was regarded as negligible compared to the influent and effluent mass and the equation was simplified to (equation 7.5).

$$C_I \cdot Q_i = C_O \cdot Q_O \qquad (7.5)$$

Theoretically, the system should have reached stabilisation when the influent mass of chlorides was equal to the effluent mass, however, chloride is not a highly conservative tracer and may be biologically and chemically transformed during its passage through the bed, resulting in a slightly lower mass in the effluent than in the influent at stabilisation.

The results from the chloride tracer analyses for the Control, *Leersia hexandra, Vetiver zizanioides* and *Phragmites australis* are presented in Figures 7.5, 7.6, 7.7 and 7.8 respectively (Raw data presented in Appendix D2 and D3).

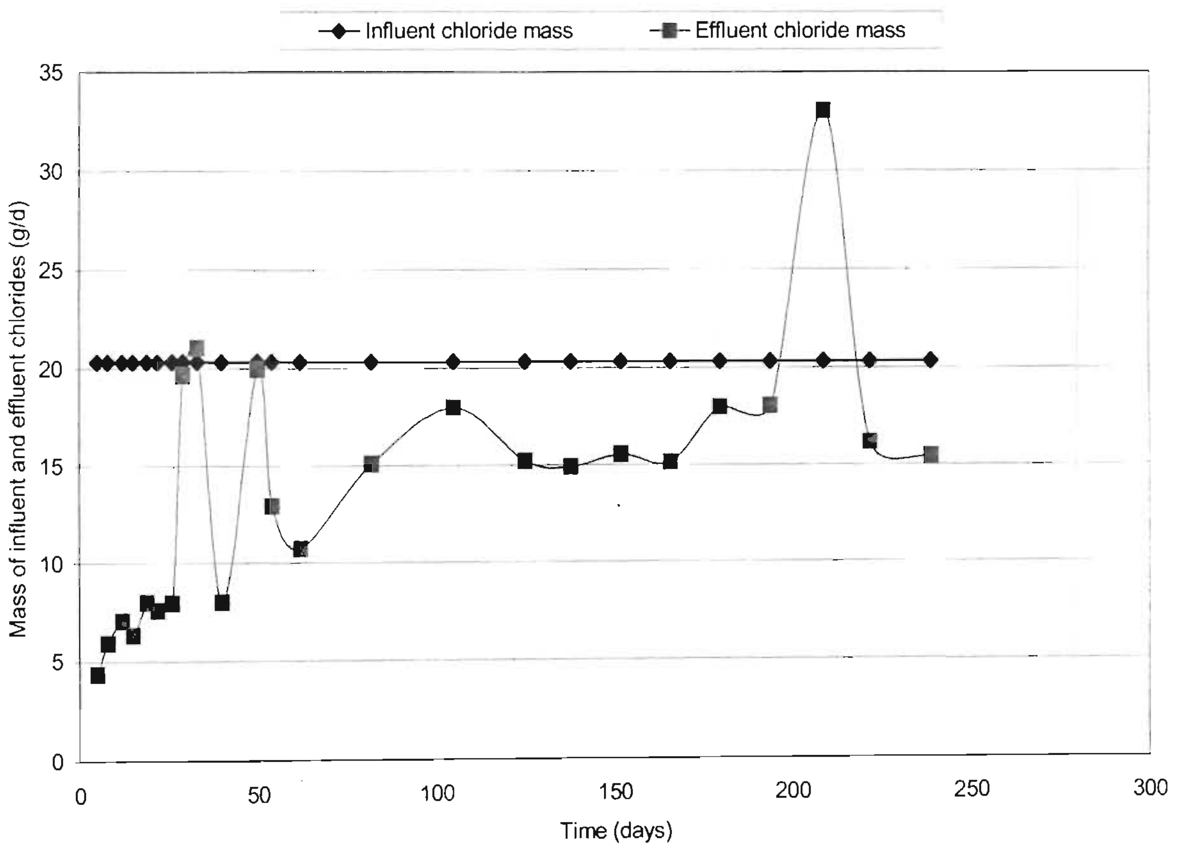


Figure 7.5: Chloride tracer analyses for the Control VSB.

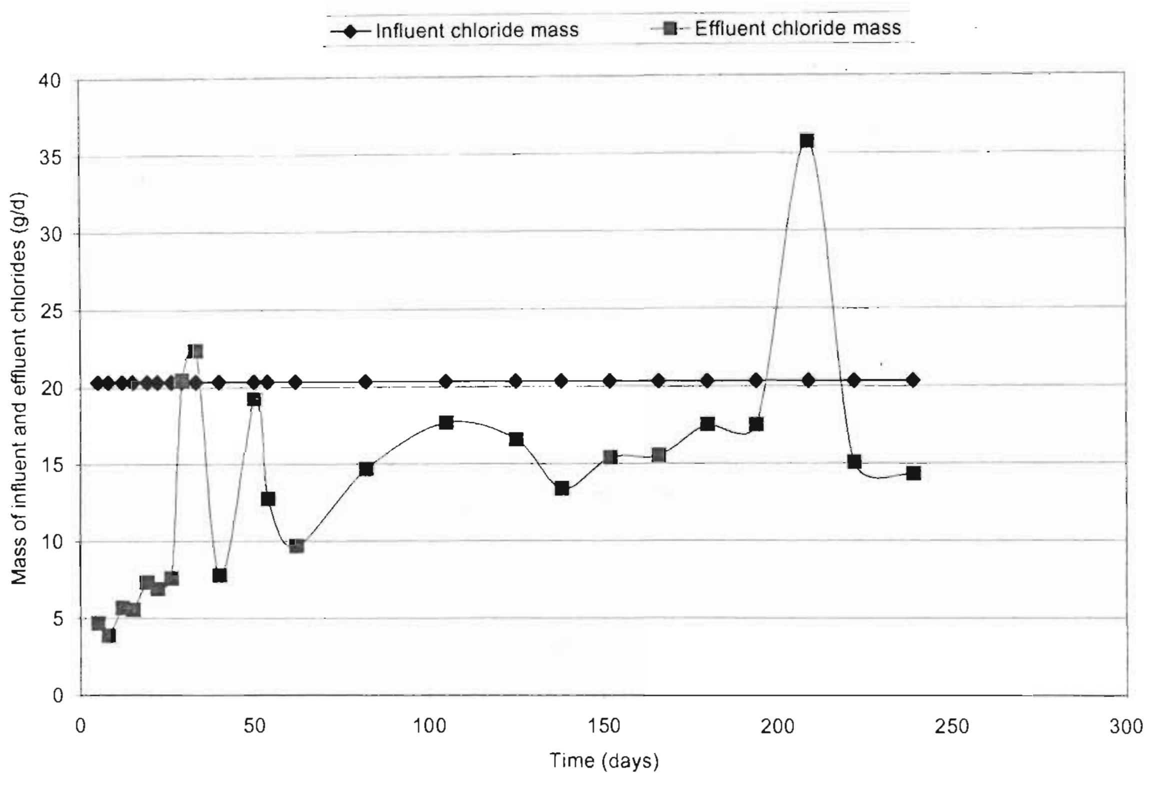


Figure 7.6: Chloride tracer analyses for the *Leersia hexandra* VSB.

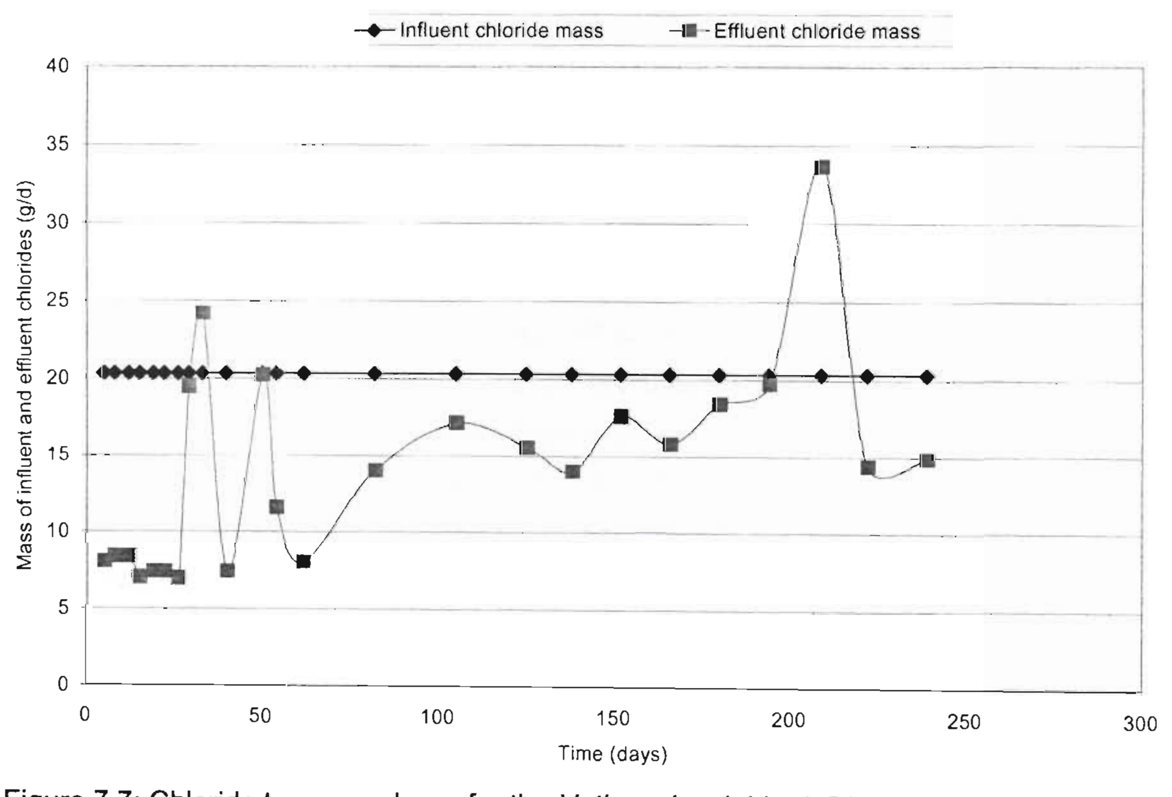


Figure 7.7: Chloride tracer analyses for the *Vetiver zizanioides* VSB.

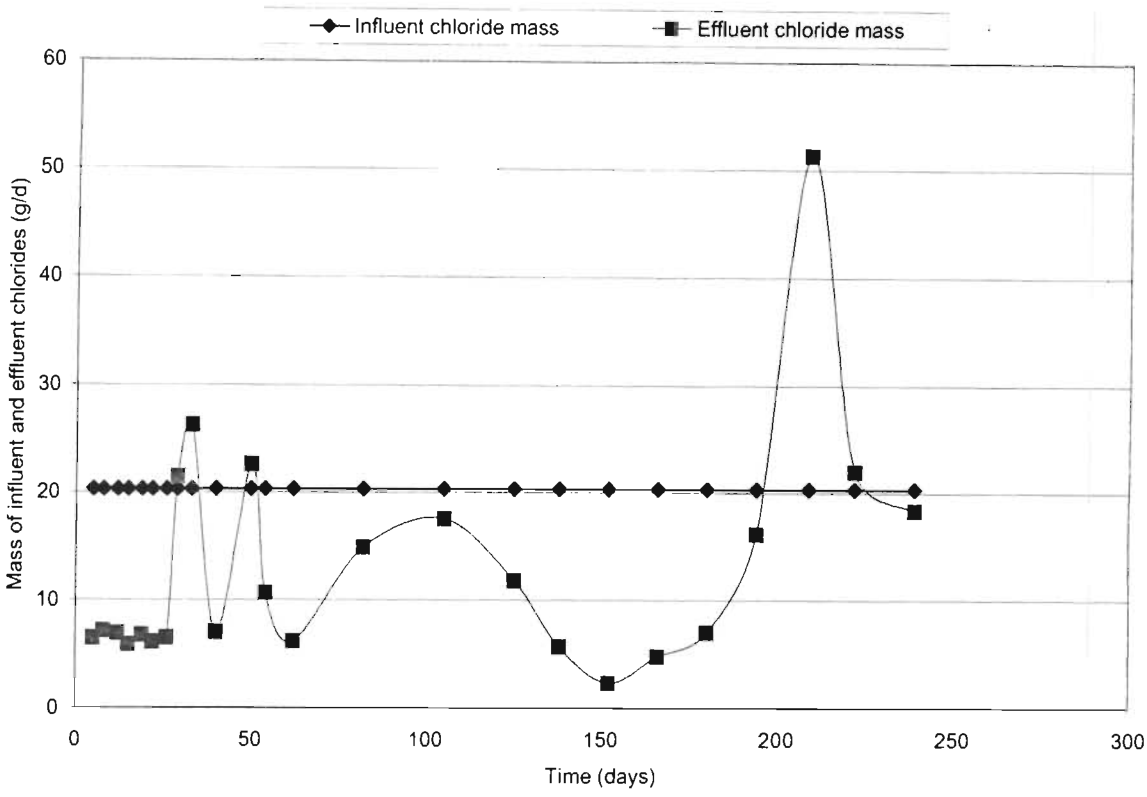


Figure 7.8: Chloride tracer analyses for the *Phragmites australis* VSB.

From the results it was noted that during events of heavy rainfall (Figure 7.2) the mass of chlorides in the effluent was occasionally higher than the influent mass. This was arguably due to the fact that the VSB cells were not completely mixed reactors where instantaneous mixing of the rainfall and bed volume would occur; instead, the pulse of rain caused the outflow of the bed volume to occur before mixing could take place. The effluent volumes were higher during these periods but the chloride concentration had not been diluted yet, hence, the large mass of chloride in the effluent. In order to asses the long term stability of the system these localised events were removed from the data sets and the Figures were reconstructed and presented in 7.9 to 7.12 respectively. A best-fit line was constructed through the data points to estimate the time of stabilisation and the chloride mass in the effluent at stabilisation. Five out of the twenty-four data points were eliminated.

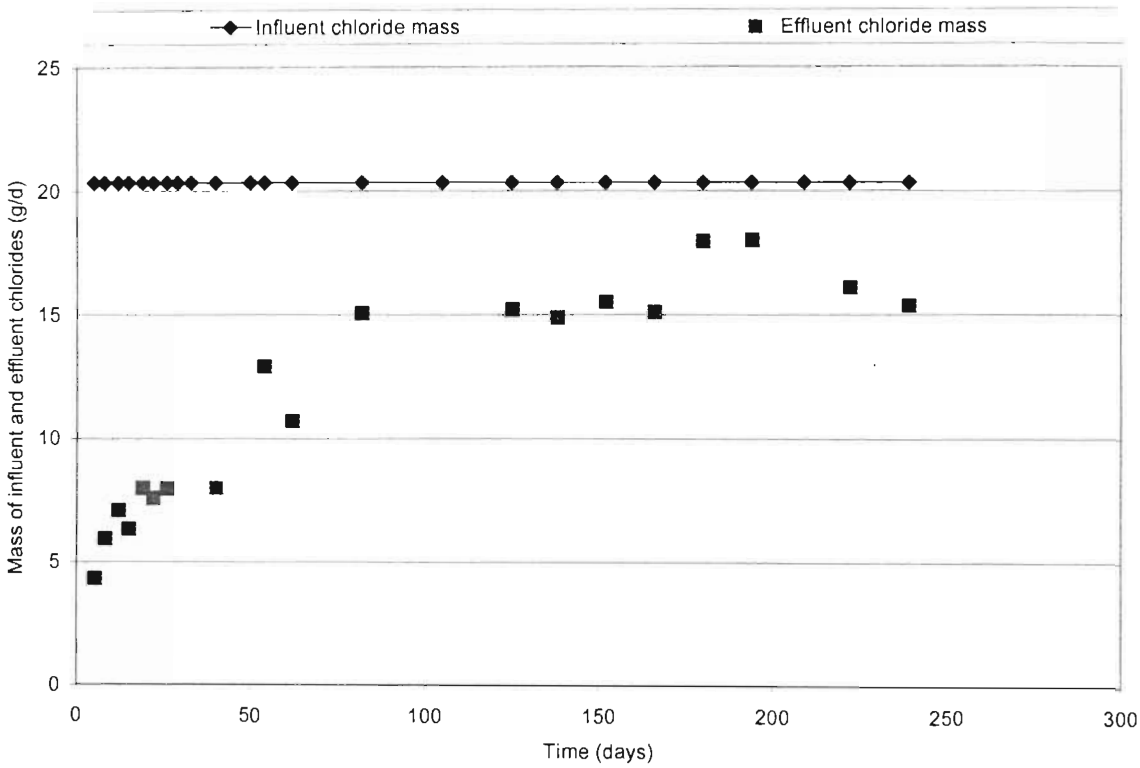


Figure 7.9: Modified chloride tracer analyses for the Control VSB.

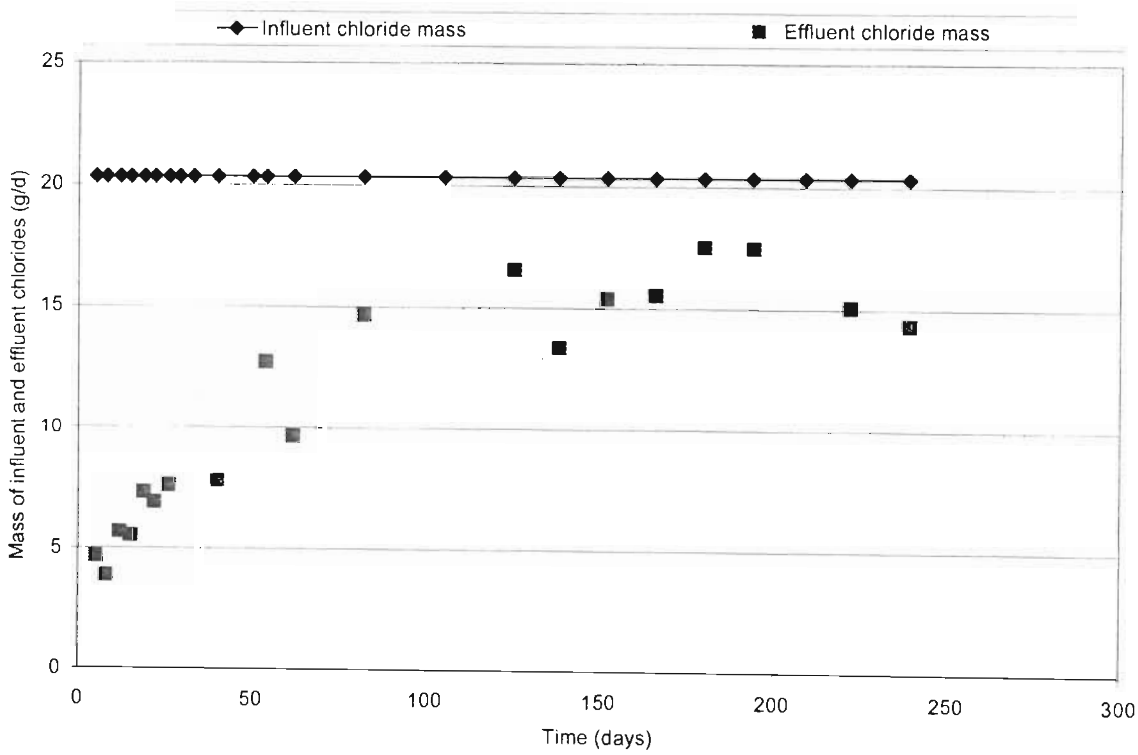


Figure 7.10: Modified chloride tracer analyses for the *Leersia hexandra* VSB.

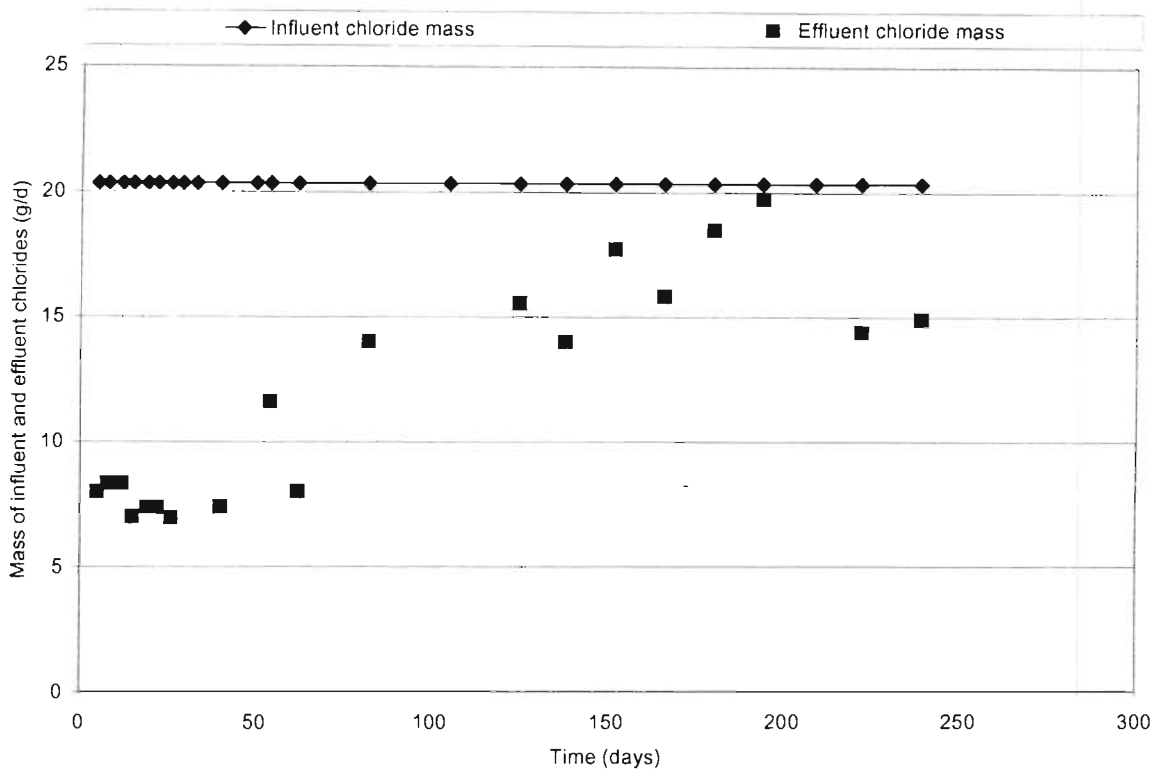


Figure 7.11: Modified chloride tracer analyses for the *Vetiver zizanioides* VSB.

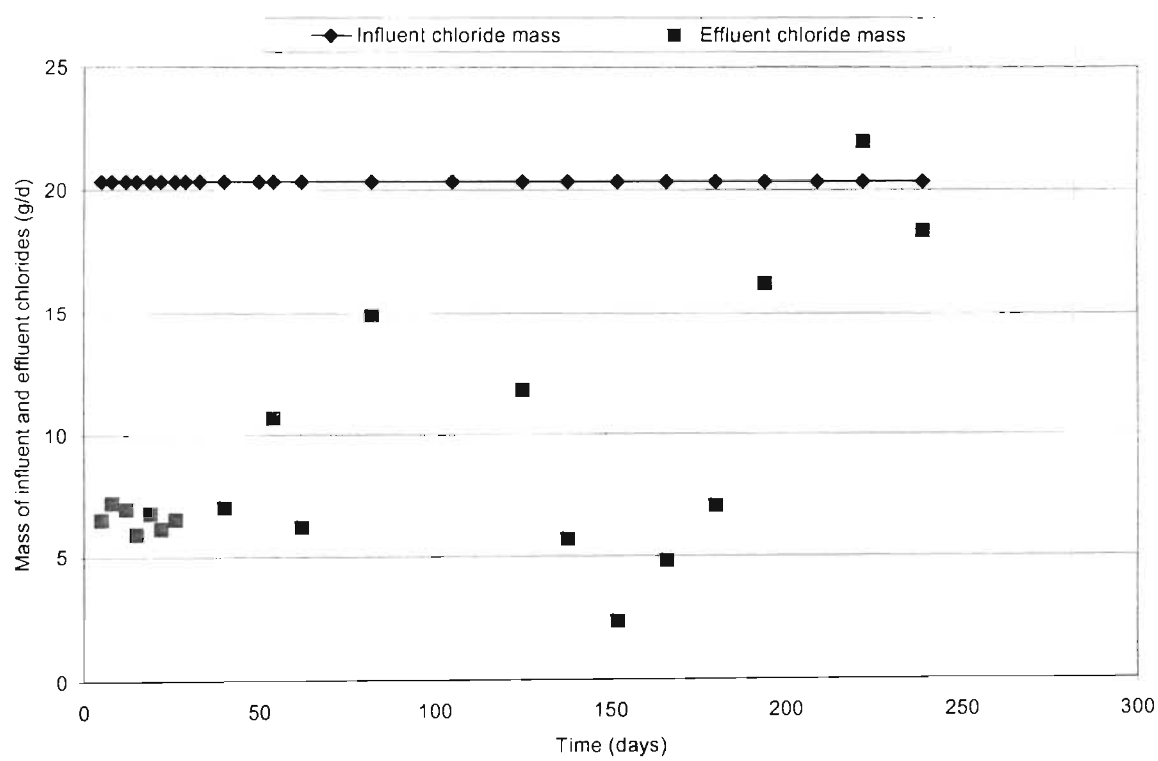


Figure 7.12: Modified chlorides tracer analyses for the *Phragmites australis* VSB.

The results from the modified tracer analyses conducted on the Control, *Leersia hexandra* and *Vetiver zizanioides* are presented in Table 7.2:

Table 7.2: Results from the modified chloride tracer analyses.

VSB	Estimated day of stabilisation (days)	Estimate effluent chloride mass at stabilisation (g/d)
Control	163	16
Leersia hexandra	184	17
Vetiver zizanioides	184	16

It was not possible to estimate the point of stabilisation for the *Phragmites australis* VSB due to the continual variations in the effluent volumes, initially from further establishment of the vegetation and due to the weaverbird incident. The time taken for stabilisation to occur was found to be much longer than expected. With a design HRT of 10.6 days for the treatment zone (Table 5.7), it required approximately 15 to 17 bed volumes to reach stabilisation. If the over all volume of the bed was considered, including the rooting medium at a porosity of 40% (Figure 3.9), the theoretical HRT was calculated to be 60 days, resulting in 2.7 to 3 bed volumes to reach stabilisation (Robinson, 2001b).

7.4 Effluent concentrations and system dynamics

7.4.1 Chlorides

Chloride concentrations were also used to establish the dilution and concentrating effects of rainfall and ET respectively and to understand the dynamics of the system. The results of the chloride concentrations at sample pipes along the VSB cells and in the respective effluents during the treatability trials are presented in Figures 7.13 to 7.16 (Raw data presented in Appendix D3). After analysing the effluent flows which were measured during the treatability trials it was found that a concentrating effect from ET would be the predominant long-term influence, with deviations from this effect during localised rainfall periods. The ET effects on the effluent volumes were also found to be seasonal, indicating that the overall concentrations of the effluent would vary according to the seasonal effects on ET.

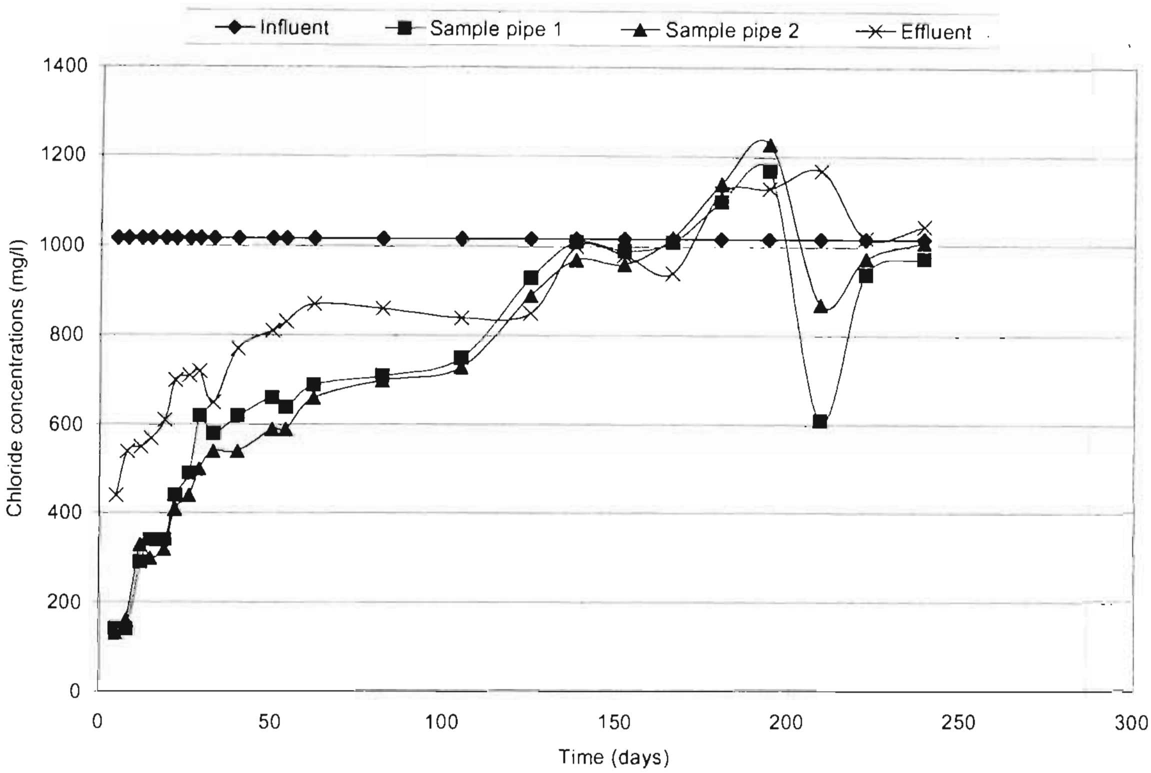


Figure 7.13: Chloride concentrations for the Control VSB.

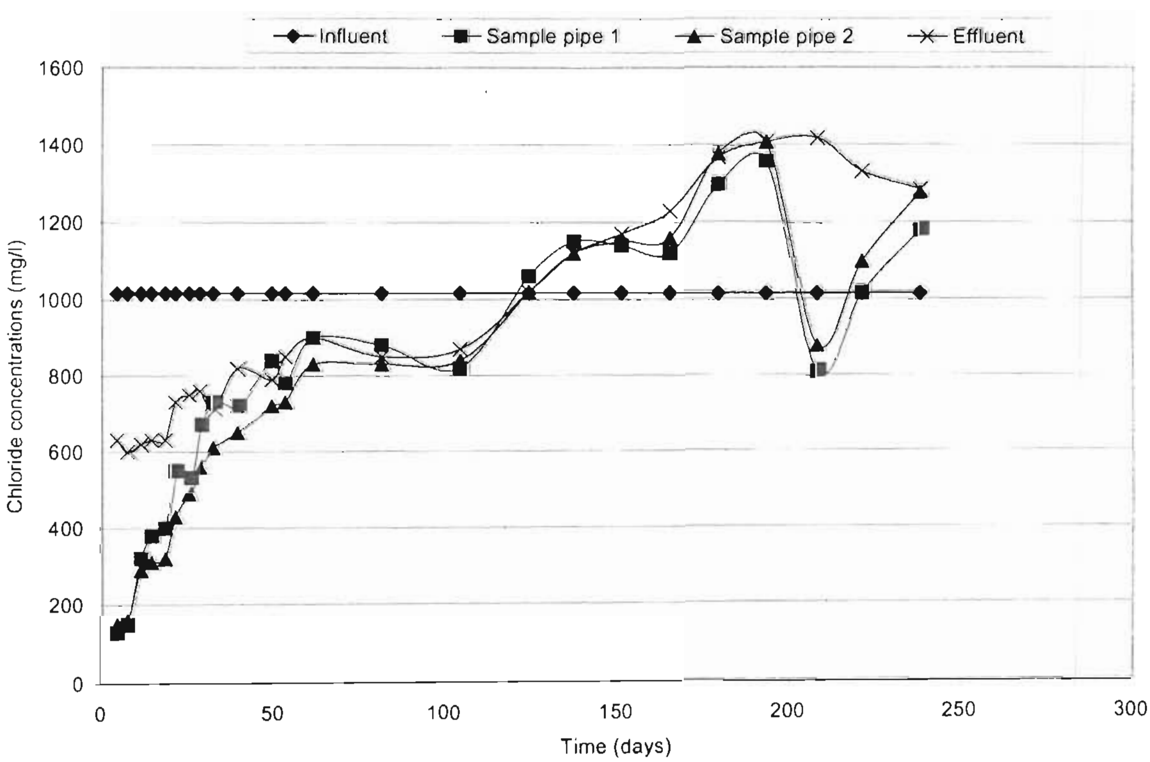


Figure 7.14: Chloride concentrations for the *Leersia hexandra* VSB.

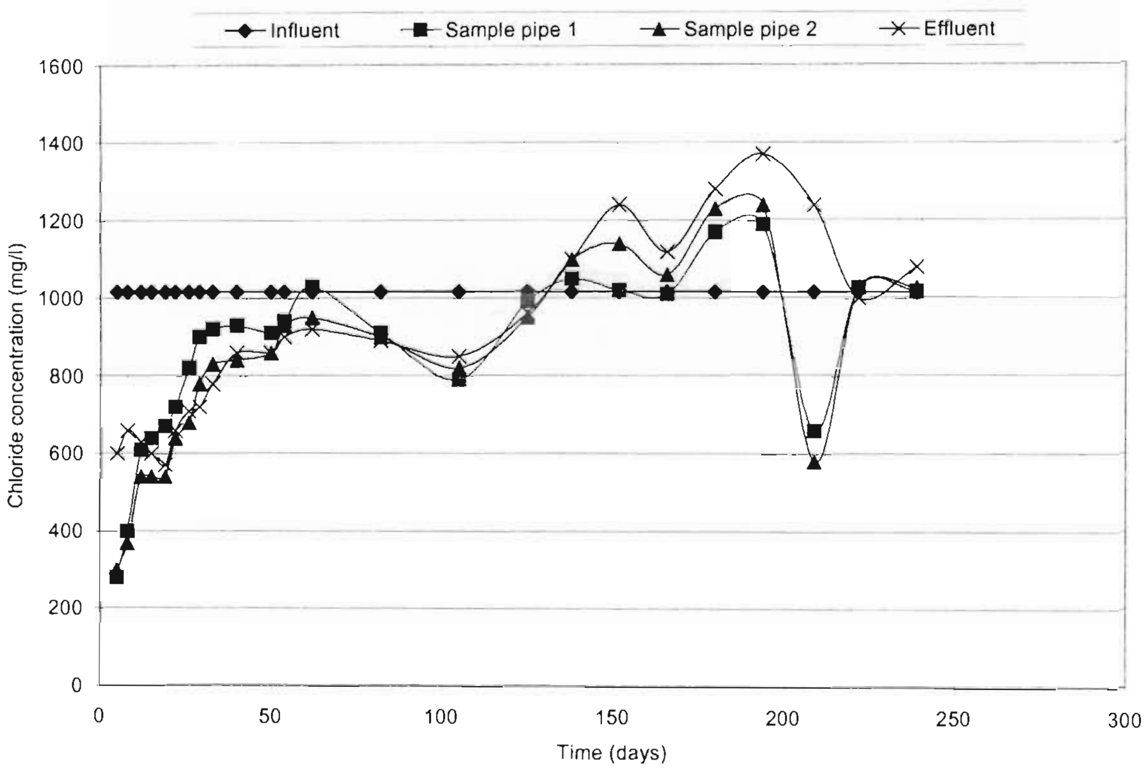


Figure 7.15: Chloride concentrations for the *Vetiver zizanioides* VSB.

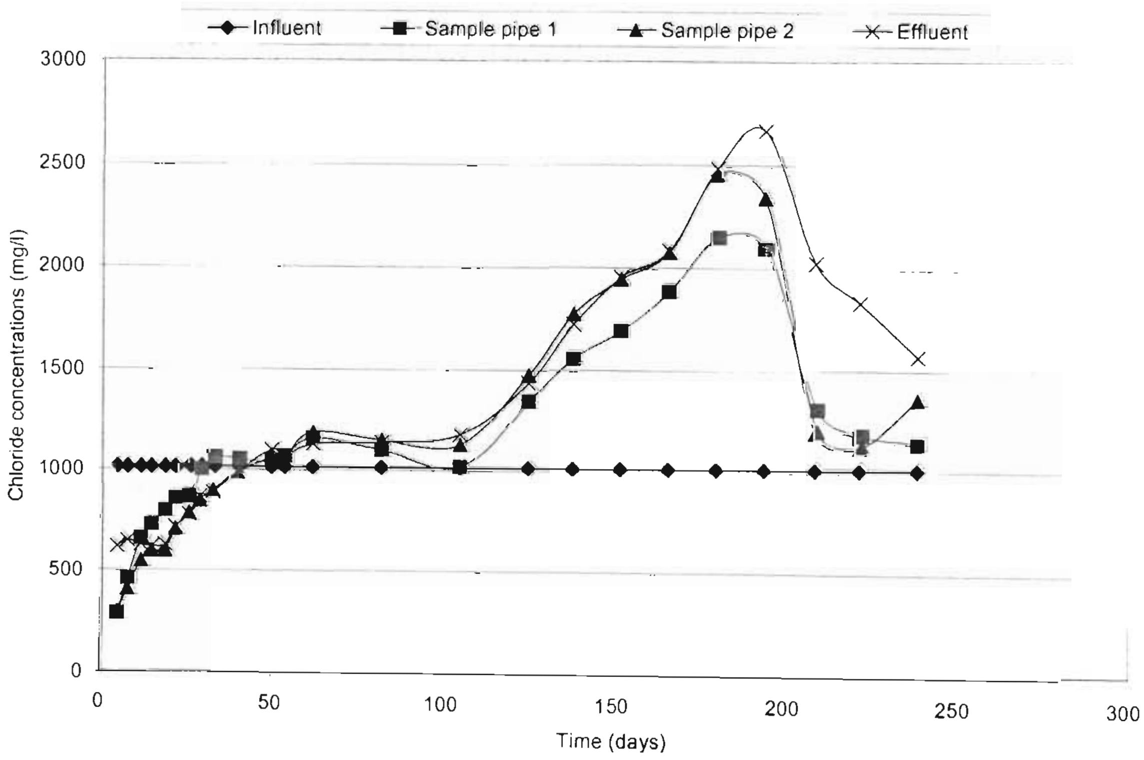


Figure 7.16: Chloride concentrations for the *Phragmites australis* VSB.

Before commenting on the results presented in Figures 7.13 to 7.16, it is necessary to evaluate the monthly average effluent flows and estimate the expected concentration of chlorides due to the effects of ET using mass balances (Table 7.3).

Table 7.3: Estimated chloride concentrations due to ET effects.

Month	BRE (l/d)	BR Cl^- (mg/l)	RE (l/d)	R Cl^- (mg/l)	BLE (l/d)	BL Cl^- (mg/l)	OE (l/d)	O Cl^- (mg/l)
J	12.83	1268	10.65	1528	13.63	1194	11.49	1416
F	19.84	820	19.18	848	18.17	896	15.71	1036
M	16.06	1013	14.48	1124	14.05	1158	10.92	1490
A	19.97	815	18.77	867	18.90	861	12.27	1326
M	15.41	1056	12.58	1293	13.28	1225	2.49	6535
J	16.07	1013	12.78	1273	14.25	1142	2.56	6356
J	22.58	721	19.23	846	21.23	766	16.53	984
A	15.06	1080	11.38	1430	14.11	1153	11.80	1379

Note: 1. Influent chlorides = 1017 mg/l.
2. Influent flow = 20 l/d.
3. Chloride efficiency factor = 0.8 (Table 7.2).
4. BRE = Control effluent flow; BR Cl^- = Estimated effluent chloride conc. for Control
5. RE = *Leersia* effluent flow; BLE = *Vetiver* effluent flow; OE = *Phragmites* effluent flow.

When evaluating the preceding Figures and Table 7.3 it was found that after stabilisation had occurred (Table 7.2) the effluent chloride concentrations varied according to the estimated concentrations for the respective VSB cells. The effect of the localised rainfall events are pointed out, in the preceding Figures, during the periods from day 50 to 120 and day 200 to 220, where the rainfall had a diluting effect on the effluent concentrations. This diluting effect had a greater influence on the sample points in the sample pipes as they were located in the top 200 mm of the cell (upper stone layer). The subsurface outlet system was found to behave in a more stable manner proving that the instantaneous mixing of the rainfall and the entire bed volume did not occur. The estimated effluent chloride concentration for the *Phragmites* VSB was much higher than the other three cells. Figure 7.15 points out the larger effluent concentration gradient experienced in this cell, tending towards the estimated value until the weaver bird incident caused the effluent flows to increase, diluting the effluent chloride concentration. The above findings showed that after the commencement of the treatability trials there was an initial period (approximately 3 bed volumes) were the effluent concentrations increased towards a stable range, which at the scale of the pilot VSB CW was strongly influenced by climatic variations, both seasonal and localised, as well as other external influences such as the weaver birds.

7.4.2 COD

Total organics were the primary pollutant of concern for the research with an objective to assess whether the pilot scale VSB CW would be able to remove the residual COD and meet the 75 mg/l general discharge limit. The results for COD from the treatability trials are presented in Figures 7.17 to 7.20 (Raw data presented in Appendix D3).

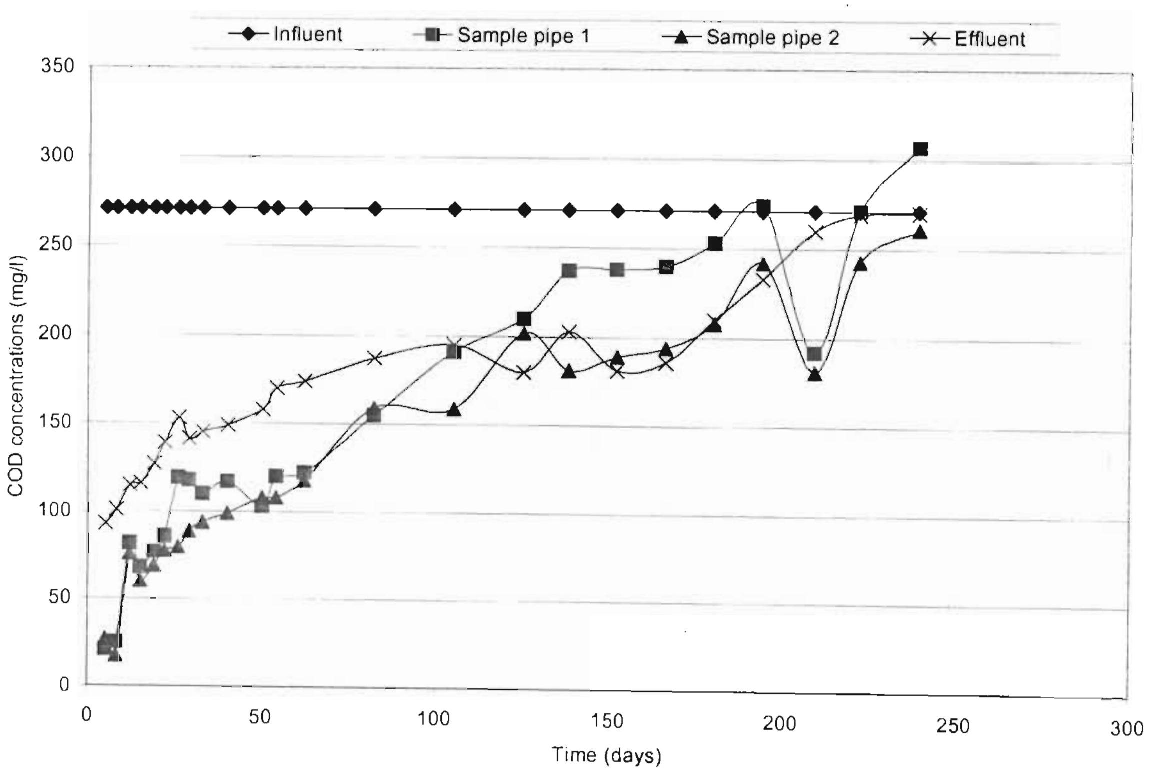


Figure 7.17: COD concentrations for the Control VSB.

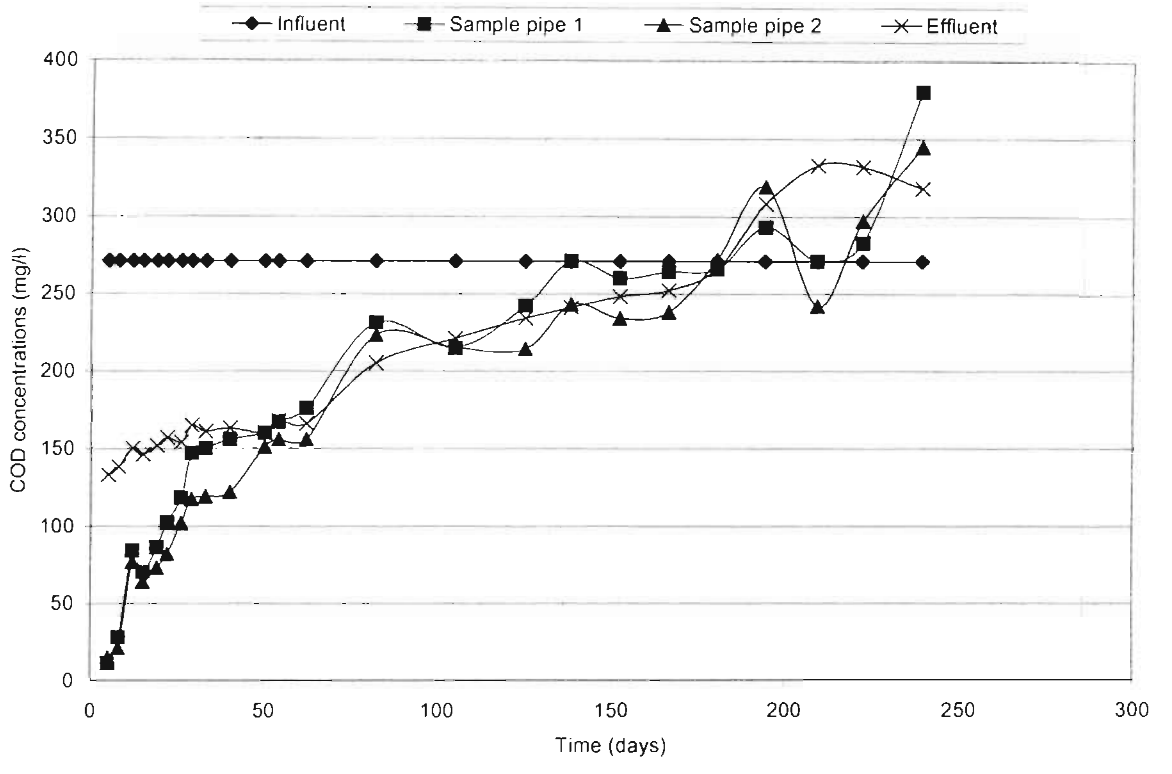


Figure 7.18: COD concentrations for the *Leersia hexandra* VSB.

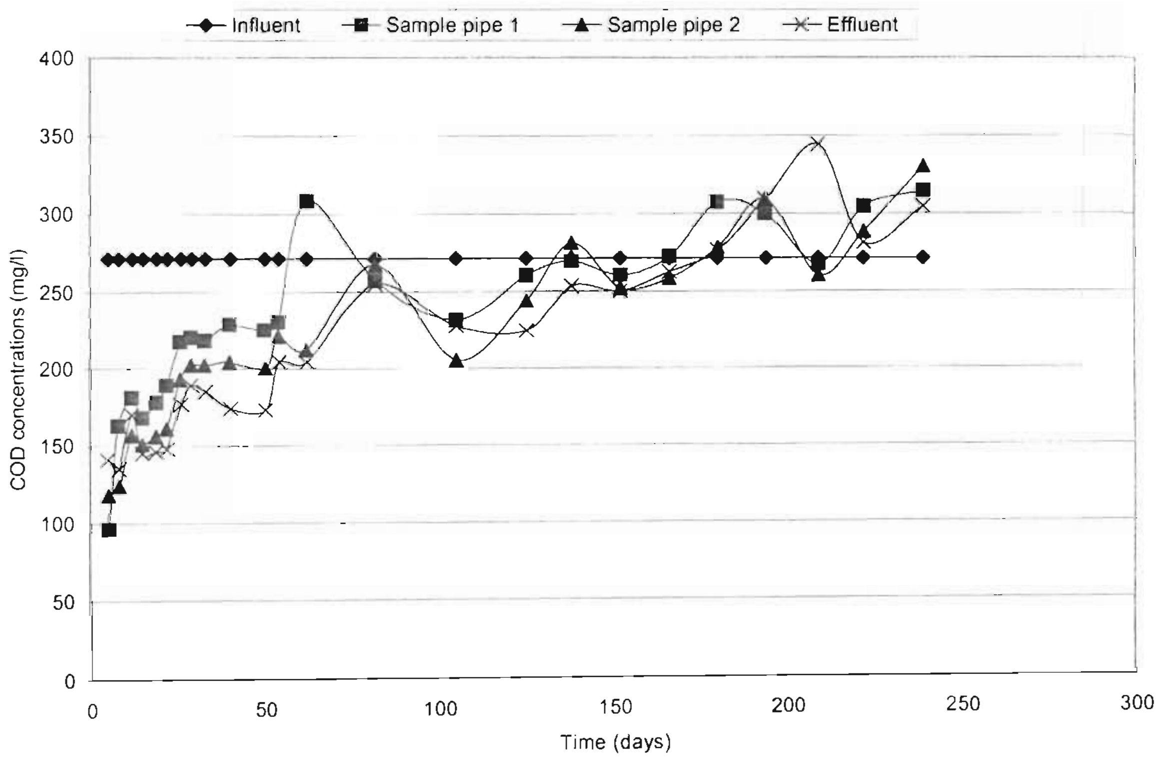


Figure 7.19: COD concentrations for the *Vetiver zizanioides* VSB.

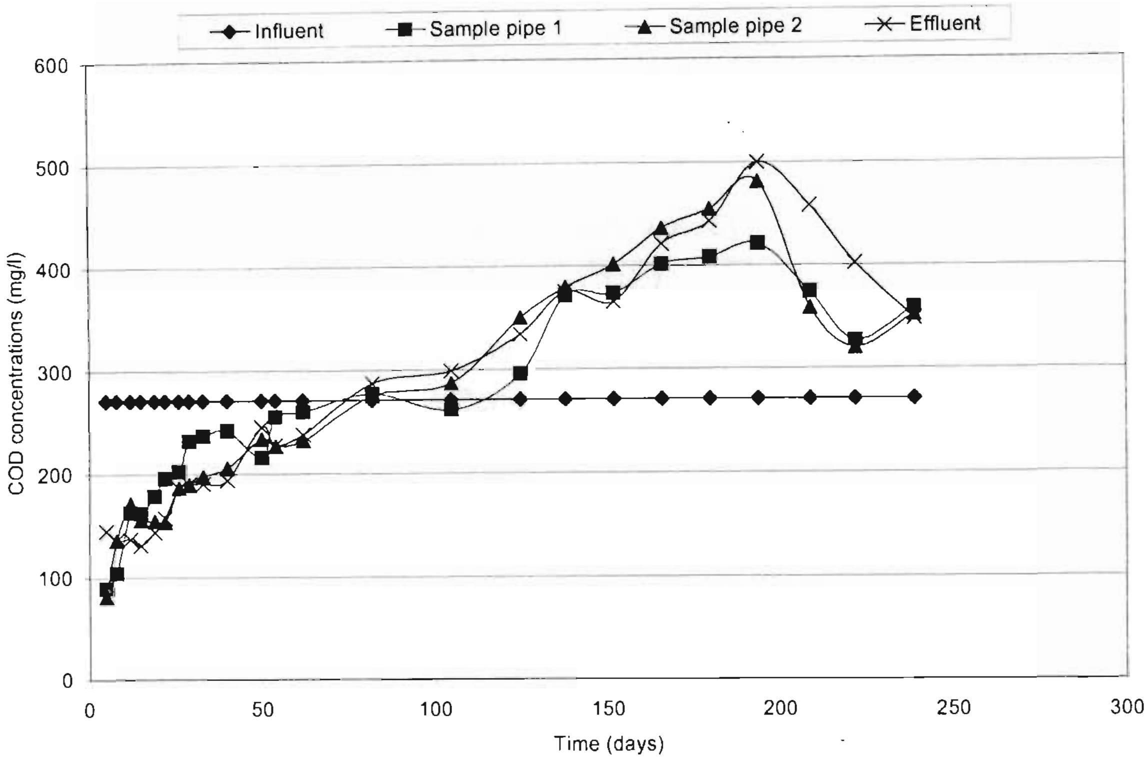


Figure 7.20: COD concentrations for the *Phragmites australis* VSB.

The effluent and sample pipe COD concentrations were found to follow a very similar trend to the chloride concentrations, indicating the initial increase in concentrations, tending towards a stable range. The influence of the localised rainfall events are also evident. The Control cell showed the expected tendency for the COD concentrations in the effluent and at sample pipe 2 to be lower than sample pipe 1, however, the other three planted cells did not show this idealised pattern and instead showed a very close pattern. During rainfall events, however, as expected after analysing the chlorides, the sample pipes were affected more than the effluent concentrations. The findings from the treatability trials pointed out that the specified discharge standard could not be met and required a removal efficiency of 72%. Arguably the scale of the system and hence the magnified effects of ET on the effluent concentrations did not help in achieving this goal. However, it did help increase the HRT. It may also be said that the residual COD comprised compounds with very low biodegradability and to obtain such high removal efficiencies in a biological system was not possible. Yet the question of toxicity and actual biological oxygen demand were still of concern.

7.4.3 BOD_5 and BOD_{20}

The results of the BOD_5 and BOD_{20} analyses conducted during the treatability trials are presented in Table 7.4 and 7.5 respectively.

Table 7.4: Results of the BOD_5 analyses conducted during the treatability trials.

Day	IN BOD_5 Influent (mgO_2/l)	OUT BOD_5 Control (mgO_2/l)	OUT BOD_5 *Leersia* (mgO_2/l)	OUT BOD_5 *Vetiver* (mgO_2/l)	OUT BOD_5 *Phragmites* (mgO_2/l)
5	10.0	6.8	7.4	10.9	10.0
40	12.1	10.9	11.8	15.1	10.0
62	10.9	8.3	9.4	7.7	7.4
82	6.8	6.5	5.0	6.8	5.3
125	5.3	2.1	1.5	2.7	3.1
152	4.0	0.0	0.0	0.3	0.0
180	3.4	0.0	0.0	0.3	0.0
209	4.0	0.0	0.0	0.0	0.0

Table 7.5: Results of the BOD_{20} analyses conducted during the treatability trials.

Day	IN BOD_{20} Influent (mgO_2/l)	OUT BOD_{20} Control (mgO_2/l)	OUT BOD_{20} *Leersia* (mgO_2/l)	OUT BOD_{20} *Vetiver* (mgO_2/l)	OUT BOD_{20} *Phragmites* (mgO_2/l)
5	15.4	12.4	11.8	24.2	15.1
40	19.8	12.7	16.2	25.4	12.7
62	16.5	12.4	10.6	15.6	10.9
82	11.5	8.0	6.8	12.7	10.9
125	28.2	4.1	4.3	7.6	8.0
152	13.2	1.4	1.5	5.3	1.1
180	10.8	0.4	3.5	6.5	2.8
209	10.9	0.0	2.0	4.2	1.0

The influent BOD concentrations were found to decrease during the treatability trials, arguably due to biological processes present in the primary storage tank. The control, *Leersia* and *Phragmites* cells all initially demonstrated limited reductions in levels of BOD_5

and BOD_{20}. The Vetiver cell initially showed slight increases, which was probably due to the alcohols and other biodegradable organics present in the oily substance excreted by the roots. The results do, however, show a reduction in BOD through the VSB CW indicating the presence of biological activity. Towards the end of the trials the BOD_5 concentration in the effluents tended towards zero, as do the BOD_{20} concentrations. The findings from these results point out that, although the effluent COD concentrations were above the required discharge limit, there was absolutely no biological oxygen demand risk to the receiving environment.

7.4.4 Ammoniacal nitrogen

The influent ammoniacal nitrogen concentrations already met discharge standards. However, there was concern of an increase along the VSB CW. The results of the monitoring of the ammoniacal nitrogen concentrations during the treatability trials are presented in Table 7.6.

Table 7.6: Results of the ammoniacal-N analyses conducted during the treatability trials.

Days	IN Ammoniacal Nitrogen Influent ($mgNH_3$-N/l)	OUT Ammoniacal Nitrogen Control ($mgNH_3$-N/l)	OUT Ammoniacal Nitrogen *Leersia* ($mgNH_3$-N/l)	OUT Ammoniacal Nitrogen *Vetiver* ($mgNH_3$-N/l)	OUT Ammoniacal Nitrogen *Phragmites* ($mgNH_3$-N/l)
12	0.85	0.4	0.5	2.2	0.7
54	0.85	0.4	0.6	1.6	0.7
82	0.85	0.4	0.6	1.4	0.7
138	0.85	0.5	0.5	1.4	0.7
166	0.85	0.6	0.7	0.7	1.0
194	0.85	0.6	0.7	0.7	1.0

The results from the monitoring of the ammoniacal nitrogen showed very little change in concentration through the VSB CW. They demonstrate a slight decrease in ammoniacal nitrogen for the control, *Leersia* and *Vetiver* VSB CW. However, ammoniacal nitrogen concentrations within the Vetiver cell did initially increase. This increase was probably due to the biological breakdown of the organic oily substance that is excreted from the Vetiver's roots. The *Phragmites* cell showed a slight increase towards the end of the trials. However, this may arguably be due to the effects of ET. The effluent ammoniacal nitrogen concentration from all the cells was found to remain below the specified discharge standard of 3 mg/l.

7.4.5 Total suspended solids

Influent Total Suspended Solids (TSS) values were slightly above the general standard of 25 mg/l. The initially high TSS values were arguably due to residual solids that were in the unwashed gravel after construction. Following the flushing out of these residual solids after a prolonged feeding period, effluent TSS concentrations were generally kept below the general discharge standard (Table 7.7).

Table 7.7: Results of the TSS analyses conducted during the treatability trials.

Days	IN TSS Influent (mg/l)	OUT TSS Control (mg/l)	OUT TSS Leersia (mg/l)	OUT TSS Vetiver (mg/l)	OUT TSS Phragmites (mg/l)
12	40	79	50	101	85
54	40	0	9	19	15
82	40	5	6	13	14
138	40	10	11	25	16
166	40	37	47	60	61
194	40	5	0	24	1

7.4.6 Conductivity

In order to meet the general discharge standards the influent electrical conductivity had to be reduced from 421 mS/m to 184 mS/m. This included the removal of salts such as chloride, which were being used in the trials as a tracer due to its ability to move through the system without being removed or transformed. The results from the treatability trials are presented in Appendix D3; they follow a very similar trend to chlorides, which have been explained earlier; the results pointed out that the VSB CW could not meet the specified discharge limit.

7.4.7 pH

pH values remained fairly stable throughout the trial period, with no dramatic changes and remained within the specified general limits of 5.5 to 9.5. There was a general trend for the sample pipe pH values to be slightly lower than that of the effluents. The results from the treatability trials are presented in Figures 7.21 to 7.24 (Raw data in Appendix D3).

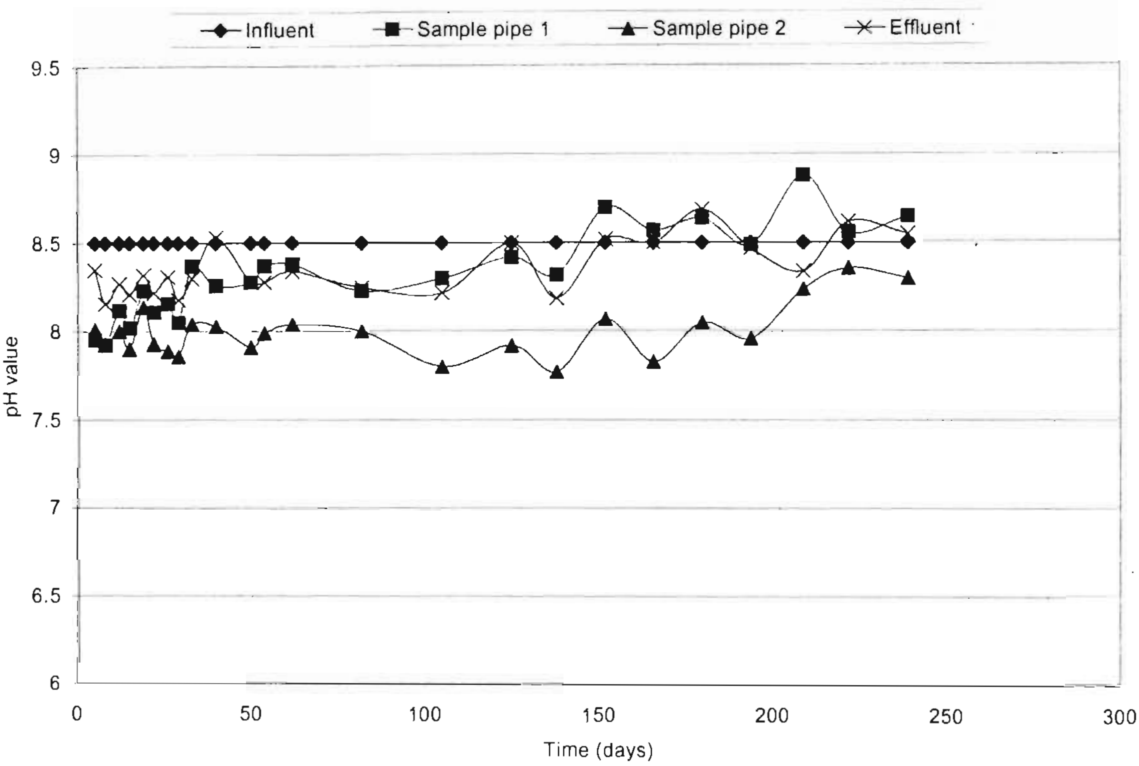


Figure 7.21: pH values for the Control VSB.

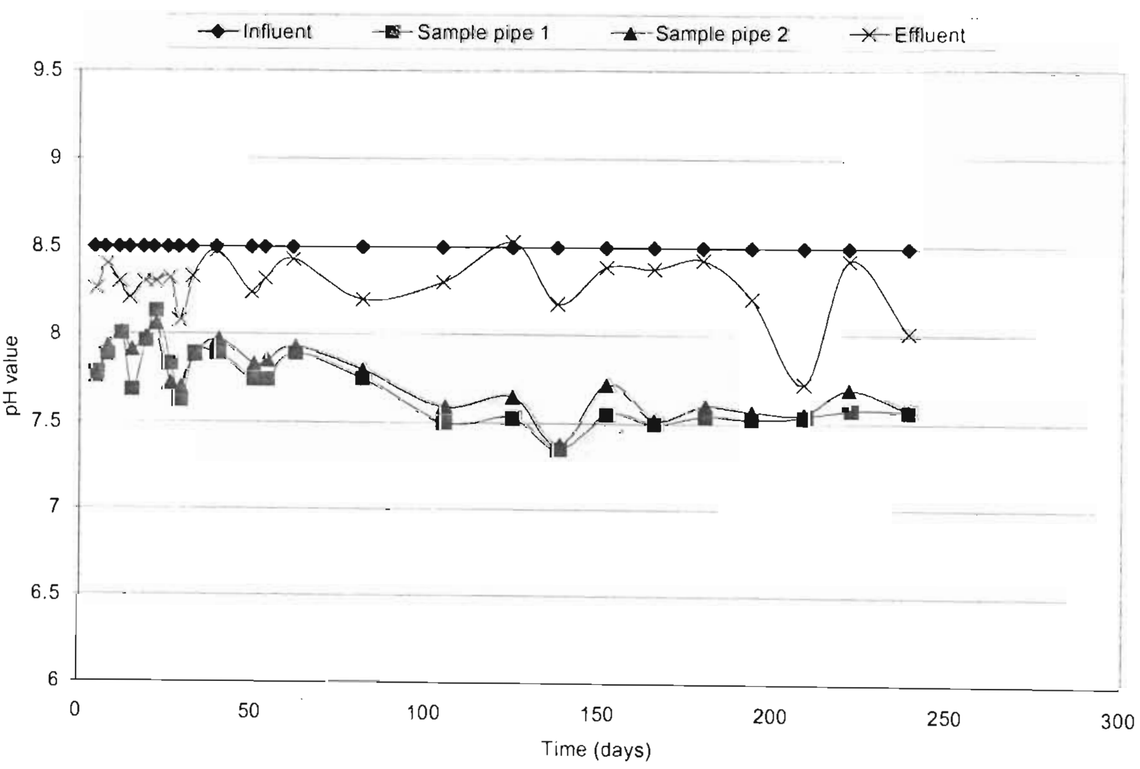


Figure 7.22: pH values for the *Leersia hexandra* VSB.

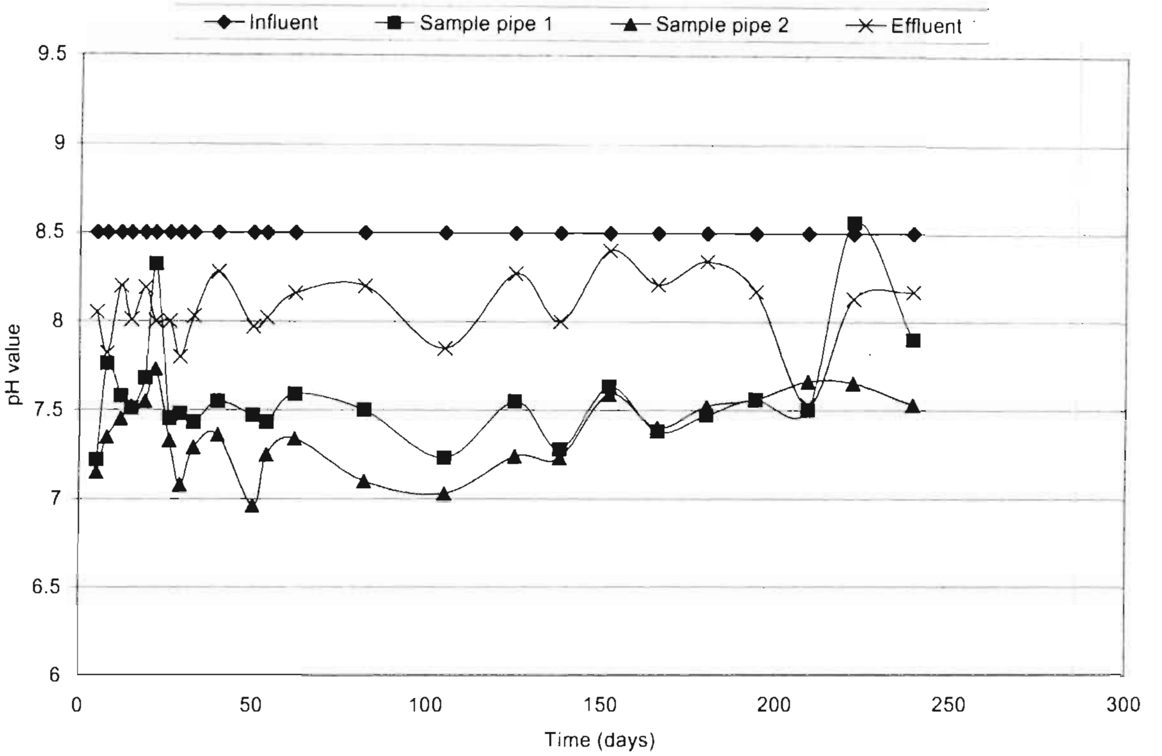


Figure 7.23: pH values for the *Vetiver zizanioides* VSB.

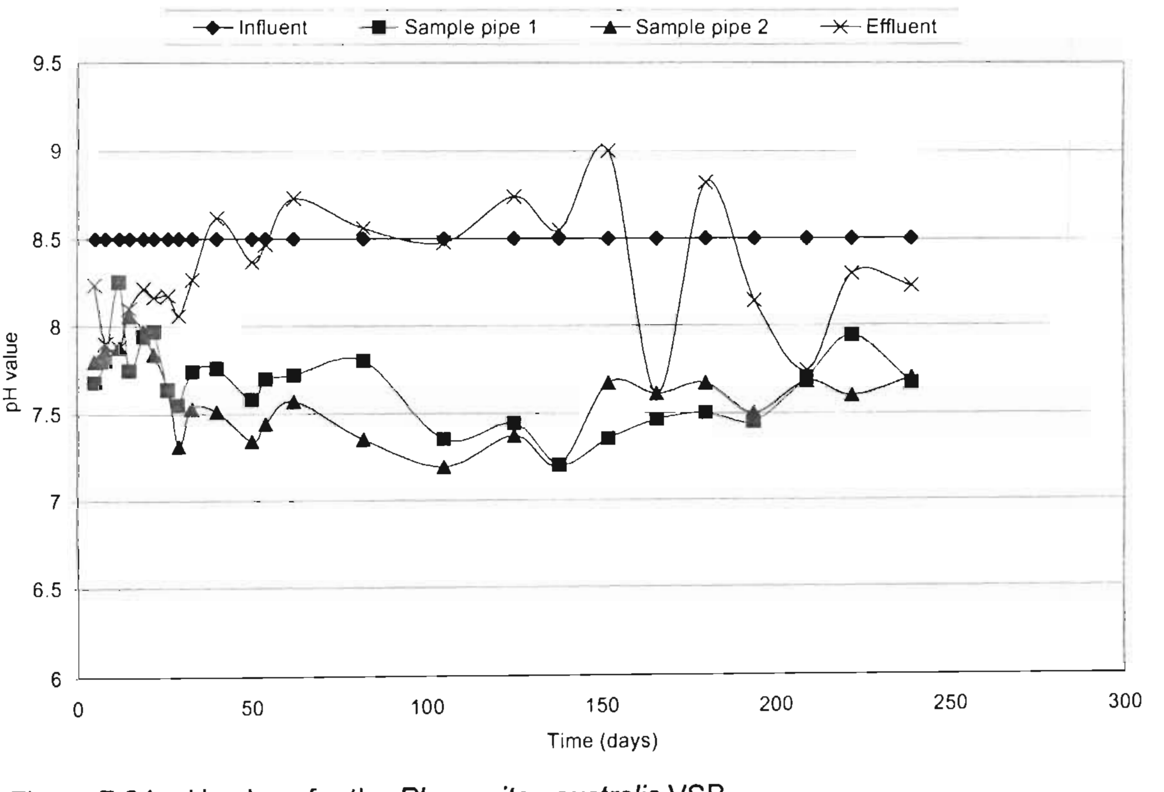


Figure 7.24: pH values for the *Phragmites australis* VSB.

7.4.8 Alkalinity

The results from the alkalinity tests are presented in Figures 7.25 to 7.28 (Raw data presented in Appendix D3).

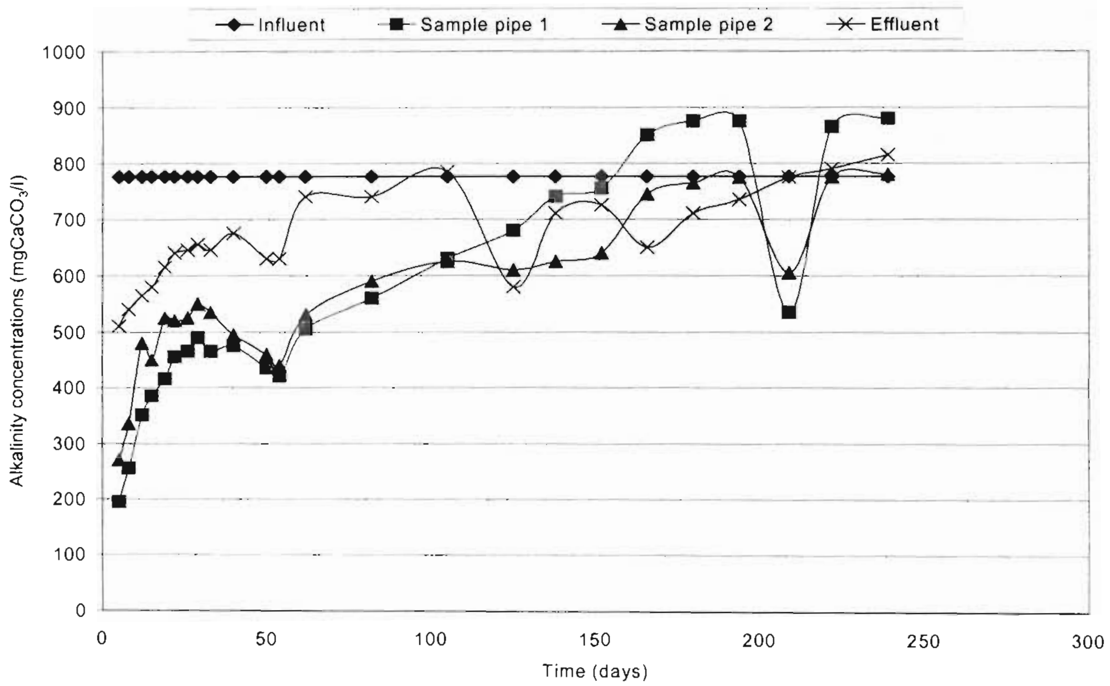


Figure 7.25: Alkalinity concentrations for the Control VSB.

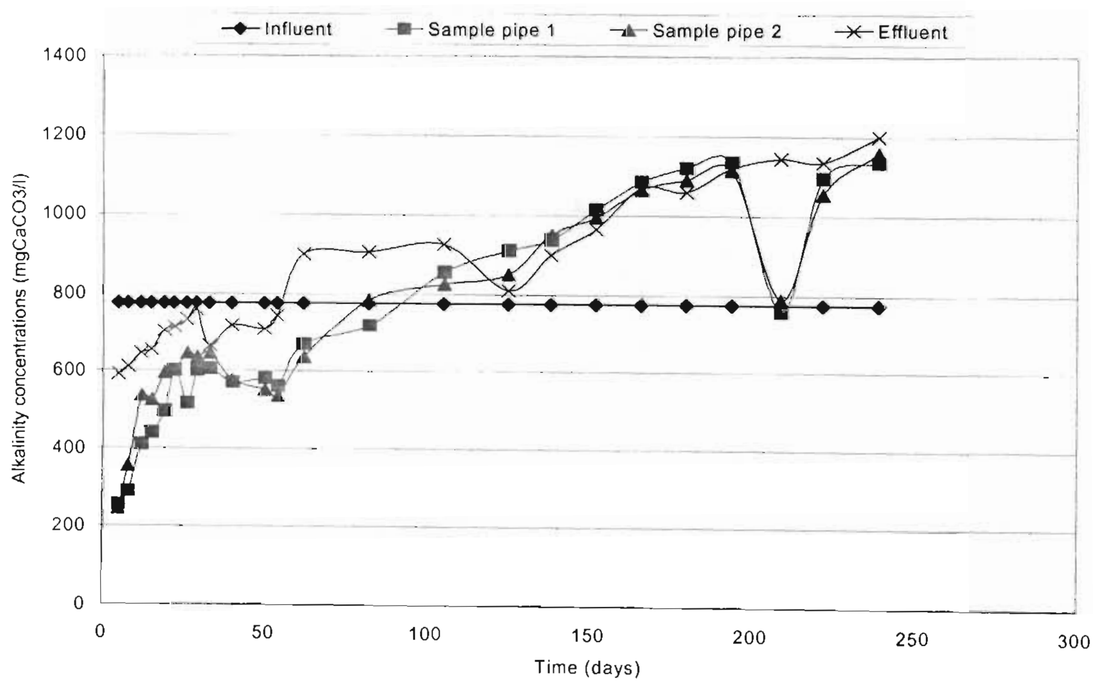


Figure 7.26: Alkalinity concentrations for the *Leersia hexandra* VSB.

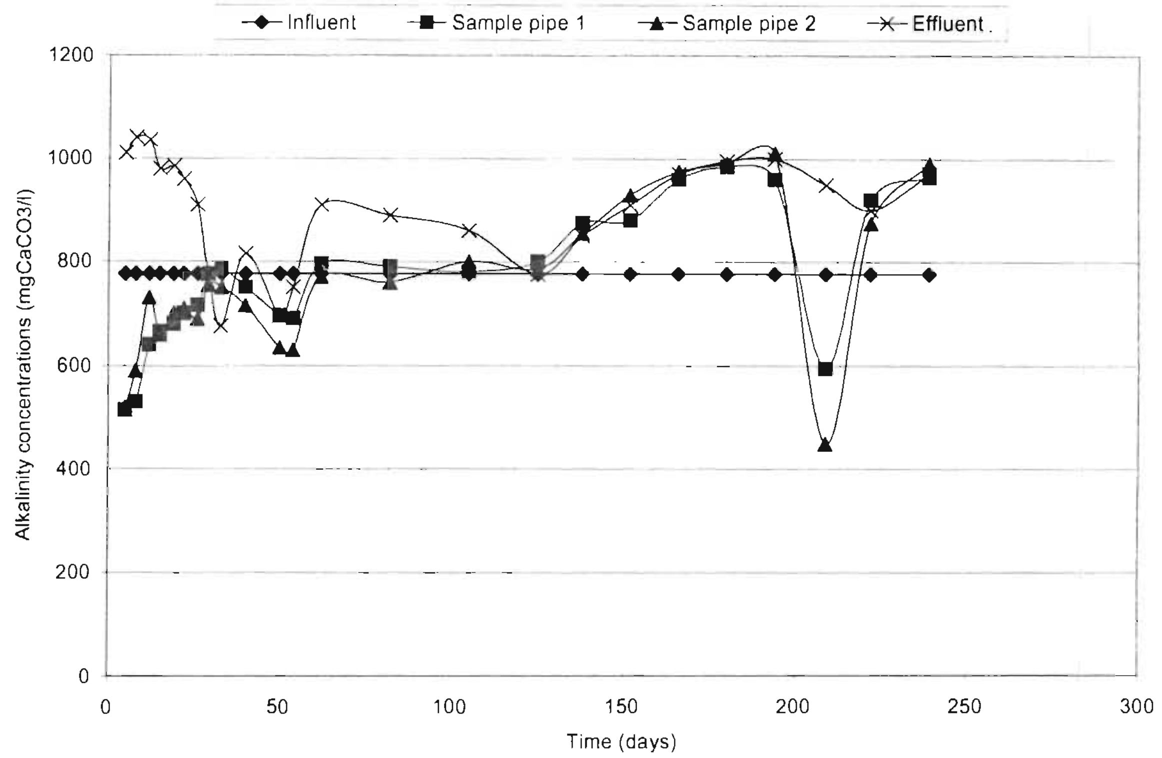


Figure 7.27: Alkalinity concentrations for the *Vetiver zizanioides* VSB.

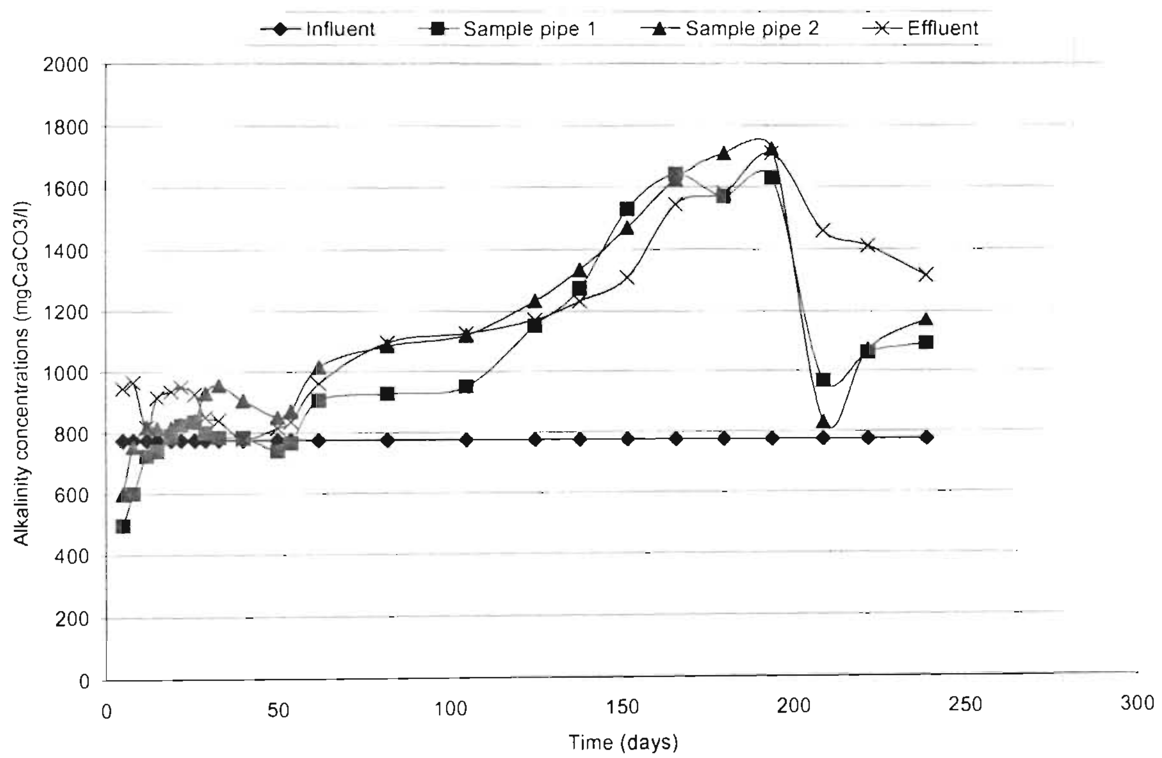


Figure 7.28: Alkalinity concentrations for the *Phragmites australis* VSB.

The results showed that the sample pipe and effluent alkalinity concentrations were also effected by the ET and localised rainfall events, however, the three planted cells showed a larger increase in alkalinity than the Control; reaching influent concentrations far before stabilisation at about 50 to 60 days. This indicated that there was an internal source of alkalinity, possibly due to microbial transformations. However, this would then have also been evident in the control unless the vegetation had enhanced the biological process in some way. It may have been from the vegetation alone, excreting some source of alkalinity into the rhizoshpere, which is quite possible for the *Vetiver zizanioides*.

7.5 COD and alkalinity mass balance

7.5.1 COD

By analysing the influent and effluent COD masses it was possible to view the effect that the VSB CW had on the mass of COD without the masking influence of ET. Rainfall events did however give theoretically erroneous results, as explained earlier in the tracer analyses. However, the overall trend may still be seen. The results from the COD mass balance are presented in Figures 7.29 to 7.32 (Raw data in Appendix D2 and D3).

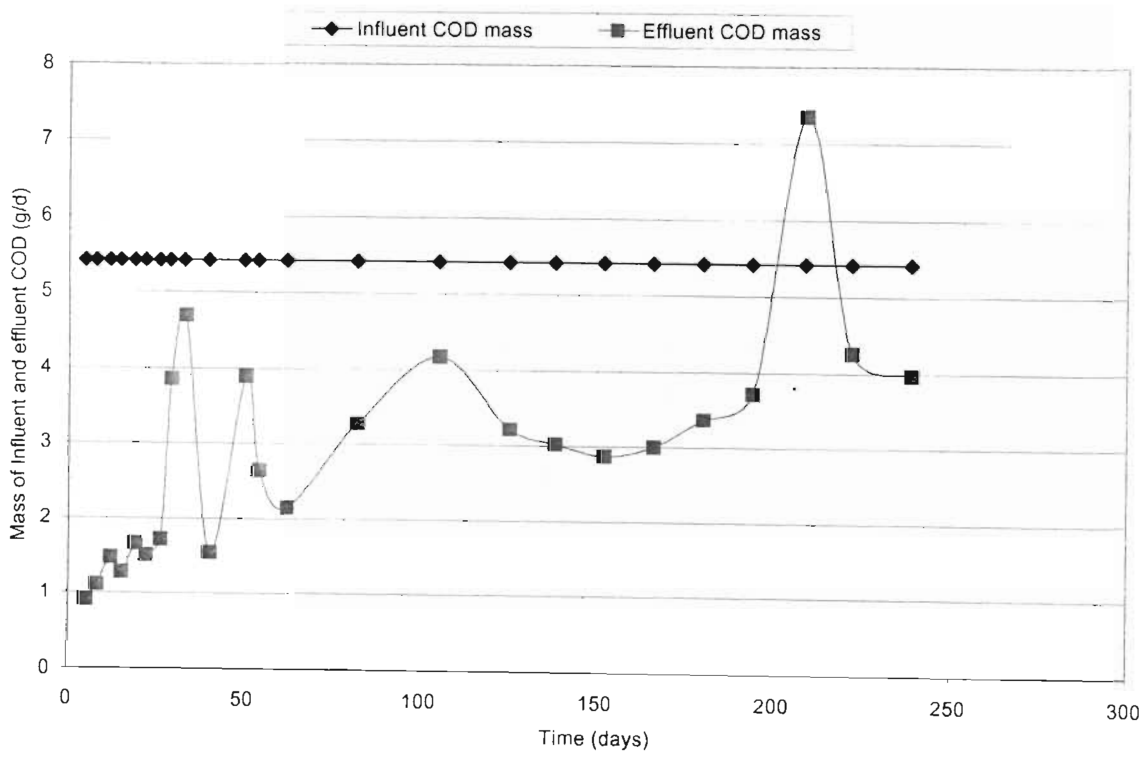


Figure 7.29: Influent and effluent COD masses for the Control VSB.

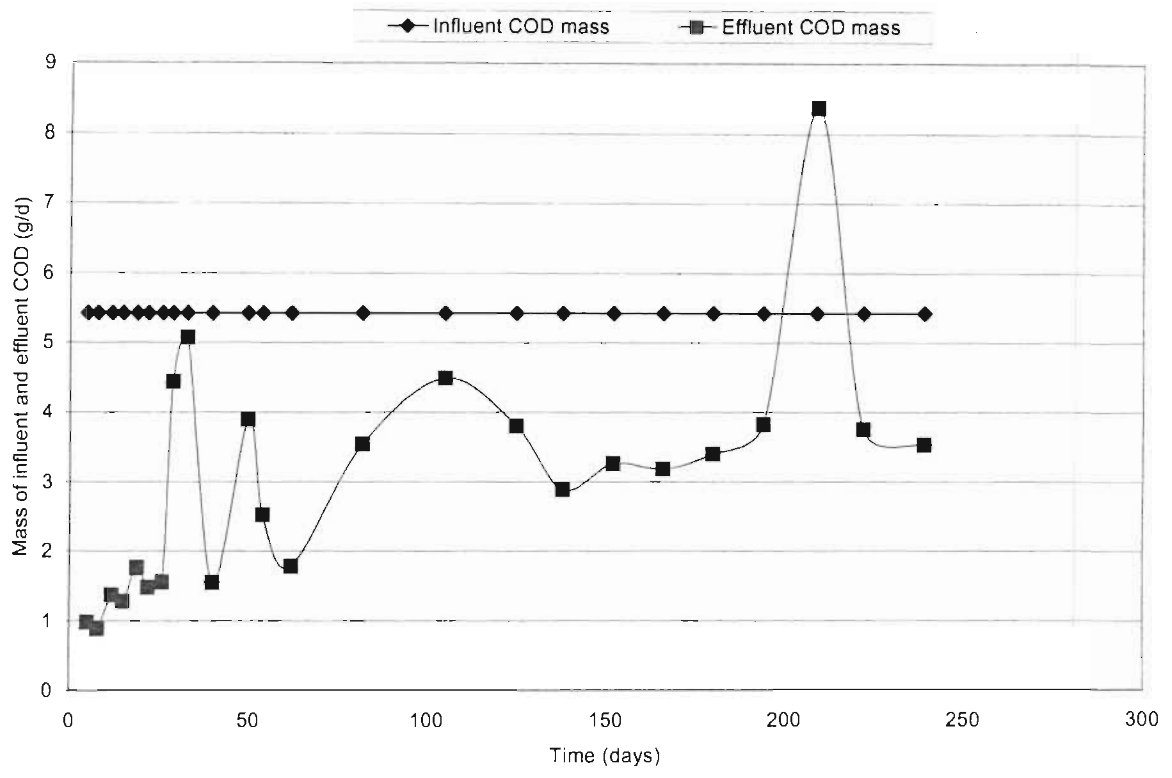


Figure 7.30: Influent and effluent COD masses for the *Leersia hexandra* VSB.

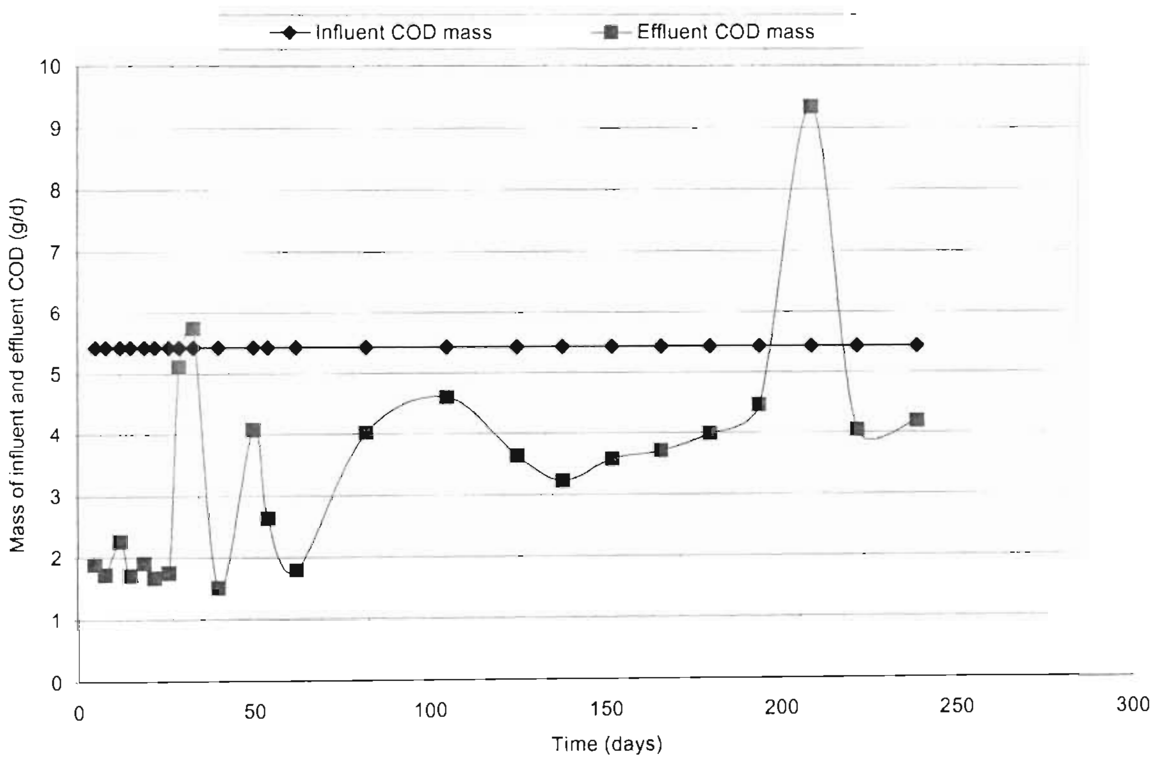


Figure 7.31: Influent and effluent COD masses for the *Vetiver zizanioides* VSB.

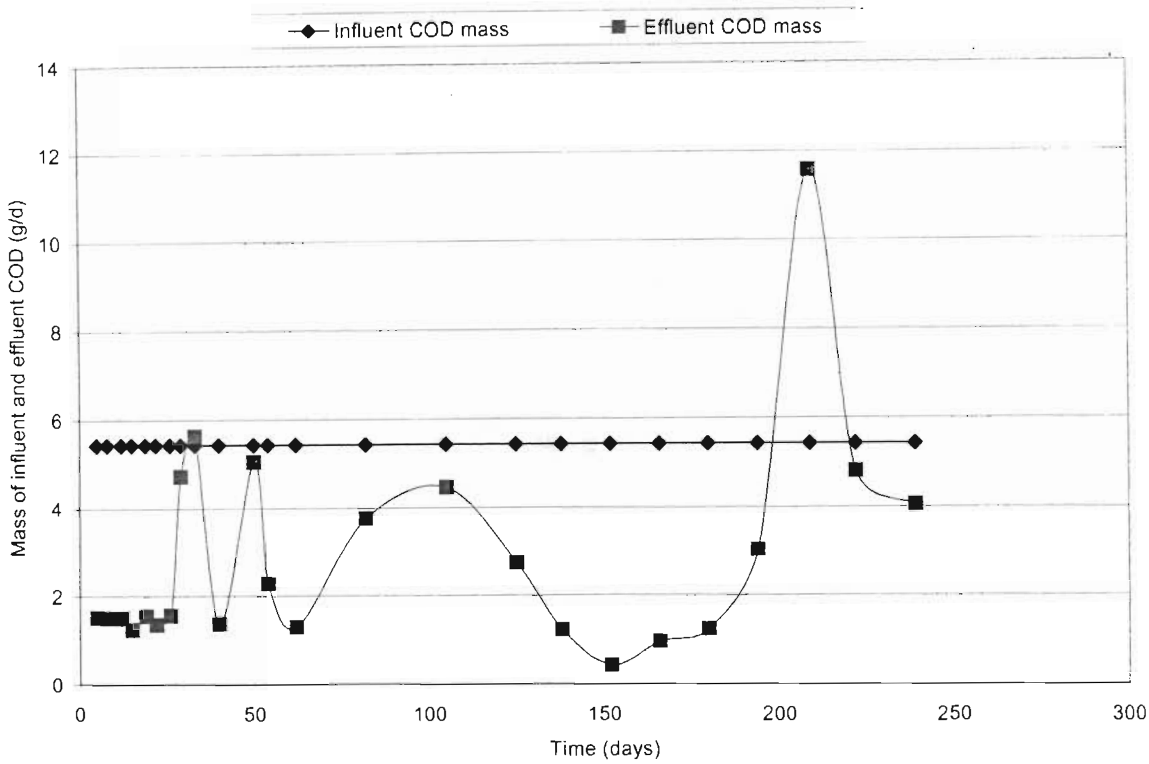


Figure 7.32: Influent and effluent COD masses for the *Phragmites australis* VSB.

All of the above figures point out that even though there was a net increase in COD concentrations there was actually a net decrease in the mass of COD in the respective effluents. This indicated that the VSB CW did actually remove a certain amount of the influent COD. However, the ET effects had a large enough influence to mask this removal. It could be said that if the discharge standards were set not on concentration but on mass per day then the VSB CW could aid as a mass removal system. However, during the localised rainfall events and due to the dynamics of the system, mass outputs can substantially increase; even beyond the influent mass. These events are, however, localised and recirculation either back onto the landfill or to the inlet of the VSB CW could be practised.

7.5.2 Alkalinity

Analyses of the influent and effluent alkalinity masses aided in the understanding of whether there was an internal source of alkalinity or if the effects of ET were once again responsible for the increase in the effluent concentrations. The influent and effluent alkalinity masses are presented in Figures 7.33 to 7.36 (Raw data in Appendix D2 and D3).

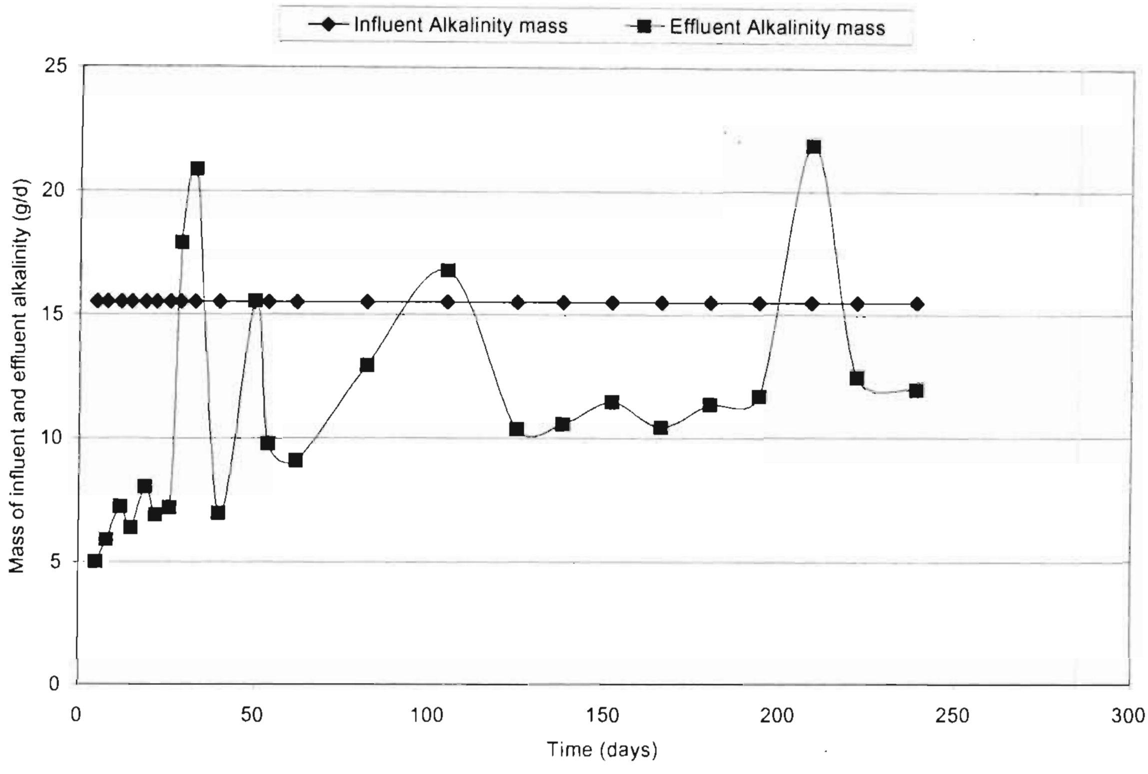


Figure 7.33: Influent and effluent alkalinity masses for the Control VSB.

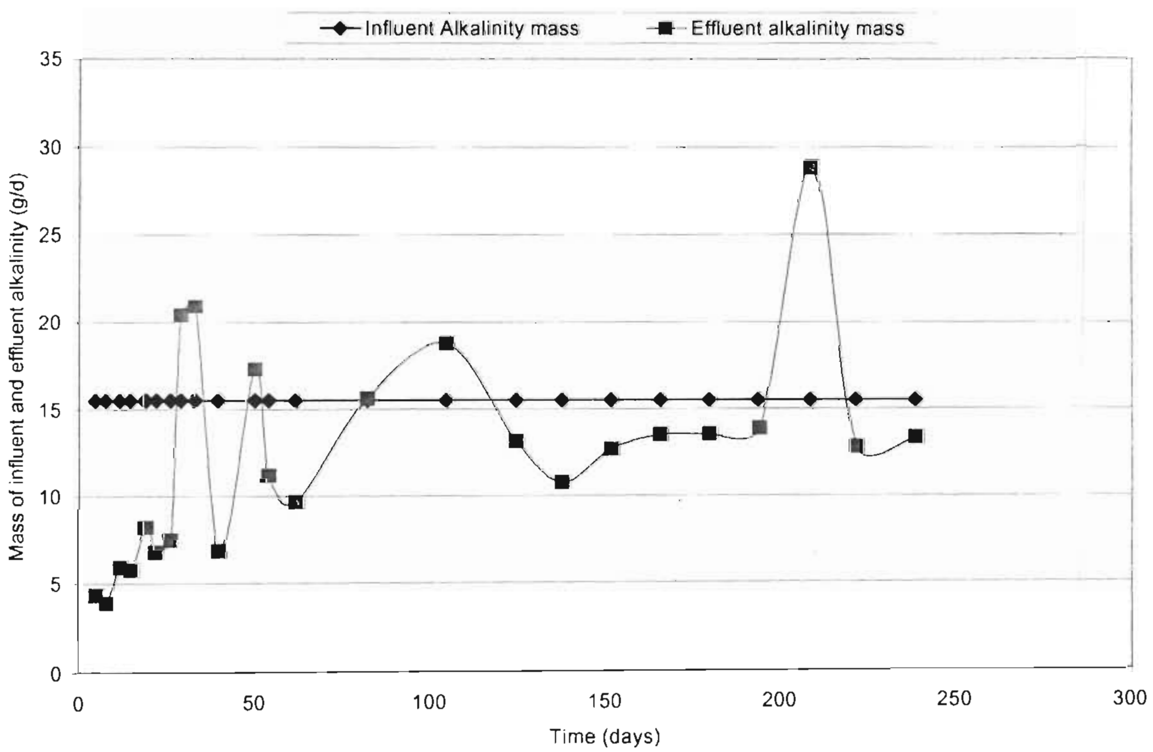


Figure 7.34: Influent and effluent alkalinity masses for the *Leersia hexandra* VSB.

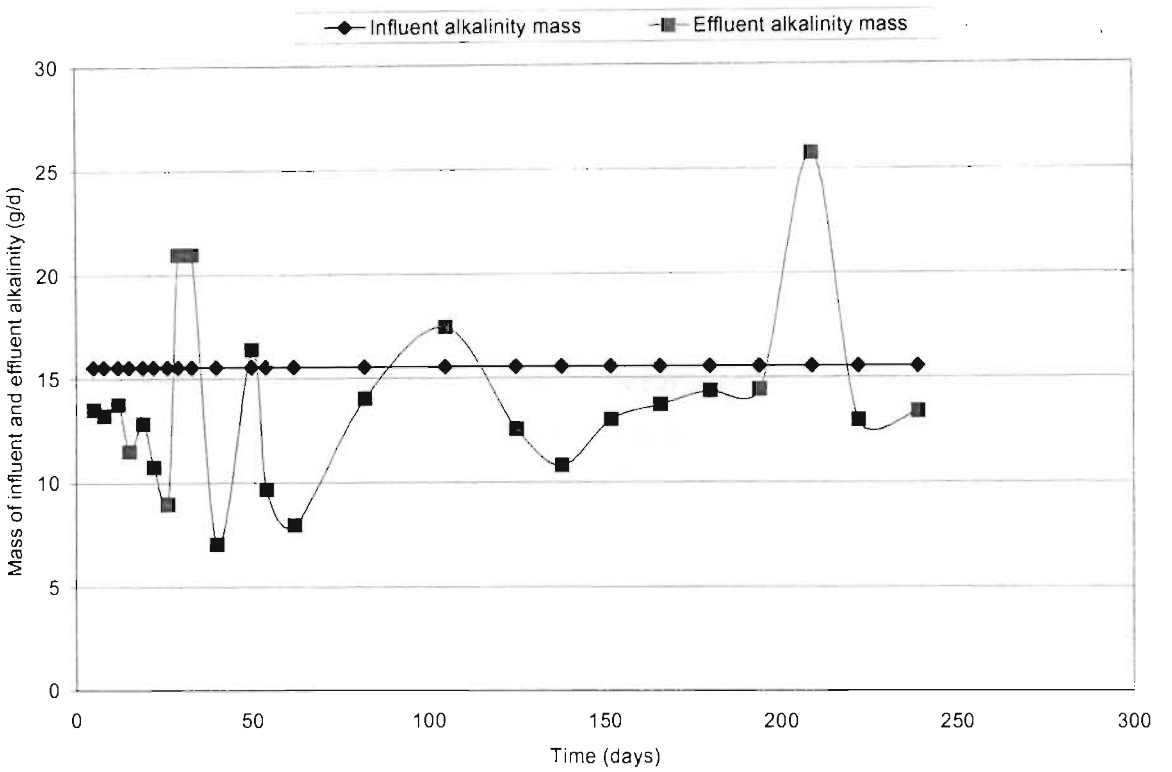


Figure 7.35: Influent and effluent alkalinity masses for the *Vetiver zizanioides* VSB.

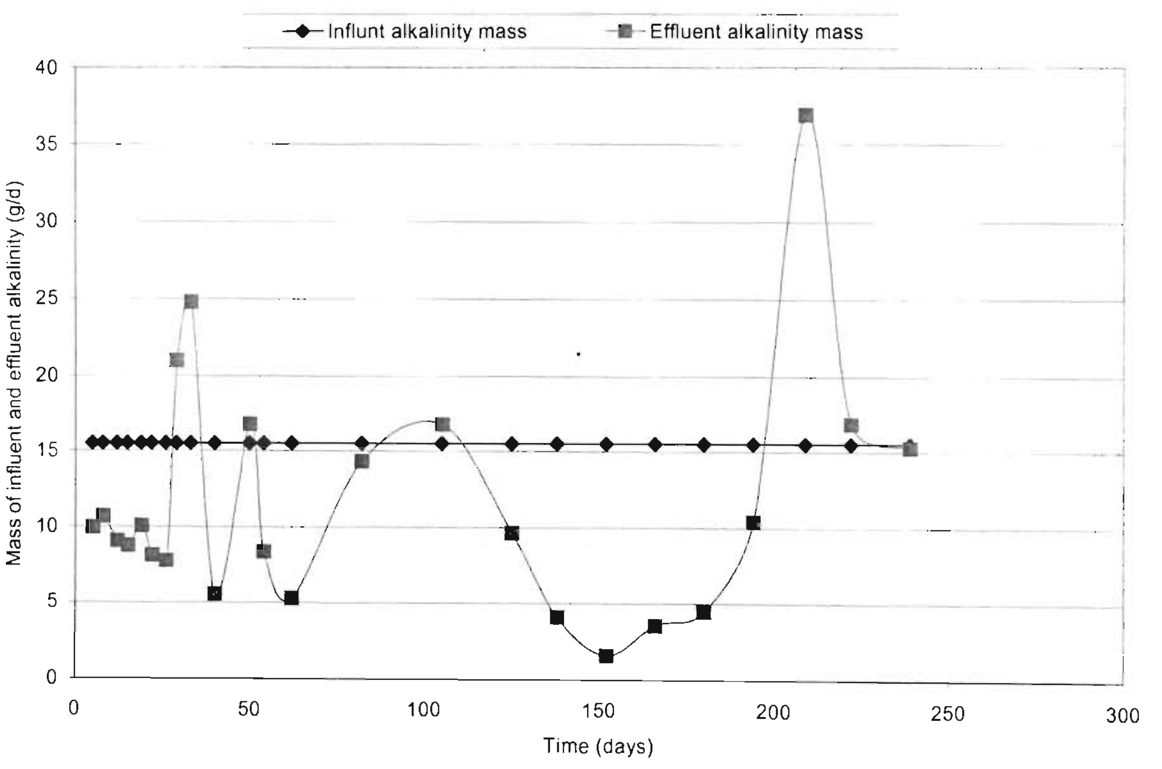


Figure 7.36: Influent and effluent alkalinity masses for the *Phragmites australis* VSB.

The results from the alkalinity mass balance pointed out that, in fact, there was no major internal source of alkalinity as the effluent mass would generally be higher than the influent mass if this was true and this only occurred during rainfall events. This showed again the masking effects of the ET.

7.6 Treatment efficiency

7.6.1 COD

The efficiency of removal is expressed in terms of concentration and mass. The results in terms of concentration removal are presented in Figure 7.37 (Raw data presented in Appendix D3), while the results in terms of mass removal are presented in Figure 7.38 (Raw data presented in Appendix D2 and D3).

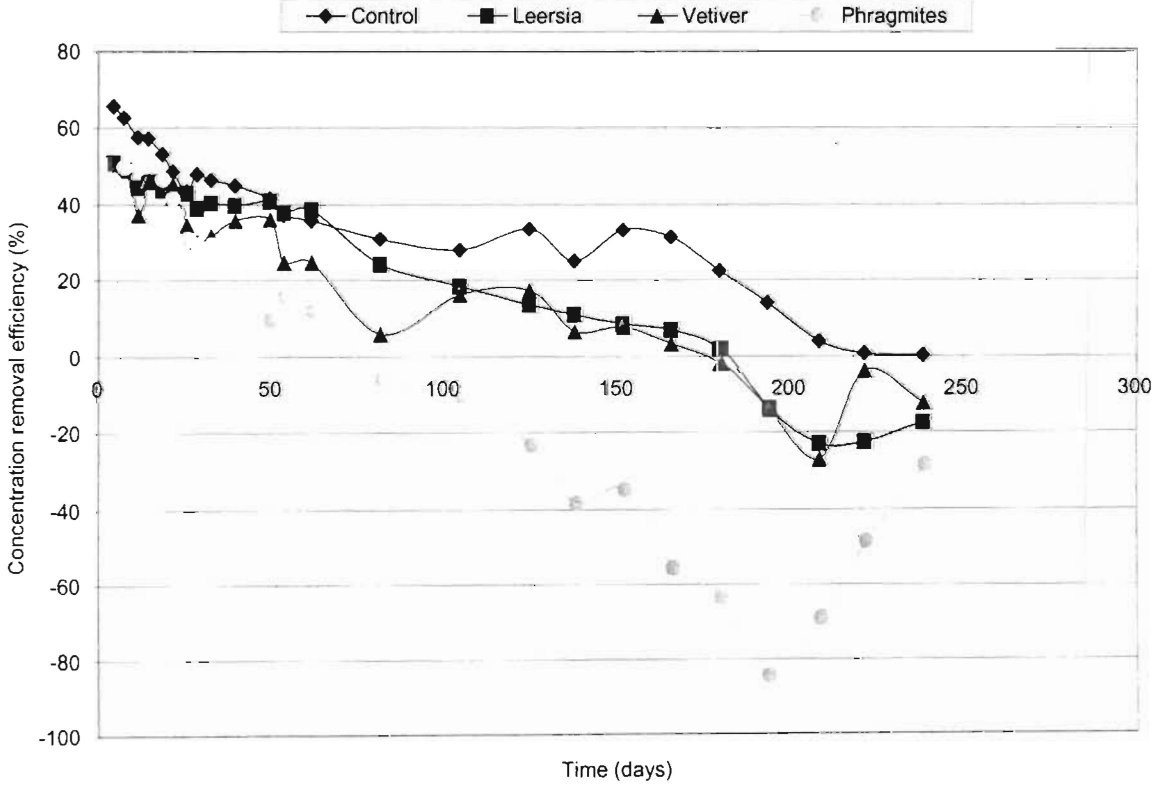


Figure 7.37: Percentage COD concentration removal efficiency.

Figure 7.37 clearly points out that the pilot scale VSB CW could not meet the specified discharge standard, which required 72% concentration removal efficiency. The negative removal efficiencies are entirely due to the effects of ET and the scale of the system.

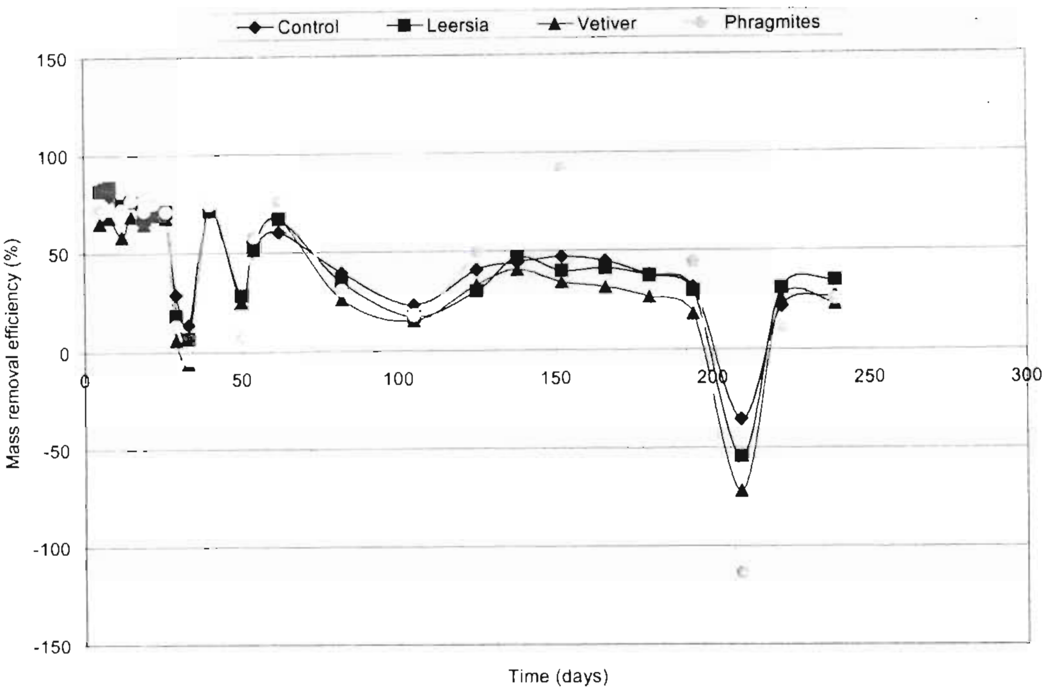


Figure 7.38: Percentage COD mass removal efficiency.

The mass removal efficiency of the VSB CW was generally found to be quite high with the maximum removal experienced in the *Phragmites australis* cell, however, the cell had not yet reached stabilisation and may tend towards the other cells in time. Localised rainfall events do, however, have a large adverse effect on the efficiency; due to the scale and dynamics of the system. Excluding rainfall events, the mass removal efficiency was found to be generally between 30 and 40%. Thus for a full scale system with a flow of 200 m^3/d and an influent COD concentration of 271 mg/l, 32520 g of 54200 would exit the VSB constructed wetland. By converting the general standard of 75 mg/l to a mass per day value by using the maximum discharge allowed of 2000 m^3/d (Figure 2.5), the limit mass allowed to be discharged per day would be 150000 g which means that the effluent from the VSB constructed wetland meet the mass discharge limit. During days of rainfall automated systems could discharge allowed volumes according to mass flow and recirculate overflows.

A further method of determining the COD removal efficiency of the pilot scale CW, taking account of the effects of dilution or concentration, is to plot the COD to chloride ratio against time, as shown in Figure 7.39.

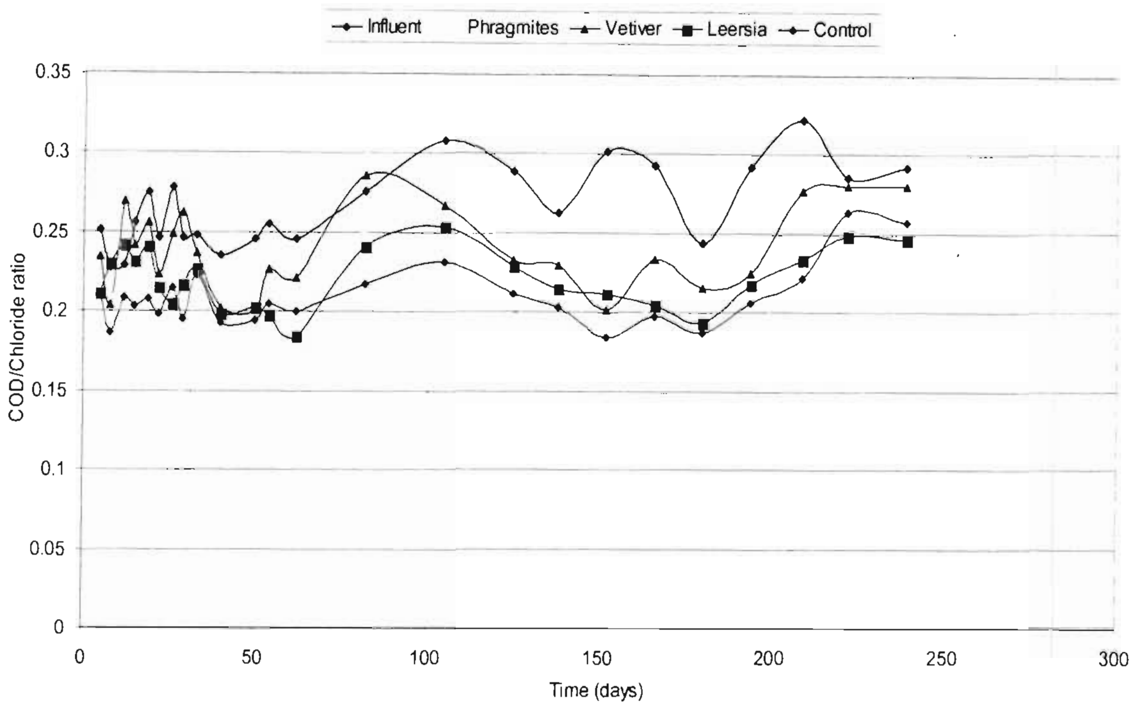


Figure 7.39: Removal efficiency in terms of COD to chloride ratio.

Further to the findings in Section 7.3, the estimated time of stabilisation for the pilot scale CW was 160 to 180 days. The ratios for the respective pilot scale VSB and the influent from 160 days to the completion of the trial are presented in Table 7.8.

Table 7.8: COD to chloride ratios.

Day	Influent	Vetiver	Control	Phragmites	Leersia
166	0.293	0.234	0.198	0.202	0.205
180	0.245	0.216	0.188	0.178	0.194
194	0.292	0.226	0.206	0.187	0.218
209	0.333	0.277	0.222	0.226	0.235
222	0.286	0.281	0.264	0.219	0.249
239	0.292	0.281	0.258	0.221	0.247
Average	**0.290**	**0.253**	**0.223**	**0.206**	**0.225**
Removal efficiency (%)		**12**	**26**	**38**	**31**

The above results show again that the greatest removal efficiency was found in the *Phragmites australis* cell. Both mass removal efficiency and COD to chloride ratio indicate that there was a net reduction in COD when taking account of the effects of dilution and concentration.

7.7 Toxicity and the Simulated natural receivers

The simulated receivers were monitored from day 54 until the end of the treatability trials on a monthly basis, however, data from day 102 is presented. It has already been established that the effluent from the VSB CW posed no biological oxygen demand threat to the receiving environment. The simulated receivers were used to show that the residual COD, which could not be reduced in the VSB CW, would also pose no toxic threat to the receiving environment. In order to achieve this the simulated receivers were monitored and the effluent from the respective VSB CW was sent for Microtox tests. The results from the Microtox tests are presented in Table 7.9.

Table 7.9: Microtox results for the effluent of the VSB CW.

VSB	Test 1	Test 2	Test 3
Control	Non-toxic	Non-toxic	Non-toxic
Leersia hexandra	Non-toxic	Non-toxic	Non-toxic
Vetiver zizanioides	Non-toxic	Non-toxic	Non-toxic
Phragmites australis	Non-toxic	Non-toxic	Non-toxic

Note: The 3 tests were done at equally spaced intervals during the treatability trials.

The results from the monitoring of the simulated receivers are presented in Tables 7.10 to 7.13.

Table 7.10: Results of the Control simulated receiver.

Day	COD (mgO_2/l)	Chlorides ($mgCl^-/l$)	pH	Alkalinity ($mgCaCO_3/l$)	Ammoniacal Nitrogen ($mgNH_3$-N/l)	Conductivity (mS/m)
102	370	790	9.53	520	0.9	327
129	522	930	9.39	540	0.9	361
156	370	1170	8.94	665	0.9	421
184	388	1420	9.00	705	1.0	476
216	356	1400	9.02	710	1.0	468

Table 7.11: Results of the *Leersia hexandra* simulated receiver.

Day	COD (mgO_2/l)	Chlorides ($mgCl^-/l$)	pH	Alkalinity ($mgCaCO_3/l$)	Ammoniacal Nitrogen ($mgNH_3$-N/l)	Conductivity (mS/m)
102	290	1000	9.18	765	0.6	411

129	238	1110	9.23	755	0.7	431
156	300	1420	9.08	865	1.1	519
184	339	1729	9.21	1030	1.1	614
216	314	1779	9.20	1030	1.0	605

Table 7.12: Results of the *Vetiver zizanioides* simulated receiver.

Day	COD (mgO_2/l)	Chlorides ($mgCl^-/l$)	pH	Alkalinity ($mgCaCO_3/l$)	Ammoniacal Nitrogen ($mgNH_3$-N/l)	Conductivity (mS/m)
102	370	1100	9.75	830	0.8	443
129	286	1170	9.23	850	0.9	454
156	370	1440	9.17	945	1.1	528
184	396	1659	9.32	1105	1.3	595
216	335	1649	9.51	1085	1.2	578

Table 7.13: Results of the *Phragmites australis* simulated receiver.

Day	COD (mgO_2/l)	Chlorides ($mgCl^-/l$)	pH	Alkalinity ($mgCaCO_3/l$)	Ammoniacal Nitrogen ($mgNH_3$-N/l)	Conductivity (mS/m)
102	526	1320	9.42	1010	1.2	538
129	526	1530	9.32	1055	1.4	578
156	510	1859	9.08	1200	1.7	685
184	677	2539	9.42	1535	1.7	882
216	517	2629	9.40	1545	1.7	872

The results show even increases in COD and chlorides due to the effects of ET, however, visual monitoring of the simulated receivers showed no sign of toxic effects. Fish populations continually increased, indicating reproduction. The respective plant species established and continued to grow, in fact the *Phragmites* simulated receiver showed the best growth in vegetation with the slower growth occurring in the less concentrated Control. The increased conductivity did not seem to pose a threat to the fish species and the ammoniacal nitrogen concentrations were kept below the discharge limit of 2 mg/l, hence posing no threat to the flora and fauna.

CHAPTER 8

8 CONCLUSIONS OF THE RESEARCH

8.1 Maintenance of the nitrification/denitrification SBR

The SBR system continued to prove its success; being easily maintainable and achieving excellent removal efficiencies for nitrogen. Foaming must, however, be expected as part of the operational problems of a nitrification/denitrification system. It was found that the addition of an anti-foam solution during the foaming periods suppressed the foaming and eventually caused it to stop. The foaming did not, however, reduce the nitrogen removal efficiency.

8.2 Maintenance of the pilot scale VSB constructed wetlands

The inlet system was found to work well, achieving even distribution. 'Clogging' of the system did occur twice during the treatability trials. However, it was not due to the influent solids, but due to wind driven particulates including plant vegetation and seeds. The encroachment of the vegetation into the inlet and outlet zones did not pose a problem for the treatability trials. However, over a longer time period and for a full-scale system the inlet and outlet zones would have to be occasionally cleared to avoid clogging of the inlet and outlet systems by plant foliage and rhizomes. The subsurface outlet system was also found to work well during the treatability trials. The system did not 'clog'. However, for a full-scale system it would be advisable to include a 'rodding' hole that would aid in the occasional cleaning of the accumulated solids over long periods of time. The *Leersia hexandra* and *Vetiver zizanioides* proved to be easily maintainable with high pest resistance. The *Leersia* was found to show rapid lateral growth, which may, under full-scale conditions, require more frequent inlet and outlet clearing. The *Phragmites australis* was found to require a longer period for establishment and was very susceptible to pests such as aphids (spring to autumn) and weaverbirds (spring). The use of the *Phragmites* foliage by the weaverbirds for nest building may be seen as an environmental value. However, depending on the scale of the system, they may have an impact on the overall system performance. All three vegetation species were successfully planted via the rhizome method during mid-winter.

8.3 Influence of the climatic factors

Due to the low BOD_5 to COD ratios of the influent mix, it was understood that typical HRT of a few days would not be adequate in removing the slowly biodegradable organics that constituted the influent COD and that a longer HRT would be necessary. The HRT is inversely proportional to the HLR and in order to achieve the increased HRT the HLR must decrease, thus the influence of ET on the system performance increases. The dynamics of the system proved that rainfall events had a localised influence on the system performance, while the effects of ET had a more lasting overall influence. For the Durban area it was found that the dominating climatic influence was the ET, which had a concentrating effect on the system pollutants. The ET does help to increase the HRT, however, due to the low biodegradability of the residual COD the concentrating effect of the ET dominated.

8.3.1 Vegetation and ET

All the vegetated cells showed larger evapotranspiration losses than that of the unvegetated control, indicating the influence of the transpiration component. It was found that the transpiration component of the ET for the *Phragmites* was substantially larger than that of the *Leersia* and *Vetiver* (1.7 times the Class-A-Pan), arguably due to the different plant structures and leaf surface areas. The losses for the *Leersia* and *Vetiver* were found to be very similar (0.3 times the Class-A-Pan).

8.4 Treatment performance of the pilot scale VSB constructed wetlands

8.4.1 Concentration removal efficiency

Due to the low biodegradability of the influent COD, even at the increased HRT, the VSB constructed wetlands were not able to reach the specified discharge standard of 75 mg/l. It was actually found that the increased HRT had an adverse effect on the effluent concentrations, due to the influence of the ET. As mentioned, by increasing the HLR the influence of the ET would decrease. However, by doing so, the system would still not be able to meet the specified discharge standard in terms of concentration. The findings of the research proved that the influent organics could not be removed, in terms of concentration, and that it composed of very low biodegradable compounds that would pose very little or no oxygen demand on the receiving environment as measured by the BOD or COD analytical tests. The simulated natural receivers also proved that the effluent

COD concentrations from the VSB constructed wetlands posed no toxic effect on the receiving environment.

8.4.2 Mass removal efficiency

The findings of the research showed the ability of VSB constructed wetlands to act as a mass removal system. The influence of the ET on the effluent concentration was found to be negative. However, the converse was found to be true for mass removal with the maximum mass removal efficiency occurring in the *Phragmites* VSB. Localised rainfall events did show an increased mass output, however, recirculation of the effluent during these periods could be practiced in order to reduce the mass of contaminant discharged into the receiver. Usually the pollutant mass input to a natural watercourse is of greater importance than the pollutant concentration. When assessing the allowable COD mass output set by the discharge standards it was shown that the VSB constructed wetlands can easily achieve the allowable mass output per day. The idealised case would be a zero discharge system. However, this becomes impractical when dealing with large flows due to the size of VSB required to achieve zero discharge.

8.5 Full scale recommendations and alternative treatment systems

The pilot scale treatability trials showed that VSB constructed wetlands would not be capable of meeting current discharge standards, set according to concentration. It is recommended that the current discharge standards be reevaluated for the case of the Bisasar Road landfill site as the effluent COD from the VSB constructed wetlands showed little or no oxygen demand and toxicity threat to a natural receiver. If the COD concentration standard had to be met, it could not be done in a biological system due to the very low biodegradability of the organic compounds that constitute the residual COD and a physical or chemical treatment system, such as activated carbon adsorption or reverse osmosis should be used. It is, however, important to point out that constructed wetlands for the polishing of landfill leachate may carry a potential value by creating an educational and recreational environment that can positively affect public perception.

8.6 Future research

The findings of this research show that a biological system will not be capable of removing the residual COD, in the effluent from the Bisasar Road Landfill site pilot scale SBR, to current discharge consent. If DWAF do not increase the COD consent value it will be necessary to implement a different treatment technology, other than a biological system, to achieve consent. Research into the use of activated carbon adsorbers may be one option. However, activated carbon adsorbers may be found to be costly due to the continual carbon renewal requirements as the carbon's adsorption capacity is reduced. Another option may be reverse osmosis. Research into the costs of using such a system and the implications of the concentrate disposal with regards to legislation would be required before implementation into a full scale plant.

Research into the sociological effects that a CW has on public perception may be of benefit for landfills in areas where public pressure is the dominant driving force behind the landfill's development. Findings may indicate that although the CW may not be playing any part in the treatment, by having a small CW system after the actual treatment may beneficially influence the publics perception.

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APPENDIX A

NOTE: Information regarding the drainage regions referred to in Table A1 can be obtained from DWAF, upon written request.

Table A1: Areas excluded from General Authorisation for discharges to water resources

Primary drainage region	Tertiary drainage region	Description of main river in drainage region
B	B11, B12 B20 B31, B32 B41, B42 B60	Olifants River Wilge River Olifants River Steelpoort River Blyde River
W	W51, W52, W53, W54, W55, W56, W57	Usutu River
X	X11, X12, X13, X14 X21, X22, X23, X24 X31, X32, X33 X40	Nkomati River

Table A2: Listed Water Resources.

	WATER RESOURCE
1	Hout Bay River to tidal water
2	Eerste River to tidal water
3	Lourens River to tidal water
4	Steenbras River to tidal water
5	Berg and Dwars Rivers to their confluence
6	Little Berg River to Vogelvlei weir
7	Sonderend, Du Toits and Elandskloof Rivers upstream and inclusive of The Waterskloof Dam
8	Witte River to confluence with Breede River
9	Dwars River to Ceres divisional boundary
10	Olifants River to the Ceres divisional boundary
11	Helsloot and Smalblaar (or Molenaars) Rivers to their confluence with Breede River
12	Hex River to its confluence with Breede River
13	Van Stadens River to tidal water
14	Buffalo River from its source to where it enters the King Williams Town municipal area
15	Klipplaat River from its source to Waterdown Dam
16	Swart Kei River to its confluence with the Klipplaat River
17	Great Brak River
18	Bongola River to Bongola Dam
19	Kubusi River to the Stutterheim municipal boundary
20	Langkloof River from its source to Barkly East municipal boundary
21	Kraai River to its confluence with the Langkloof River
22	Little Tsomo River

	WATER RESOURCE
23	Xuka River to the Elliot district boundary
24	Tsitsa and Inxu Rivers to their confluence
25	Mvenyane and Mzimvubu Rivers from sources to their confluence
26	Mzintlava River to its confluence with the Mvalweni River
27	Ingwangwana River to its confluence with Umzimkulu River
28	Umzimkulu and Polela Rivers to their confluence
29	Elands River to the Pietermaritzburg-Bulwer main road
30	Umtamvuma and Weza Rivers to their confluence
31	Umkomaas and Isinga Rivers to their confluence
32	Lurane River to its confluence with the Umkomaas River
33	Sitnundjwana Spruit to its confluence with the Umkomaas River
34	Inudwini River to the Polela district boundary
35	Inkonza River to the bridge on the Donnybrook-Creighton road
36	Umlaas to the bridge on District Road 334 on the farm Maybole
37	Umgeni and Lions River to their confluence
38	Mooi River to the road bridge at Rosetta
39	Little Mooi and Hlatikula Rivers to their confluence
40	Bushmans River to Wagendrift Dam
41	Little Tugela River and Sterkspruit to their confluence
42	M'Lambonjwa and Mhlawazeni Rivers to their confluence
43	Mnweni and Sandhlwana Rivers to their confluence
44	Tugela River to its confluence with the Kombe Spruit
45	Inyamvubu (or Mnyamvubu) River to Craigie Burn Dam
46	Umvoti River to the bridge on the Seven Oaks-Rietvlei road
47	Yarrow River to its confluence with the Karkloof River
48	Incandu and Ncibidwane Rivers to their confluence
49	Ingogo River to its confluence with the Harte River
50	Pivaan River to its confluence with Soetmelkspruit
51	Slang River and the Wakkerstroom to their confluence
52	Elands and Swartkoppie Spruit to their confluence
53	All tributaries of the Komati River between Nooitgedacht Dam and its confluence with and including Zevenfontein Spruit
54	Seekoeispruit to its confluence with Buffelspruit
55	Crocodile River and Buffelskloofspruit to their confluence
56	All tributaries of the Steelpoort River down to its confluence with and including the Dwars River
57	Potspruit to its confluence with the Waterval River
58	Dorps River (or Spekboom River) to its confluence with the Marambanspruit
59	Ohrigstad River to the Ohrigstad Dam
60	Klein-Spekboom River to its confluence with the Spekboom River
61	Blyde River to the Pilgrim's Rest municipal boundary
62	Sabie River to the Sabie municipal boundary .
63	Nels River to the Pilgrim's Rest district boundary
64	Houtbosloop River to the Lydenburg district boundary
65	Blinkwaterspruit to Longmere Dam
66	Assegaai River upstream and inclusive of the Heyshope Dam
67	Komati River upstream and inclusive of the Nooitgedacht Dam and the Vygeboom Dam

	WATER RESOURCE		
68	Ngwempisi River upstream and inclusive of Jericho Dam and Morgenstond Dam		
69	Slang River upstream and inclusive of Zaaihoek Dam		
70	All streams flowing into the Olifants River upstream and inclusive of Loskop Dam, Witbank Dam and Middelburg Dam		
71	All streams flowing into Ebenezer Dam on the Great Letaba River		
72	Dokolewa River to its confluence with the Politzi River		
73	Ramadiepa River to the Merensky Dam on the farm Westfalia 223, Letaba		
74	Pienaars River and tributaries as far as Klipvoor Dam		
	RAMSAR LISTED WETLANDS:	PROVINCE	LOCATION
75	Barberspan	North-West	26°33' S 25°37' E
76	Blesbokspruit	Gauteng	26°17' S 28°30' E
77	De Hoop Vlei	Western Cape	34°27' S 20°20' E
78	De Mond (Heuningnes Estuary)	Western Cape	34°43' S 20°07' E
79	Kosi Bay	Kwazulu-Natal	27°01' S 32°48' E
80	Lake Sibaya	Kwazulu-Natal	27°20' S 32°38' E
81	Langebaan	Western Cape	33°06' S 18°01' E
82	Orange River Mouth	Northern Cape	28°40' S 16°30' E
83	St Lucia System	Kwazulu-Natal	28°00' S 32°28' E
84	Seekoeivlei Nature Reserve	Free State	27°34' S 29°35' E
85	Verlorenvlei	Western Cape	32°24' S 18°26' E
86	Verloren Valei	Mpumalanga	25°14' S 30°4' E
87	Nylsvlei	Northern	24°39' S 28°42' E
88	Wilderness Lakes	Western Cape	33°59' S 22°39' E

APPENDIX B

Table B1: Class-A-Pan Data, courtesy of Courtesy of the South African Weather Bureau.

YEAR	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC
1957	86.9	80.5	136.4	132.1	170.2	155.7	209	*	*	*	*	*
1958	179.6	169.7	178.6	136.9	113	95.5	91.7	128.3	142.2	206.8	184.4	194.3
1959	205	178.3	212.3	160.8	93.5	85.9	96	122.9	152.7	162.1	170.2	191.8
1960	214.9	159	169.4	106.9	92.7	77.2	88.4	102.4	149.9	144	176.3	186.7
1961	211.3	176	164.8	102.9	91.7	74.4	86.4	126.5	140.2	170.4	162.3	205.2
1962	187.7	169.7	171.2	141.5	111.3	89.9	107.2	133.6	146.3	146.6	160.5	193
1963	184.7	189.5	132.8	125.7	114.6	67.6	70.6	119.9	120.9	125	175.5	206
1964	179.3	184.7	175.8	128.5	94.5	83.6	90.9	122.9	121.7	89.2	155.4	191
1965	180.6	173.5	179.6	142.5	110.2	89.2	76.2	106.2	122.4	150.9	129.8	208.5
1966	183.9	164.8	223	116.1	95.2	75.9	109.5	121.4	154.4	175.3	177	206
1967	191.8	164.8	160.5	109	99.6	88.1	104.4	146.6	147.8	210.3	180.3	232.9
1968	195.5	196.6	143.6	134	111.7	110.4	89.9	104.7	135.3	148.6	166.6	210.3
1969	254.5	171.1	151.1	126.3	99.5	89.6	103.5	145.5	120.1	147.2	151.6	187
1970	205.8	190	235	152.4	96.5	102.2	106.6	119.2	139.4	156.9	142.2	213.2
1971	163.6	181.5	156.5	120.1	92.3	92.2	87.5	127.8	95	157.6	182.9	183.2
1972	243.3	171.4	196.6	143	80.6	80.3	95.9	112.1	139.4	171	196.6	205.9
1973	191.2	183.6	169.9	131	112.7	93.3	108.6	103.2	123.6	167.4	190.7	230.6
1974	198.2	164.2	178.9	139.4	128.3	86.6	116.2	146.9	171.4	172.6	177.7	187.5
1975	198.2	150.1	147.3	111	85.2	78.6	96.3	123.9	103.8	164.6	168.6	214.1
1976	249.7	208.4	214.7	144.8	125.7	132.7	149	164.1	165.3	200.3	208.6	*
1977	190.4	166.1	158.4	146.7	136.1	100.5	120.6	129.9	143.7	160.3	188.3	222.1
1978	200.9	157.2	168.2	133.5	109.6	91.1	88.1	84	128.9	159.2	166.4	228.5
1979	217.8	219.6	207.3	141.8	112.5	113.7	107.1	86.8	131.7	165.4	184.1	196.3
1980	265	209.1	197.6	165.5	127.2	104	103.7	131.4	130.5	191.1	196.6	276
1981	182.6	162.1	205	176.8	107.7	104.9	112.3	130.5	144.3	222.8	185.1	219.2
1982	193.9	210.3	196.2	141.9	124.2	99.2	121.1	153.2	174.3	192	225.9	258.4
1983	234.6	215.9	212.7	172.6	152.1	100.3	130.8	160.2	163.3	165.7	154.2	200
1984	192.7	183.7	155	156.4	117.9	85.4	87.5	171.7	148.8	153.6	182	221.7
1985	197.8	160.7	183.5	172	136	103.4	144	168.8	173.9	204.5	222	238.7

1986 1987	228.2 166.1	234.4 202.1	282.4 177.4	128.3 149.6	121.8 129.5	107.4 99.2	112 113	124.2 145	139.1 104.1	168.8 137.8	169.2 138.8	203.5 173.1
AVE	203	182.3	183.5	138.6	110.8	91.9	102.6	128.5	138.9	165.3	174.8	209.8
YEAR	AVE	:	1830									

Table B2: Evapotranspiration Data.

Month	#	Wind Velocity (m/s)	Temp. (^{0}C)	Relative Humidity (%)	Air Pressure (Kpa)	Net Radiation (W/m^2)	% Sun (%)	Class-A-Pan Evap. (mm/day)	C_T	C_W	C_H	C_S	ET (mm/day) Pan Fact.
J	1	4.44	24.58	80.69	101.318	412	54	6.55	1.020	0.907	1.118	0.949	4.85
F	2	4.51	24.69	82.24	101.447	392	52	6.51	1.020	0.907	1.125	0.946	4.84
M	3	4.27	24.35	80.88	101.518	372	52	5.92	1.019	0.907	1.119	0.946	4.37
A	4	4.27	22.58	78.49	101.821	353	50	4.62	1.012	0.907	1.106	0.943	3.34
M	5	3.45	19.47	75.75	101.788	328	44	3.57	0.997	0.919	1.092	0.935	2.52
J	6	3.14	17.14	71.42	102.087	285	42	3.06	0.985	0.929	1.068	0.932	2.11
J	7	3.85	17.65	72.9	102.27	260	42	3.31	0.988	0.911	1.077	0.932	2.26
A	8	4.07	18.29	72.44	102.11	306	46	4.15	0.991	0.908	1.074	0.938	2.84
S	9	5.12	19.13	75.2	101.838	363	48	4.63	0.996	0.915	1.089	0.940	3.26
O	10	5.51	20.17	76.08	101.919	426	50	5.33	1.001	0.926	1.094	0.943	3.85
N	11	4.9	22.29	78.59	101.539	381	52	5.83	1.011	0.911	1.107	0.946	4.24
D	12	4.68	23.61	81.11	101.416	376	54	6.77	1.016	0.908	1.120	0.949	5.01

Application Report AL 99005

Kat.-Nr. 418 395

Determining Biochemical Oxygen Demand (BOD) with BSB/BOD-Sensors

- manometric method[2] -

by Dipl.-Biol. Markus Robertz

Introduction

The Biochemical Oxygen Demand over a testing period of n days (**BOD_n**) is precisely defined and associated with experimental standards. It represents the quantity of oxygen aspirated in the course of aerobic breakdown of organic substances by microorganisms.

BOD is thus an substantial feature in determining the effect of discharged effluents on the oxygen content of a water-course or on the oxygen demand of an effluent treatment plant. BOD levels are stated in mg/l of oxygen and are usually measured over a period of 5 days (**BOD_5**).

Principle of Measurement

Microorganisms[1] feed on the organic compounds contained in a water sample, which they consume in the presence of oxygen (O_2) - that is, the compounds are biochemically oxidised and thus broken down, either partially or completely. The complete breakdown of organic materials (C_{org}) results in their oxidation to carbon dioxide (CO_2) and inorganic salts (mineralisation), as covered by expression 1:

$$(1)\quad C_{org} + O_2 \xrightarrow{microorganisms} CO_2 + Salts$$

The manometric method for BOD-determination[2] is based on the fact that the oxygen which is converted to carbon dioxide is removed from the gas phase of the sample by the use of potassium hydroxide KOH (HÜTTER, 1984). Therefore, in the closed system BOD-flask/BOD-sensor, a drop in pressure occurs, which is proportional to the amount of oxygen consumed.

BSB/BOD-Sensors

With BOD-sensors, the change in pressure resulting from the consumption of oxygen is measured in the flasks by electronic pressure sensors and calculated directly in terms of mg/l BOD (details are given in the instruction manual for the equipment).

[1] bacteria, fungus, archaea and protozoa
[2] to DIN 38409-H52

This method is outstandingly suitable for routine analysis work and provides a range of advantages in comparison to the dilution method for BOD determination[3]: the sample can usually be used without pre-dilution; individual measurement ranges are much wider; all measured values are stored automatically; the BOD-graph (see Fig. 1) is easily drawn up and there is considerably less work involved.

Selecting the Measurement Range

The BOD value of a sample depends on the level of bio-available organic substances contained. The range of the measurement system should be selected to ensure that the expected readings will be roughly within the upper half of the scale. Thus, where BOD values of 250 mg/l are expected, the range 0-400 mg/l would be ideal (see Table 1). For samples where BOD-values are unknown, they could be estimated by taking 80 % of the COD[4] level as the *maximum* BOD value. **Important!** Note that, if the measurement range is exceeded, no BOD-(end-) value will be obtained! However, the individual daily values may be used to make an estimate of the final figure (see Fig. 1).

Table 1: Measurement ranges with the associated sample volumes and the required amount of nitrification inhibitor (ATH) from AQUALYTIC®.

measurement range mg/l BOD	sample volume ml	ATH drops
0 - 40	428	10
0 - 80	360	10
0 - 200	244	5
0 - 400	157	5
0 - 800	94	3
0 - 2000	56	3
0 - 4000	21,7	1

The BOD-values for samples with a BOD in excess of 4000 mg/l[5] can be determined by pre-treatment with the use of so-called dilution water (see AQUALYTIC® Application Report).

[3] to DIN 38409-H51
[4] Chemical Oxygen Demand (COD)
[5] reserve of measurement range till 5000 mg/l BOD

Preparing the Sample

- **pH value** of the sample: for biochemical oxidation the most suitable pH value is between pH 6 and pH 9. If the pH value of a sample is outside this range, it should be set within range, since any greater deviation results in an underestimation of the BOD value. Too high, a pH value can be reduced with 1-N-sulphuric acid, while too low, a pH value can be increased with 1-N-sodium hydroxide solution.
- **Homogenisation:** the sample should be homogenised or pre-treated to any special requirements for obtaining the total BOD of a sample, including contained particles. Comparable BOD values can only be obtained if the pre-treatment of each sample is carried out similar.
- **Volume** of the homogenised sample: the sample volume can be determined from Table 1, depending on the measurement range required. It can then be measured precisely, using the relevant overflow vessel and poured into the sample flask. We recommend that three, or at the very least, two determinations should be made for each sample. **Important!** Note that the BOD sensors are fitted to operate with the sample volumes given in Table 1 and with the BOD-flasks of AQUALYTIC®. The use of other volumes and/or flasks may lead to inaccuracies in measurement
- **Inhibiting of nitrification:** to suppress this considerable source of irritation, the nitrification inhibitor N-allylthiourea (ATH) from AQUALYTIC® should be added in drops to the sample, as detailed in Table 1.

Nitrification is caused by two groups of nitrifying bacteria: the first group oxidises ammonium (NH_4^+) to nitrite (NO_2^-), representing the substrate for the second group, which forms nitrate (NO_3^-): see expression 2:

$$(2)\quad NH_4^+ \xrightarrow{+1\frac{1}{2}O_2} NO_2^- \xrightarrow{+\frac{1}{2}O_2} NO_3^-$$

This conversion requires 4.57 mg/l O_2 per mg of NH_4^+ and has a significant effect on the BOD, which is intended to determine only the oxygen consumed in the course of carbon oxidation (C-BOD).

- **Sealing the sample flasks:** to ensure correct gas exchange by agitation during the incubation period, a magnetic stirring rod[6] from AQUALYTIC® must be inserted into the sample. A dry, grease-free gasket is filled with two drops of potassium hydroxide solution from AQUALYTIC® and inserted into the neck of the flask. The vessel is then sealed by screwing a BOD-sensor onto the BOD-bottle.
- **Tempering the sample**: the Auto-Start-Function[7] allows to use the sample without pre-tempering, provided the sample temperature is not more than 5°C below the incubation temperature selected (generally 20°C). **Important!** To eliminate artificially high readings, samples which are warmer than the selected incubation temperature need to be cooled down before starting the measurement! Thus, where the selected incubation temperature is 20°C, samples which are warmer than 20°C must be cooled and samples cooler than 15°C should be heated to between 15°C and 20°C. This can be achieved, for example, by placing the sample vessels in a tempered water bath.

Starting & Evaluating Measurements

The process is started as described in the operating instructions for the equipment. The sample is then incubated in a thermostatically controlled cabinet for the selected incubation period (5 days in the case of BOD_5 measurement) and at the selected incubation temperature (generally 20°C). The sample is agitated constantly in order to ensure oxygen delivery from the gas phase of the measurement system into the water sample, in which oxygen is consumed. **Important!** The incubation temperature ($T_{ink.}$) must be maintained within the range of T_{ink} ± 1 °C - otherwise, errors of up to 10 % BOD per 1 °C can occur!

The BOD value is determined as described in the instruction manual for the equipment. Should slight deviations occur within the parallel samples (normally < 10 %), then usual the mean value of measurements is taken.

Cleaning

We recommend repeated rinsing in hot water to clean every item which get in contact with a sample, to prevent contamination by materials such as tensides[8] which would affect the BOD measurement. In the case of severe contamination a cleaning agent should be used; the equipment must then be rinsed very thoroughly with distilled water.

Advice on Evaluation of Results

- BOD values do not increase in a linear manner; after a day they must always be higher than on the previous day but the

[6] of defined volume

[7] for details see instruction manual of the device

[8] cleaning agents

daily increase in mg/l BOD becomes ever smaller (see Fig. 1).

- if BOD readings become linear, the sample is outside the measurement range (overflow). To obtain BOD values, a higher measurement range must be chosen.
- if BOD readings suddenly increase during the measurement period, it is possible that nitrification has started (see above).
- if BOD readings fall in the course of measurement, the system may have developed a leak, or the sample material has become problematic (for example, anaerobiosis).

Interpretation

BOD_n values can be used to reach conclusions regarding the characteristics of a water body, as well as the biological activity of the incubated microflora. For example, the introduction of effluents with a high level of oxygen consumption (high BOD value) can lead to an oxygen starvation of the water-course (fish killing). In an other case, the performance of an effluent treatment plant can be checked by comparing the BOD levels before and after an effluent treatment.

In general, the following conclusions may be drawn:

- high BOD reading indicates a high content of biodegradable organic materials in the sample - in other words, without further pretreatment, this sample will cause stress on the oxygen level of a water course.
- a low BOD reading in the sample indicates either a low content of organic materials (that means low stress on the oxygen level of a water course), or substances which are difficult to break down, or various functional problems (the sample may contain poisons or inhibiting substances, or have an extremely high pH, etc.). This can be evaluated in detail by the comparison with the results of other analyses, as explained below.
- the BOD graph (see Fig. 1) provides further information on the significance of the measurement (conformance with the measurement range; errors; kinetics of the biological degradation process).

The BOD gains informative value if evaluated in association with other parameters, such as COD, DOC, POC, TOC. An example is provided by comparing the obtained BOD value with the corresponding COD value:

- a small difference indicates that a large proportion of the organic substances can be broken down.
- a large difference suggests either that the organic substances are not easily biodegradable, or that there is an error.

Note

The comments and explanations set out in this paper refer to regular samples and conventional reactions of microorganisms in the course of a BOD measurement and cover the majority of all samples. Thus, this method is used with success and without problems in practically all municipal effluent treatment plants.

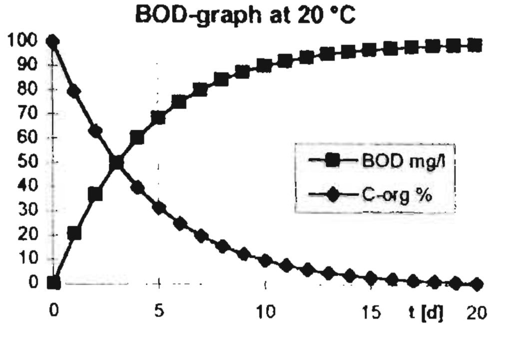

Fig. 1: Idealised BOD-graph at 20 °C (to HABECK-TROPFKE, 1992) compared with the proportional reduction in biodegradable organic compounds (C_{org}). After 5 days incubation, approx. 70 % of the C_{org} has been broken down: this is the equivalent of the BOD_5 value.

Special cases are always a possibility, however, and arise from specific, local circumstances. For example, therefore, underestimated BOD values might be the result of a severe inhibition, or the presence of certain, disturbing constituents in the sample, or maybe even the result of special effluent treatment processes in front of the site the sample was taken from. Extreme conditions are frequently encountered with industrial effluents. They often contain very high or very low BOD loadings, as well as oxidising or toxic materials. Cases of this kind must be analysed with care and the problems which arise must be treated on an individual basis (please ask for our special AQUALYTIC® Application Reports).

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APPENDIX C2

MICROTOX (Promotion and training video)

Microbics (UK)LTD
The Coach House
24a Tilehouse Street
Hitchin
Hertfordshire SG5 20V
Tel: (0462) 442210
Fax: (0462) 440880

Introduction

The Microtox test is aimed at trying to find the concentration of pollutant at which the light emitted by the *Photobacterium phosphorium* is reduced by half. This concentration is called the EC 50 (e.g. EC 50 of 1 ppm indicates that the sample is quite toxic). The test has been used world wide to monitor wastewater treatment plants.

Reagents

- Microtox reagent (stored at –20 °C) contains a freeze-dried pellet of microorganisms that emit light as a byproduct of their respiration. When the microorganisms are reconstituted with water they emit light; if a toxic sample harms them the light output drops.
- Reconstitution solution is specially processed pure water (non-toxic).
- Microtox dilution solution is a non-toxic, 2% sodium chloride solution, used to dilute samples.
- Microtox osmotic adjustment solution is a 22% sodium chloride solution. This is added to the solution, as the *Photobacterium phosphorium* microorganisms are marine organisms.

Standard Microtox test procedure

(For the model 500 apparatus shown in Figure C1)

- Place clean unused cuvettes into wells A1 to A5, B1 to B5 and reagent well.
- Add 1 ml of reconstitution solution into the reagent well.
- Add 0.5 ml Microtox dilution solution into cuvettes B1 to B5.

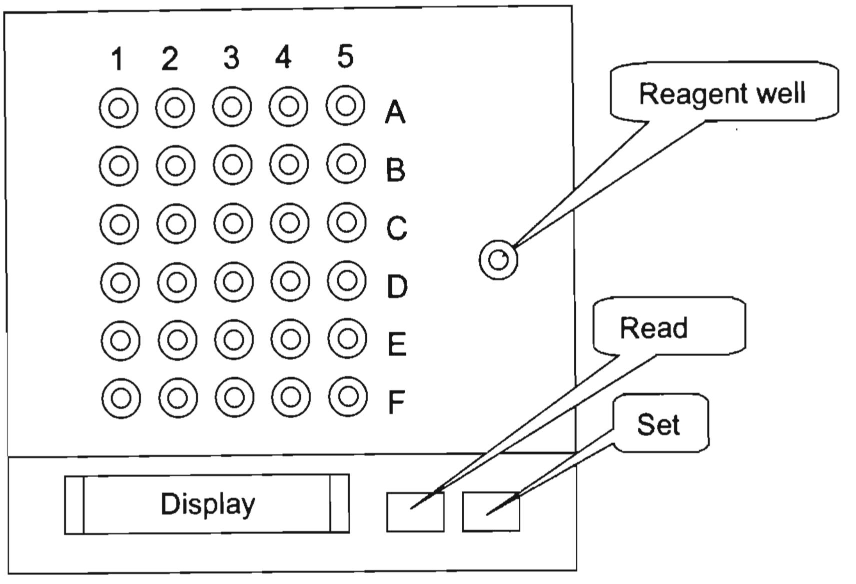

Figure C1: Model 500 apparatus.

- Add 1 ml Microtox dilution solution into cuvettes A1 to A4 and leave A5 empty.
- Add 0.25 ml Microtox osmotic adjustment solution to well A5.
- Add 2.5 ml of test sample into A5.
- Transfer 1 ml from A5 to A4.
- Transfer 1 ml from A4 to A3.
- Transfer 1 ml from A3 to A2.
- Wait approximately 5 minutes for temperature equilibrium in incubator block at 15 °C.
- Remove Microtox reagent from freezer and open.
- Pour 1ml of reconstitution solution, which had been stored at 5.5 °C in the reagent well, into the freshly opened bottle, mix, and pour back into the reagent cuvette and replace in the reagent well.
- Transfer 0.01 ml of reconstituted Microtox reagent to B1 to B5.
- Use B1 to set (calibrate for that reagent) and then read zero readings for B1 to B5 before adding samples.
- Add 0.5 ml from A1 to B1; A2 to B2; A3 to B3; A4 to B4 and A5 to B5.
- 5 minutes after adding sample, read the light emitted and do this again after 15 minutes.

Results

Usually the results are data logged and processed by a computer, which produces a print out after the analyses.

APPENDIX D1

Table D1: Rainfall data for the treatability trials.

Month	#	Durban Area (mm/month)	Bisasar (mm/month)
Jan	1	134.0	105.5
Feb	2	113.0	71.5
March	3	120.0	39.0
April	4	73.0	59.5
May	5	59.0	4.0
June	6	28.0	0.0
July	7	39.0	63.0
Aug	8	62.0	2.0

APPENDIX D2

Table D2: Control flow data

DATE	#	INFLUENT (l/d)	RAINFALL (mm/d)	TOTAL INFLUENT (l/d)	EFFLUENT (l/d)	TOTAL LOSS (l/d)	LOSS (mm/d)	AVERAGE INFLUENT (l/d)	AVERAGE RAINFALL (l/d)	AVERAGE EFFLUENT (l/d)
4-Jan-01	1	20	0.00	20.00	11.24	8.76	2.74			
5-Jan-01	2	20	0.00	20.00	11.10	8.90	2.78			
6-Jan-01	3	20	0.00	20.00	10.46	9.54	2.98			
7-Jan-01	4	20	0.00	20.00	8.50	11.50	3.59			
8-Jan-01	5	20	0.00	20.00	7.74	12.26	3.83	20.00	0.00	9.81
9-Jan-01	6	20	0.00	20.00	10.00	10.00	3.13			
10-Jan-01	7	20	0.00	20.00	11.60	8.40	2.63			
11-Jan-01	8	20	0.00	20.00	11.24	8.76	2.74	20.00	0.00	10.95
12-Jan-01	9	20	0.00	20.00	11.32	8.68	2.71			
13-Jan-01	10	20	0.00	20.00	11.50	8.50	2.66			
14-Jan-01	11	20	3.00	29.60	15.30	14.30	4.47			
15-Jan-01	12	20	0.50	21.60	13.14	8.46	2.64	20.00	2.80	12.82
16-Jan-01	13	20	0.00	20.00	13.08	6.92	2.16			
17-Jan-01	14	20	0.00	20.00	11.08	8.92	2.79			
18-Jan-01	15	20	0.00	20.00	8.94	11.06	3.46	20.00	0.00	11.03
19-Jan-01	16	20	0.00	20.00	10.30	9.70	3.03			
20-Jan-01	17	20	0.00	20.00	12.54	7.46	2.33			
21-Jan-01	18	20	1.00	23.20	15.76	7.44	2.33			
22-Jan-01	19	20	0.00	20.00	13.72	6.28	1.96	20.00	0.80	13.08
23-Jan-01	20	20	0.00	20.00	10.00	10.00	3.13			
24-Jan-01	21	20	0.00	20.00	12.30	7.70	2.41			
25-Jan-01	22	20	0.00	20.00	10.12	9.88	3.09	20.00	0.00	10.81
26-Jan-01	23	20	0.00	20.00	10.48	9.52	2.98			
27-Jan-01	24	20	0.00	20.00	11.84	8.16	2.55			
28-Jan-01	25	20	0.00	20.00	12.62	7.38	2.31			
29-Jan-01	26	20	0.00	20.00	9.80	10.20	3.19	20.00	0.00	11.19
30-Jan-01	27	20	12.00	58.40	48.40	10.00	3.13			
31-Jan-01	28	20	0.00	20.00	15.10	4.90	1.53			

Total		560.00	16.50	612.80	359.22	253.58	79.24			
Average		20.00	0.59	21.89	12.83	9.06	2.83			

1-Feb-01	29	20	6.00	39.20	18.50	20.70	6.47	20.00	19.20	27.33
2-Feb-01	30	20	0.00	20.00	17.86	2.14	0.67			
3-Feb-01	31	20	23.00	93.60	83.60	10.00	3.13			
4-Feb-01	32	20	2.00	26.40	17.34	9.06	2.83			
5-Feb-01	33	20	0.00	20.00	10.54	9.46	2.96	20.00	20.00	32.34
6-Feb-01	34	20	0.00	20.00	10.24	9.76	3.05			
7-Feb-01	35	20	0.00	20.00	9.70	10.30	3.22			
8-Feb-01	36	20	0.00	20.00	9.10	10.90	3.41			
9-Feb-01	37	20	2.50	28.00	9.44	18.56	5.80			
10-Feb-01	38	20	0.00	20.00	16.22	3.78	1.18			
11-Feb-01	39	20	0.00	20.00	12.00	8.00	2.50			
12-Feb-01	40	20	0.00	20.00	5.74	14.26	4.46	20.00	1.14	10.35
13-Feb-01	41	20	10.00	52.00	42.00	10.00	3.13			
14-Feb-01	42	20	8.00	45.60	35.60	10.00	3.13			
15-Feb-01	43	20	0.00	20.00	11.20	8.80	2.75			
16-Feb-01	44	20	0.00	20.00	12.92	7.08	2.21			
17-Feb-01	45	20	0.00	20.00	16.36	3.64	1.14			
18-Feb-01	46	20	18.00	77.60	67.60	10.00	3.13			
19-Feb-01	47	20	0.00	20.00	18.70	1.30	0.41			
20-Feb-01	48	20	0.00	20.00	15.20	4.80	1.50			
21-Feb-01	49	20	0.00	20.00	16.30	3.70	1.16			
22-Feb-01	50	20	0.00	20.00	10.54	9.46	2.96	20.00	11.52	24.64
23-Feb-01	51	20	0.00	20.00	17.02	2.98	0.93			
24-Feb-01	52	20	2.00	26.40	19.70	6.70	2.09			
25-Feb-01	53	20	0.00	20.00	13.80	6.20	1.94			
26-Feb-01	54	20	0.00	20.00	11.64	8.36	2.61	20.00	1.60	15.54
27-Feb-01	55	20	0.00	20.00	12.60	7.40	2.31			
28-Feb-01	56	20	0.00	20.00	14.00	6.00	1.88			
Total		560.00	71.50	788.80	555.46	233.34	72.92			
Average		20.00	2.55	28.17	19.84	8.33	2.60			

1-Mar-01	57	20	0.00	20.00	13.60	6.40	2.00			
2-Mar-01	58	20	0.00	20.00	12.16	7.84	2.45			
3-Mar-01	59	20	0.00	20.00	11.78	8.22	2.57			
4-Mar-01	60	20	0.00	20.00	12.00	8.00	2.50			
5-Mar-01	61	20	0.00	20.00	11.10	8.90	2.78			
6-Mar-01	62	20	0.00	20.00	11.24	8.76	2.74	20.00	0.00	12.31
7-Mar-01	63	20	0.00	20.00	11.54	8.46	2.64			
8-Mar-01	64	20	0.00	20.00	13.54	6.46	2.02			
9-Mar-01	65	20	0.00	20.00	10.00	10.00	3.13			
10-Mar-01	66	20	0.00	20.00	10.92	9.08	2.84			
11-Mar-01	67	20	0.00	20.00	8.50	11.50	3.59			
12-Mar-01	68	20	4.50	34.40	15.58	18.82	5.88			
13-Mar-01	69	20	0.00	20.00	13.08	6.92	2.16			
14-Mar-01	70	20	0.00	20.00	13.86	6.14	1.92			
15-Mar-01	71	20	0.00	20.00	9.76	10.24	3.20			
16-Mar-01	72	20	0.00	20.00	10.70	9.30	2.91			
17-Mar-01	73	20	6.50	40.80	30.80	10.00	3.13			
18-Mar-01	74	20	18.00	77.60	67.60	10.00	3.13			
19-Mar-01	75	20	3.00	29.60	24.38	5.22	1.63			
20-Mar-01	76	20	0.00	20.00	13.76	6.24	1.95			
21-Mar-01	77	20	1.00	23.20	16.20	7.00	2.19			
22-Mar-01	78	20	0.00	20.00	16.00	4.00	1.25			
23-Mar-01	79	20	2.00	26.40	18.74	7.66	2.39			
24-Mar-01	80	20	4.00	32.80	20.44	12.36	3.86			
25-Mar-01	81	20	0.00	20.00	13.40	6.60	2.06			
26-Mar-01	82	20	0.00	20.00	11.50	8.50	2.66	20.00	6.24	17.52
27-Mar-01	83	20	0.00	20.00	14.00	6.00	1.88			
28-Mar-01	84	20	0.00	20.00	14.00	6.00	1.88			
29-Mar-01	85	20	0.00	20.00	13.02	6.98	2.18			
30-Mar-01	86	20	0.00	20.00	12.30	7.70	2.41			
31-Mar-01	87	20	0.00	20.00	11.72	8.28	2.59			
Total		560.00	39.00	684.80	449.68	235.12	73.48			
Average		20.00	1.39	24.46	16.06	8.40	2.62			

1-Apr-01	88	20	16.00	71.20	61.20	10.00	3.13			
2-Apr-01	89	20	3.00	29.60	19.16	10.44	3.26			
3-Apr-01	90	20	1.50	24.80	19.82	4.98	1.56			
4-Apr-01	91	20	13.00	61.60	51.60	10.00	3.13			
5-Apr-01	92	20	5.00	36.00	21.50	14.50	4.53			
6-Apr-01	93	20	0.00	20.00	16.76	3.24	1.01			
7-Apr-01	94	20	0.00	20.00	11.04	8.96	2.80			
8-Apr-01	95	20	7.00	42.40	32.40	10.00	3.13			
9-Apr-01	96	20	15.00	68.00	58.00	10.00	3.13			
10-Apr-01	97	20	4.00	32.80	22.52	10.28	3.21			
11-Apr-01	98	20	0.00	20.00	14.82	5.18	1.62			
12-Apr-01	99	20	0.00	20.00	12.92	7.08	2.21			
13-Apr-01	100	20	0.00	20.00	14.30	5.70	1.78			
14-Apr-01	101	20	0.50	21.60	12.84	8.76	2.74			
15-Apr-01	102	20	0.00	20.00	14.80	5.20	1.63			
16-Apr-01	103	20	0.00	20.00	16.60	3.40	1.06			
17-Apr-01	104	20	0.00	20.00	15.40	4.60	1.44			
18-Apr-01	105	20	0.00	20.00	11.30	8.70	2.72	20.00	9.04	21.39
19-Apr-01	106	20	0.00	20.00	13.60	6.40	2.00			
20-Apr-01	107	20	0.00	20.00	12.52	7.48	2.34			
21-Apr-01	108	20	1.50	24.80	13.56	11.24	3.51			
22-Apr-01	109	20	1.50	24.80	15.06	9.74	3.04			
23-Apr-01	110	20	0.00	20.00	16.30	3.70	1.16			
24-Apr-01	111	20	0.00	20.00	15.44	4.56	1.43			
25-Apr-01	112	20	0.00	20.00	15.50	4.50	1.41			
26-Apr-01	113	20	6.00	39.20	29.20	10.00	3.13			
27-Apr-01	114	20	2.00	26.40	25.00	1.40	0.44			
28-Apr-01	115	20	0.00	20.00	17.00	3.00	0.94			
29-Apr-01	116	20	0.00	20.00	14.54	5.46	1.71			
30-Apr-01	117	20	4.00	32.80	24.70	8.10	2.53			
Total		540.00	59.50	730.40	539.22	191.18	59.74			
Average		20.00	2.20	27.05	19.97	7.08	2.21			

1-May-01	118	20	3.00	29.60	21.46	8.14	2.54			
2-May-01	119	20	4.00	32.80	25.00	7.80	2.44			
3-May-01	120	20	0.00	20.00	19.84	0.16	0.05			
4-May-01	121	20	0.00	20.00	13.70	6.30	1.97			
5-May-01	122	20	4.00	32.80	25.00	7.80	2.44			
6-May-01	123	20	0.00	20.00	15.88	4.12	1.29			
7-May-01	124	20	0.00	20.00	16.20	3.80	1.19			
8-May-01	125	20	0.00	20.00	8.00	12.00	3.75	20.00	4.16	17.88
9-May-01	126	20	0.00	20.00	14.40	5.60	1.75			
10-May-01	127	20	0.00	20.00	14.60	5.40	1.69			
11-May-01	128	20	0.00	20.00	16.52	3.48	1.09			
12-May-01	129	20	0.00	20.00	15.00	5.00	1.56			
13-May-01	130	20	0.00	20.00	13.82	6.18	1.93			
14-May-01	131	20	0.00	20.00	16.90	3.10	0.97			
15-May-01	132	20	0.00	20.00	14.88	5.12	1.60			
16-May-01	133	20	0.00	20.00	16.20	3.80	1.19			
17-May-01	134	20	0.00	20.00	14.00	6.00	1.88			
18-May-01	135	20	0.00	20.00	14.98	5.02	1.57			
19-May-01	136	20	0.00	20.00	15.12	4.88	1.53			
20-May-01	137	20	0.00	20.00	16.40	3.60	1.13			
21-May-01	138	20	0.00	20.00	10.80	9.20	2.88	20.00	0.00	14.89
22-May-01	139	20	0.00	20.00	16.40	3.60	1.13			
23-May-01	140	20	0.00	20.00	16.40	3.60	1.13			
24-May-01	141	20	0.00	20.00	16.70	3.30	1.03			
25-May-01	142	20	0.00	20.00	15.50	4.50	1.41			
26-May-01	143	20	0.00	20.00	15.60	4.40	1.38			
27-May-01	144	20	0.00	20.00	15.40	4.60	1.44			
28-May-01	145	20	0.00	20.00	16.90	3.10	0.97			
29-May-01	146	20	0.00	20.00	15.32	4.68	1.46			
30-May-01	147	20	0.00	20.00	15.46	4.54	1.42			
31-May-01	148	20	0.00	20.00	15.40	4.60	1.44			
Total		560.00	4.00	572.80	431.48	141.32	44.16			
Average		20.00	0.14	20.46	15.41	5.05	1.58			

1-Jun-01	149	20	0.00	20.00	18.24	1.76	0.55			
2-Jun-01	150	20	0.00	20.00	16.24	3.76	1.18			
3-Jun-01	151	20	0.00	20.00	15.80	4.20	1.31			
4-Jun-01	152	20	0.00	20.00	12.40	7.60	2.38	20.00	0.00	15.84
5-Jun-01	153	20	0.00	20.00	15.30	4.70	1.47			
6-Jun-01	154	20	0.00	20.00	16.30	3.70	1.16			
7-Jun-01	155	20	0.00	20.00	15.26	4.74	1.48			
8-Jun-01	156	20	0.00	20.00	16.00	4.00	1.25			
9-Jun-01	157	20	0.00	20.00	17.44	2.56	0.80			
10-Jun-01	158	20	0.00	20.00	17.04	2.96	0.93			
11-Jun-01	159	20	0.00	20.00	15.64	4.36	1.36			
12-Jun-01	160	20	0.00	20.00	16.86	3.14	0.98			
13-Jun-01	161	20	0.00	20.00	16.32	3.68	1.15			
14-Jun-01	162	20	0.00	20.00	16.68	3.32	1.04			
15-Jun-01	163	20	0.00	20.00	15.46	4.54	1.42			
16-Jun-01	164	20	0.00	20.00	16.10	3.90	1.22			
17-Jun-01	165	20	0.00	20.00	15.44	4.56	1.43			
18-Jun-01	166	20	0.00	20.00	15.40	4.60	1.44	20.00	0.00	16.09
19-Jun-01	167	20	0.00	20.00	15.64	4.36	1.36			
20-Jun-01	168	20	0.00	20.00	15.24	4.76	1.49			
21-Jun-01	169	20	0.00	20.00	16.00	4.00	1.25			
22-Jun-01	170	20	0.00	20.00	16.70	3.30	1.03			
23-Jun-01	171	20	0.00	20.00	16.20	3.80	1.19			
24-Jun-01	172	20	0.00	20.00	15.60	4.40	1.38			
25-Jun-01	173	20	0.00	20.00	16.00	4.00	1.25			
26-Jun-01	174	20	0.00	20.00	16.30	3.70	1.16			
27-Jun-01	175	20	0.00	20.00	17.50	2.50	0.78			
28-Jun-01	176	20	0.00	20.00	17.20	2.80	0.88			
29-Jun-01	177	20	0.00	20.00	16.70	3.30	1.03			
30-Jun-01	178	20	0.00	20.00	17.12	2.88	0.90			
Total		540.00	0.00	540.00	433.84	106.16	33.18			
Average		20.00	0.00	20.00	16.07	3.93	1.23			

1-Jul-01	179	20	0.00	20.00	15.40	4.60	1.44			
2-Jul-01	180	20	0.00	20.00	13.10	6.90	2.16	20.00	0.00	16.05
3-Jul-01	181	20	0.00	20.00	14.84	5.16	1.61			
4-Jul-01	182	20	0.00	20.00	15.62	4.38	1.37			
5-Jul-01	183	20	0.00	20.00	15.06	4.94	1.54			
6-Jul-01	184	20	0.00	20.00	16.42	3.58	1.12			
7-Jul-01	185	20	0.00	20.00	16.62	3.38	1.06			
8-Jul-01	186	20	0.00	20.00	15.94	4.06	1.27			
9-Jul-01	187	20	0.00	20.00	15.80	4.20	1.31			
10-Jul-01	188	20	0.00	20.00	15.90	4.10	1.28			
11-Jul-01	189	20	0.00	20.00	16.70	3.30	1.03			
12-Jul-01	190	20	0.00	20.00	16.80	3.20	1.00			
13-Jul-01	191	20	0.00	20.00	16.10	3.90	1.22			
14-Jul-01	192	20	0.00	20.00	16.84	3.16	0.99			
15-Jul-01	193	20	0.00	20.00	16.68	3.32	1.04			
16-Jul-01	194	20	0.00	20.00	14.00	6.00	1.88	20.00	0.00	15.95
17-Jul-01	195	20	0.00	20.00	17.96	2.04	0.64			
18-Jul-01	196	20	0.00	20.00	17.60	2.40	0.75			
19-Jul-01	197	20	0.00	20.00	16.70	3.30	1.03			
20-Jul-01	198	20	0.00	20.00	14.98	5.02	1.57			
21-Jul-01	199	20	1.00	23.20	16.82	6.38	1.99			
22-Jul-01	200	20	0.00	20.00	15.30	4.70	1.47			
23-Jul-01	201	20	0.00	20.00	15.90	4.10	1.28			
24-Jul-01	202	20	0.00	20.00	14.00	6.00	1.88			
25-Jul-01	203	20	0.00	20.00	15.10	4.90	1.53			
26-Jul-01	204	20	0.00	20.00	17.70	2.30	0.72			
27-Jul-01	205	20	0.00	20.00	15.06	4.94	1.54			
28-Jul-01	206	20	0.00	20.00	15.62	4.38	1.37			
29-Jul-01	207	20	50.00	180.00	170.00	10.00	3.13			
30-Jul-01	208	20	12.00	58.40	48.40	10.00	3.13			
31-Jul-01	209	20	0.00	20.00	12.50	7.50	2.34	20.00	13.44	28.24
Total		560.00	63.00	761.60	632.12	129.48	40.46			
Average		20.00	2.25	27.20	22.58	4.62	1.45			

1-Aug-01	210	20	0.00	20.00	19.40	0.60	0.19			
2-Aug-01	211	20	0.00	20.00	17.20	2.80	0.88			
3-Aug-01	212	20	0.00	20.00	15.60	4.40	1.38			
4-Aug-01	213	20	0.00	20.00	16.70	3.30	1.03			
5-Aug-01	214	20	1.00	23.20	21.40	1.80	0.56			
6-Aug-01	215	20	1.00	23.20	17.30	5.90	1.84			
7-Aug-01	216	20	0.00	20.00	15.50	4.50	1.41			
8-Aug-01	217	20	0.00	20.00	14.62	5.38	1.68			
9-Aug-01	218	20	0.00	20.00	14.60	5.40	1.69			
10-Aug-01	219	20	0.00	20.00	14.50	5.50	1.72			
11-Aug-01	220	20	0.00	20.00	12.34	7.66	2.39			
12-Aug-01	221	20	0.00	20.00	13.80	6.20	1.94			
13-Aug-01	222	20	0.00	20.00	12.70	7.30	2.28	20.00	0.49	15.82
14-Aug-01	223	20	0.00	20.00	15.22	4.78	1.49			
15-Aug-01	224	20	0.00	20.00	16.96	3.04	0.95			
16-Aug-01	225	20	0.00	20.00	16.56	3.44	1.08			
17-Aug-01	226	20	0.00	20.00	17.06	2.94	0.92			
18-Aug-01	227	20	0.00	20.00	14.00	6.00	1.88			
19-Aug-01	228	20	0.00	20.00	14.32	5.68	1.78			
20-Aug-01	229	20	0.00	20.00	14.10	5.90	1.84			
21-Aug-01	230	20	0.00	20.00	15.02	4.98	1.56			
22-Aug-01	231	20	0.00	20.00	13.72	6.28	1.96			
23-Aug-01	232	20	0.00	20.00	13.62	6.38	1.99			
24-Aug-01	233	20	0.00	20.00	14.84	5.16	1.61			
25-Aug-01	234	20	0.00	20.00	14.72	5.28	1.65			
26-Aug-01	235	20	0.00	20.00	13.40	6.60	2.06			
27-Aug-01	236	20	0.00	20.00	14.44	5.56	1.74			
28-Aug-01	237	20	0.00	20.00	15.92	4.08	1.28			
29-Aug-01	238	20	0.00	20.00	14.00	6.00	1.88			
30-Aug-01	239	20	0.00	20.00	12.40	7.60	2.38	20.00	0.00	14.72
31-Aug-01	240	20	0.00	20.00	18.00	2.00	0.63			
Total		560.00	2.00	566.40	421.76	144.64	45.20			
Average		20.00	0.07	20.23	15.06	5.17	1.61			

Table D3: *Leersia hexandra* flow data.

DATE	#	INFLUENT (l/d)	RAINFALL (mm/d)	TOTAL INFLUENT (l/d)	EFFLUENT (l/d)	TOTAL LOSS (l/d)	LOSS (mm/d)	AVERAGE INFLUENT (l/d)	AVERAGE RAINFALL (l/d)	AVERAGE EFFLUENT (l/d)
4-Jan-01	1	20	0.00	20.00	7.36	12.64	3.95			
5-Jan-01	2	20	0.00	20.00	8.40	11.60	3.63			
6-Jan-01	3	20	0.00	20.00	9.00	11.00	3.44			
7-Jan-01	4	20	0.00	20.00	6.96	13.04	4.08			
8-Jan-01	5	20	0.00	20.00	5.30	14.70	4.59	20.00	0.00	7.40
9-Jan-01	6	20	0.00	20.00	6.20	13.80	4.31			
10-Jan-01	7	20	0.00	20.00	7.24	12.76	3.99			
11-Jan-01	8	20	0.00	20.00	5.82	14.18	4.43	20.00	0.00	6.42
12-Jan-01	9	20	0.00	20.00	6.42	13.58	4.24			
13-Jan-01	10	20	0.00	20.00	5.98	14.02	4.38			
14-Jan-01	11	20	3.00	29.60	11.10	18.50	5.78			
15-Jan-01	12	20	0.50	21.60	13.14	8.46	2.64	20.00	2.80	9.16
16-Jan-01	13	20	0.00	20.00	10.34	9.66	3.02			
17-Jan-01	14	20	0.00	20.00	9.26	10.74	3.36			
18-Jan-01	15	20	0.00	20.00	6.74	13.26	4.14	20.00	0.00	8.78
19-Jan-01	16	20	0.00	20.00	8.00	12.00	3.75			
20-Jan-01	17	20	0.00	20.00	11.66	8.34	2.61			
21-Jan-01	18	20	1.00	23.20	11.90	11.30	3.53			
22-Jan-01	19	20	0.00	20.00	14.82	5.18	1.62	20.00	0.80	11.60
23-Jan-01	20	20	0.00	20.00	9.06	10.94	3.42			
24-Jan-01	21	20	0.00	20.00	9.84	10.16	3.18			
25-Jan-01	22	20	0.00	20.00	9.48	10.52	3.29	20.00	0.00	9.46
26-Jan-01	23	20	0.00	20.00	10.04	9.96	3.11			
27-Jan-01	24	20	0.00	20.00	11.22	8.78	2.74			
28-Jan-01	25	20	0.00	20.00	10.84	9.16	2.86			
29-Jan-01	26	20	0.00	20.00	8.42	11.58	3.62	20.00	0.00	10.13
30-Jan-01	27	20	12.00	58.40	48.40	10.00	3.13			
31-Jan-01	28	20	0.00	20.00	15.32	4.68	1.46			
Total		560.00	16.50	612.80	298.26	314.54	98.29			
Average		20.00	0.59	21.89	10.65	11.23	3.51			

1-Feb-01	29	20	6.00	39.20	16.96	22.24	6.95	20.00	19.20	26.89
2-Feb-01	30	20	0.00	20.00	16.56	3.44	1.08			
3-Feb-01	31	20	23.00	93.60	83.60	10.00	3.13			
4-Feb-01	32	20	2.00	26.40	15.90	10.50	3.28			
5-Feb-01	33	20	0.00	20.00	9.94	10.06	3.14	20.00	20.00	31.50
6-Feb-01	34	20	0.00	20.00	8.94	11.06	3.46			
7-Feb-01	35	20	0.00	20.00	8.80	11.20	3.50			
8-Feb-01	36	20	0.00	20.00	8.96	11.04	3.45			
9-Feb-01	37	20	2.50	28.00	10.12	17.88	5.59			
10-Feb-01	38	20	0.00	20.00	12.44	7.56	2.36			
11-Feb-01	39	20	0.00	20.00	10.24	9.76	3.05			
12-Feb-01	40	20	0.00	20.00	6.90	13.10	4.09	20.00	1.14	9.49
13-Feb-01	41	20	10.00	52.00	42.00	10.00	3.13			
14-Feb-01	42	20	8.00	45.60	35.60	10.00	3.13			
15-Feb-01	43	20	0.00	20.00	11.70	8.30	2.59			
16-Feb-01	44	20	0.00	20.00	11.50	8.50	2.66			
17-Feb-01	45	20	0.00	20.00	17.46	2.54	0.79			
18-Feb-01	46	20	18.00	77.60	67.60	10.00	3.13			
19-Feb-01	47	20	0.00	20.00	18.74	1.26	0.39			
20-Feb-01	48	20	0.00	20.00	13.20	6.80	2.13			
21-Feb-01	49	20	0.00	20.00	15.40	4.60	1.44			
22-Feb-01	50	20	0.00	20.00	9.96	10.04	3.14	20.00	11.52	24.32
23-Feb-01	51	20	0.00	20.00	15.62	4.38	1.37			
24-Feb-01	52	20	2.00	26.40	19.60	6.80	2.13			
25-Feb-01	53	20	0.00	20.00	13.20	6.80	2.13			
26-Feb-01	54	20	0.00	20.00	11.56	8.44	2.64	20.00	1.60	15.00
27-Feb-01	55	20	0.00	20.00	11.60	8.40	2.63			
28-Feb-01	56	20	0.00	20.00	12.80	7.20	2.25			
Total		560.00	71.50	788.80	536.90	251.90	78.72			
Average		20.00	2.55	28.17	19.18	9.00	2.81			
1-Mar-01	57	20	0.00	20.00	11.01	8.99	2.81			

2-Mar-01	58	20	0.00	20.00	10.60	9.40	2.94			
3-Mar-01	59	20	0.00	20.00	10.46	9.54	2.98			
4-Mar-01	60	20	0.00	20.00	10.44	9.56	2.99			
5-Mar-01	61	20	0.00	20.00	9.00	11.00	3.44			
6-Mar-01	62	20	0.00	20.00	9.84	10.16	3.18	20.00	0.00	10.72
7-Mar-01	63	20	0.00	20.00	8.90	11.10	3.47			
8-Mar-01	64	20	0.00	20.00	12.10	7.90	2.47			
9-Mar-01	65	20	0.00	20.00	11.32	8.68	2.71			
10-Mar-01	66	20	0.00	20.00	11.20	8.80	2.75			
11-Mar-01	67	20	0.00	20.00	9.10	10.90	3.41			
12-Mar-01	68	20	4.50	34.40	18.26	16.14	5.04			
13-Mar-01	69	20	0.00	20.00	14.46	5.54	1.73			
14-Mar-01	70	20	0.00	20.00	11.02	8.98	2.81			
15-Mar-01	71	20	0.00	20.00	8.96	11.04	3.45			
16-Mar-01	72	20	0.00	20.00	11.26	8.74	2.73			
17-Mar-01	73	20	6.50	40.80	30.80	10.00	3.13			
18-Mar-01	74	20	18.00	77.60	67.60	10.00	3.13			
19-Mar-01	75	20	3.00	29.60	23.34	6.26	1.96			
20-Mar-01	76	20	0.00	20.00	13.44	6.56	2.05			
21-Mar-01	77	20	1.00	23.20	13.30	9.90	3.09			
22-Mar-01	78	20	0.00	20.00	15.06	4.94	1.54			
23-Mar-01	79	20	2.00	26.40	18.86	7.54	2.36			
24-Mar-01	80	20	4.00	32.80	20.10	12.70	3.97			
25-Mar-01	81	20	0.00	20.00	12.60	7.40	2.31			
26-Mar-01	82	20	0.00	20.00	13.40	6.60	2.06	20.00	6.24	17.25
27-Mar-01	83	20	0.00	20.00	13.94	6.06	1.89			
28-Mar-01	84	20	0.00	20.00	12.70	7.30	2.28			
29-Mar-01	85	20	0.00	20.00	11.20	8.80	2.75			
30-Mar-01	86	20	0.00	20.00	10.62	9.38	2.93			
31-Mar-01	87	20	0.00	20.00	10.54	9.46	2.96			
Total		560.00	39.00	684.80	433.36	251.44	78.58			
Average		20.00	1.39	24.46	15.48	8.98	2.81			
1-Apr-01	88	20	16.00	71.20	61.20	10.00	3.13			

2-Apr-01	89	20	3.00	29.60	18.12	11.48	3.59			
3-Apr-01	90	20	1.50	24.80	17.48	7.32	2.29			
4-Apr-01	91	20	13.00	61.60	51.60	10.00	3.13			
5-Apr-01	92	20	5.00	36.00	25.20	10.80	3.38			
6-Apr-01	93	20	0.00	20.00	14.22	5.78	1.81			
7-Apr-01	94	20	0.00	20.00	10.20	9.80	3.06			
8-Apr-01	95	20	7.00	42.40	32.40	10.00	3.13			
9-Apr-01	96	20	15.00	68.00	58.00	10.00	3.13			
10-Apr-01	97	20	4.00	32.80	23.14	9.66	3.02			
11-Apr-01	98	20	0.00	20.00	12.42	7.58	2.37			
12-Apr-01	99	20	0.00	20.00	10.00	10.00	3.13			
13-Apr-01	100	20	0.00	20.00	11.00	9.00	2.81			
14-Apr-01	101	20	0.50	21.60	12.34	9.26	2.89			
15-Apr-01	102	20	0.00	20.00	13.32	6.68	2.09			
16-Apr-01	103	20	0.00	20.00	15.00	5.00	1.56			
17-Apr-01	104	20	0.00	20.00	14.00	6.00	1.88			
18-Apr-01	105	20	0.00	20.00	8.00	12.00	3.75	20.00	9.04	20.29
19-Apr-01	106	20	0.00	20.00	11.00	9.00	2.81			
20-Apr-01	107	20	0.00	20.00	10.26	9.74	3.04			
21-Apr-01	108	20	1.50	24.80	11.16	13.64	4.26			
22-Apr-01	109	20	1.50	24.80	14.00	10.80	3.38			
23-Apr-01	110	20	0.00	20.00	13.50	6.50	2.03			
24-Apr-01	111	20	0.00	20.00	14.72	5.28	1.65			
25-Apr-01	112	20	0.00	20.00	18.00	2.00	0.63			
26-Apr-01	113	20	6.00	39.20	29.20	10.00	3.13			
27-Apr-01	114	20	2.00	26.40	25.00	1.40	0.44			
28-Apr-01	115	20	0.00	20.00	15.30	4.70	1.47			
29-Apr-01	116	20	0.00	20.00	12.10	7.90	2.47			
30-Apr-01	117	20	4.00	32.80	21.72	11.08	3.46			
Total		540.00	59.50	730.40	506.80	223.60	69.88			
Average		20.00	2.20	27.05	18.77	8.28	2.59			
1-May-01	118	20	3.00	29.60	21.10	8.50	2.66			
2-May-01	119	20	4.00	32.80	25.00	7.80	2.44			

3-May-01	120	20	0.00	20.00	16.80	3.20	1.00			
4-May-01	121	20	0.00	20.00	12.00	8.00	2.50			
5-May-01	122	20	4.00	32.80	20.64	12.16	3.80			
6-May-01	123	20	0.00	20.00	15.86	4.14	1.29			
7-May-01	124	20	0.00	20.00	10.00	10.00	3.13			
8-May-01	125	20	0.00	20.00	7.00	13.00	4.06	20.00	4.16	16.22
9-May-01	126	20	0.00	20.00	11.00	9.00	2.81			
10-May-01	127	20	0.00	20.00	12.20	7.80	2.44			
11-May-01	128	20	0.00	20.00	13.44	6.56	2.05			
12-May-01	129	20	0.00	20.00	12.20	7.80	2.44			
13-May-01	130	20	0.00	20.00	10.30	9.70	3.03			
14-May-01	131	20	0.00	20.00	14.60	5.40	1.69			
15-May-01	132	20	0.00	20.00	12.00	8.00	2.50			
16-May-01	133	20	0.00	20.00	12.40	7.60	2.38			
17-May-01	134	20	0.00	20.00	12.10	7.90	2.47			
18-May-01	135	20	0.00	20.00	11.48	8.52	2.66			
19-May-01	136	20	0.00	20.00	12.52	7.48	2.34			
20-May-01	137	20	0.00	20.00	13.90	6.10	1.91			
21-May-01	138	20	0.00	20.00	7.20	12.80	4.00	20.00	0.00	11.95
22-May-01	139	20	0.00	20.00	13.80	6.20	1.94			
23-May-01	140	20	0.00	20.00	12.00	8.00	2.50			
24-May-01	141	20	0.00	20.00	14.70	5.30	1.66			
25-May-01	142	20	0.00	20.00	13.30	6.70	2.09			
26-May-01	143	20	0.00	20.00	11.20	8.80	2.75			
27-May-01	144	20	0.00	20.00	11.30	8.70	2.72			
28-May-01	145	20	0.00	20.00	15.00	5.00	1.56			
29-May-01	146	20	0.00	20.00	13.00	7.00	2.19			
30-May-01	147	20	0.00	20.00	13.90	6.10	1.91			
31-May-01	148	20	0.00	20.00	13.24	6.76	2.11			
Total		560.00	4.00	572.80	352.28	220.52	68.91			
Average		20.00	0.14	20.46	12.58	7.88	2.46			
1-Jun-01	149	20	0.00	20.00	15.98	4.02	1.26			
2-Jun-01	150	20	0.00	20.00	12.80	7.20	2.25			

3-Jun-01	151	20	0.00	20.00	12.40	7.60	2.38			
4-Jun-01	152	20	0.00	20.00	11.30	8.70	2.72	20.00	0.00	13.14
5-Jun-01	153	20	0.00	20.00	11.92	8.08	2.53			
6-Jun-01	154	20	0.00	20.00	13.10	6.90	2.16			
7-Jun-01	155	20	0.00	20.00	12.70	7.30	2.28			
8-Jun-01	156	20	0.00	20.00	12.48	7.52	2.35			
9-Jun-01	157	20	0.00	20.00	13.14	6.86	2.14			
10-Jun-01	158	20	0.00	20.00	14.30	5.70	1.78			
11-Jun-01	159	20	0.00	20.00	13.60	6.40	2.00			
12-Jun-01	160	20	0.00	20.00	13.14	6.86	2.14			
13-Jun-01	161	20	0.00	20.00	14.40	5.60	1.75			
14-Jun-01	162	20	0.00	20.00	12.78	7.22	2.26			
15-Jun-01	163	20	0.00	20.00	11.70	8.30	2.59			
16-Jun-01	164	20	0.00	20.00	11.90	8.10	2.53			
17-Jun-01	165	20	0.00	20.00	10.50	9.50	2.97			
18-Jun-01	166	20	0.00	20.00	11.00	9.00	2.81	20.00	0.00	12.62
19-Jun-01	167	20	0.00	20.00	11.30	8.70	2.72			
20-Jun-01	168	20	0.00	20.00	11.30	8.70	2.72			
21-Jun-01	169	20	0.00	20.00	12.90	7.10	2.22			
22-Jun-01	170	20	0.00	20.00	13.20	6.80	2.13			
23-Jun-01	171	20	0.00	20.00	12.50	7.50	2.34			
24-Jun-01	172	20	0.00	20.00	12.20	7.80	2.44			
25-Jun-01	173	20	0.00	20.00	13.70	6.30	1.97			
26-Jun-01	174	20	0.00	20.00	13.30	6.70	2.09			
27-Jun-01	175	20	0.00	20.00	13.80	6.20	1.94			
28-Jun-01	176	20	0.00	20.00	15.00	5.00	1.56			
29-Jun-01	177	20	0.00	20.00	13.62	6.38	1.99			
30-Jun-01	178	20	0.00	20.00	14.40	5.60	1.75			
Total		540.00	0.00	540.00	345.18	194.82	60.88			
Average		20.00	0.00	20.00	12.78	7.22	2.25			
1-Jul-01	179	20	0.00	20.00	11.84	8.16	2.55			
2-Jul-01	180	20	0.00	20.00	10.00	10.00	3.13	20.00	0.00	12.79
3-Jul-01	181	20	0.00	20.00	12.20	7.80	2.44			

4-Jul-01	182	20	0.00	20.00	11.00	9.00	2.81			
5-Jul-01	183	20	0.00	20.00	11.86	8.14	2.54			
6-Jul-01	184	20	0.00	20.00	11.80	8.20	2.56			
7-Jul-01	185	20	0.00	20.00	12.66	7.34	2.29			
8-Jul-01	186	20	0.00	20.00	12.00	8.00	2.50			
9-Jul-01	187	20	0.00	20.00	12.30	7.70	2.41			
10-Jul-01	188	20	0.00	20.00	11.60	8.40	2.63			
11-Jul-01	189	20	0.00	20.00	12.44	7.56	2.36			
12-Jul-01	190	20	0.00	20.00	12.00	8.00	2.50			
13-Jul-01	191	20	0.00	20.00	13.10	6.90	2.16			
14-Jul-01	192	20	0.00	20.00	12.90	7.10	2.22			
15-Jul-01	193	20	0.00	20.00	15.80	4.20	1.31			
16-Jul-01	194	20	0.00	20.00	11.92	8.08	2.53	20.00	0.00	12.40
17-Jul-01	195	20	0.00	20.00	12.90	7.10	2.22			
18-Jul-01	196	20	0.00	20.00	13.40	6.60	2.06			
19-Jul-01	197	20	0.00	20.00	12.30	7.70	2.41			
20-Jul-01	198	20	0.00	20.00	11.42	8.58	2.68			
21-Jul-01	199	20	1.00	23.20	11.58	11.62	3.63			
22-Jul-01	200	20	0.00	20.00	12.00	8.00	2.50			
23-Jul-01	201	20	0.00	20.00	13.60	6.40	2.00			
24-Jul-01	202	20	0.00	20.00	11.30	8.70	2.72			
25-Jul-01	203	20	0.00	20.00	12.00	8.00	2.50			
26-Jul-01	204	20	0.00	20.00	13.30	6.70	2.09			
27-Jul-01	205	20	0.00	20.00	12.56	7.44	2.33			
28-Jul-01	206	20	0.00	20.00	12.82	7.18	2.24			
29-Jul-01	207	20	50.00	180.00	170.00	10.00	3.13			
30-Jul-01	208	20	12.00	58.40	48.40	10.00	3.13			
31-Jul-01	209	20	0.00	20.00	9.60	10.40	3.25	20.00	13.44	25.15
Total		560.00	63.00	761.60	538.56	223.04	69.70			
Average		20.00	2.25	27.20	19.23	7.97	2.49			
1-Aug-01	210	20	0.00	20.00	10.80	9.20	2.88			
2-Aug-01	211	20	0.00	20.00	12.00	8.00	2.50			
3-Aug-01	212	20	0.00	20.00	11.40	8.60	2.69			

4-Aug-01	213	20	0.00	20.00	11.92	8.08	2.53			
5-Aug-01	214	20	1.00	23.20	15.10	8.10	2.53			
6-Aug-01	215	20	1.00	23.20	15.40	7.80	2.44			
7-Aug-01	216	20	0.00	20.00	12.80	7.20	2.25			
8-Aug-01	217	20	0.00	20.00	11.20	8.80	2.75			
9-Aug-01	218	20	0.00	20.00	10.70	9.30	2.91			
10-Aug-01	219	20	0.00	20.00	9.20	10.80	3.38			
11-Aug-01	220	20	0.00	20.00	6.90	13.10	4.09			
12-Aug-01	221	20	0.00	20.00	8.60	11.40	3.56			
13-Aug-01	222	20	0.00	20.00	10.70	9.30	2.91	20.00	0.49	11.29
14-Aug-01	223	20	0.00	20.00	11.30	8.70	2.72			
15-Aug-01	224	20	0.00	20.00	10.00	10.00	3.13			
16-Aug-01	225	20	0.00	20.00	14.86	5.14	1.61			
17-Aug-01	226	20	0.00	20.00	13.80	6.20	1.94			
18-Aug-01	227	20	0.00	20.00	10.56	9.44	2.95			
19-Aug-01	228	20	0.00	20.00	8.20	11.80	3.69			
20-Aug-01	229	20	0.00	20.00	11.66	8.34	2.61			
21-Aug-01	230	20	0.00	20.00	10.60	9.40	2.94			
22-Aug-01	231	20	0.00	20.00	10.00	10.00	3.13			
23-Aug-01	232	20	0.00	20.00	9.10	10.90	3.41			
24-Aug-01	233	20	0.00	20.00	12.56	7.44	2.33			
25-Aug-01	234	20	0.00	20.00	11.12	8.88	2.78			
26-Aug-01	235	20	0.00	20.00	10.50	9.50	2.97			
27-Aug-01	236	20	0.00	20.00	11.36	8.64	2.70			
28-Aug-01	237	20	0.00	20.00	10.00	10.00	3.13			
29-Aug-01	238	20	0.00	20.00	12.40	7.60	2.38			
30-Aug-01	239	20	0.00	20.00	10.84	9.16	2.86	20.00	0.00	11.11
31-Aug-01	240	20	0.00	20.00	17.20	2.80	0.88			
Total		560.00	2.00	566.40	318.58	247.82	77.44			
Average		20.00	0.07	20.23	11.38	8.85	2.77			

Table D4: *Vetiver zizanioides* flow data.

DATE	#	INFLUENT (l/d)	RAINFALL (mm/d)	TOTAL INFLUENT (l/d)	EFFLUENT (l/d)	TOTAL LOSS (l/d)	LOSS (mm/d)	AVERAGE INFLUENT (l/d)	AVERAGE RAINFALL (l/d)	AVERAGE EFFLUENT (l/d)
4-Jan-01	1	20	0.00	20.00	15.90	4.10	1.28			
5-Jan-01	2	20	0.00	20.00	14.74	5.26	1.64			
6-Jan-01	3	20	0.00	20.00	13.14	6.86	2.14			
7-Jan-01	4	20	0.00	20.00	12.82	7.18	2.24			
8-Jan-01	5	20	0.00	20.00	10.30	9.70	3.03	20.00	0.00	13.38
9-Jan-01	6	20	0.00	20.00	12.90	7.10	2.22			
10-Jan-01	7	20	0.00	20.00	12.80	7.20	2.25			
11-Jan-01	8	20	0.00	20.00	12.40	7.60	2.38	20.00	0.00	12.70
12-Jan-01	9	20	0.00	20.00	12.44	7.56	2.36			
13-Jan-01	10	20	0.00	20.00	12.34	7.66	2.39			
14-Jan-01	11	20	3.00	29.60	14.56	15.04	4.70			
15-Jan-01	12	20	0.50	21.60	13.76	7.84	2.45	20.00	2.80	13.28
16-Jan-01	13	20	0.00	20.00	13.10	6.90	2.16			
17-Jan-01	14	20	0.00	20.00	12.22	7.78	2.43			
18-Jan-01	15	20	0.00	20.00	9.88	10.12	3.16	20.00	0.00	11.73
19-Jan-01	16	20	0.00	20.00	9.70	10.30	3.22			
20-Jan-01	17	20	0.00	20.00	12.96	7.04	2.20			
21-Jan-01	18	20	1.00	23.20	14.80	8.40	2.63			
22-Jan-01	19	20	0.00	20.00	14.56	5.44	1.70	20.00	0.80	13.01
23-Jan-01	20	20	0.00	20.00	10.46	9.54	2.98			
24-Jan-01	21	20	0.00	20.00	13.24	6.76	2.11			
25-Jan-01	22	20	0.00	20.00	9.92	10.08	3.15	20.00	0.00	11.21
26-Jan-01	23	20	0.00	20.00	9.84	10.16	3.18			
27-Jan-01	24	20	0.00	20.00	10.06	9.94	3.11			
28-Jan-01	25	20	0.00	20.00	11.52	8.48	2.65			
29-Jan-01	26	20	0.00	20.00	7.94	12.06	3.77	20.00	0.00	9.84
30-Jan-01	27	20	12.00	58.40	48.40	10.00	3.13			
31-Jan-01	28	20	0.00	20.00	15.02	4.98	1.56			

Total		560.00	16.50	612.80	381.72	231.08	72.21			
Average		20.00	0.59	21.89	13.63	8.25	2.58			
1-Feb-01	29	20	6.00	39.20	17.64	21.56	6.74	20.00	19.20	27.02
2-Feb-01	30	20	0.00	20.00	14.34	5.66	1.77			
3-Feb-01	31	20	23.00	93.60	83.60	10.00	3.13			
4-Feb-01	32	20	2.00	26.40	16.00	10.40	3.25			
5-Feb-01	33	20	0.00	20.00	10.10	9.90	3.09	20.00	20.00	31.01
6-Feb-01	34	20	0.00	20.00	8.62	11.38	3.56			
7-Feb-01	35	20	0.00	20.00	8.56	11.44	3.58			
8-Feb-01	36	20	0.00	20.00	8.40	11.60	3.63			
9-Feb-01	37	20	2.50	28.00	8.42	19.58	6.12			
10-Feb-01	38	20	0.00	20.00	10.22	9.78	3.06			
11-Feb-01	39	20	0.00	20.00	10.40	9.60	3.00			
12-Feb-01	40	20	0.00	20.00	5.80	14.20	4.44	20.00	1.14	8.63
13-Feb-01	41	20	10.00	52.00	42.00	10.00	3.13			
14-Feb-01	42	20	8.00	45.60	35.60	10.00	3.13			
15-Feb-01	43	20	0.00	20.00	9.70	10.30	3.22			
16-Feb-01	44	20	0.00	20.00	10.22	9.78	3.06			
17-Feb-01	45	20	0.00	20.00	16.82	3.18	0.99			
18-Feb-01	46	20	18.00	77.60	67.60	10.00	3.13			
19-Feb-01	47	20	0.00	20.00	18.70	1.30	0.41			
20-Feb-01	48	20	0.00	20.00	11.90	8.10	2.53			
21-Feb-01	49	20	0.00	20.00	13.70	6.30	1.97			
22-Feb-01	50	20	0.00	20.00	9.06	10.94	3.42	20.00	11.52	23.53
23-Feb-01	51	20	0.00	20.00	11.42	8.58	2.68			
24-Feb-01	52	20	2.00	26.40	19.24	7.16	2.24			
25-Feb-01	53	20	0.00	20.00	10.96	9.04	2.83			
26-Feb-01	54	20	0.00	20.00	9.82	10.18	3.18	20.00	1.60	12.86
27-Feb-01	55	20	0.00	20.00	9.50	10.50	3.28			
28-Feb-01	56	20	0.00	20.00	10.40	9.60	3.00			
Total		560.00	71.50	788.80	508.74	280.06	87.52			
Average		20.00	2.55	28.17	18.17	10.00	3.13			

1-Mar-01	57	20	0.00	20.00	7.90	12.10	3.78			
2-Mar-01	58	20	0.00	20.00	9.84	10.16	3.18			
3-Mar-01	59	20	0.00	20.00	9.42	10.58	3.31			
4-Mar-01	60	20	0.00	20.00	7.94	12.06	3.77			
5-Mar-01	61	20	0.00	20.00	7.20	12.80	4.00			
6-Mar-01	62	20	0.00	20.00	7.74	12.26	3.83	20.00	0.00	8.74
7-Mar-01	63	20	0.00	20.00	6.32	13.68	4.28			
8-Mar-01	64	20	0.00	20.00	9.34	10.66	3.33			
9-Mar-01	65	20	0.00	20.00	8.40	11.60	3.63			
10-Mar-01	66	20	0.00	20.00	8.70	11.30	3.53			
11-Mar-01	67	20	0.00	20.00	6.68	13.32	4.16			
12-Mar-01	68	20	4.50	34.40	10.12	24.28	7.59			
13-Mar-01	69	20	0.00	20.00	11.44	8.56	2.68			
14-Mar-01	70	20	0.00	20.00	11.72	8.28	2.59			
15-Mar-01	71	20	0.00	20.00	7.92	12.08	3.78			
16-Mar-01	72	20	0.00	20.00	9.18	10.82	3.38			
17-Mar-01	73	20	6.50	40.80	30.80	10.00	3.13			
18-Mar-01	74	20	18.00	77.60	67.60	10.00	3.13			
19-Mar-01	75	20	3.00	29.60	24.38	5.22	1.63			
20-Mar-01	76	20	0.00	20.00	13.76	6.24	1.95			
21-Mar-01	77	20	1.00	23.20	16.20	7.00	2.19			
22-Mar-01	78	20	0.00	20.00	13.16	6.84	2.14			
23-Mar-01	79	20	2.00	26.40	16.80	9.60	3.00			
24-Mar-01	80	20	4.00	32.80	18.50	14.30	4.47			
25-Mar-01	81	20	0.00	20.00	12.30	7.70	2.41			
26-Mar-01	82	20	0.00	20.00	11.40	8.60	2.69	20.00	6.24	15.74
27-Mar-01	83	20	0.00	20.00	13.00	7.00	2.19			
28-Mar-01	84	20	0.00	20.00	11.86	8.14	2.54			
29-Mar-01	85	20	0.00	20.00	10.22	9.78	3.06			
30-Mar-01	86	20	0.00	20.00	10.98	9.02	2.82			
31-Mar-01	87	20	0.00	20.00	9.68	10.32	3.23			
Total		560.00	39.00	684.80	393.34	291.46	91.08			
Average		20.00	1.39	24.46	14.05	10.41	3.25			

1-Apr-01	88	20	16.00	71.20	61.20	10.00	3.13			
2-Apr-01	89	20	3.00	29.60	18.92	10.68	3.34			
3-Apr-01	90	20	1.50	24.80	17.62	7.18	2.24			
4-Apr-01	91	20	13.00	61.60	51.60	10.00	3.13			
5-Apr-01	92	20	5.00	36.00	24.20	11.80	3.69			
6-Apr-01	93	20	0.00	20.00	16.14	3.86	1.21			
7-Apr-01	94	20	0.00	20.00	9.82	10.18	3.18			
8-Apr-01	95	20	7.00	42.40	32.40	10.00	3.13			
9-Apr-01	96	20	15.00	68.00	58.00	10.00	3.13			
10-Apr-01	97	20	4.00	32.80	22.68	10.12	3.16			
11-Apr-01	98	20	0.00	20.00	13.24	6.76	2.11			
12-Apr-01	99	20	0.00	20.00	10.00	10.00	3.13			
13-Apr-01	100	20	0.00	20.00	11.50	8.50	2.66			
14-Apr-01	101	20	0.50	21.60	11.40	10.20	3.19			
15-Apr-01	102	20	0.00	20.00	13.26	6.74	2.11			
16-Apr-01	103	20	0.00	20.00	15.10	4.90	1.53			
17-Apr-01	104	20	0.00	20.00	13.10	6.90	2.16			
18-Apr-01	105	20	0.00	20.00	9.90	10.10	3.16	20.00	9.04	20.25
19-Apr-01	106	20	0.00	20.00	11.24	8.76	2.74			
20-Apr-01	107	20	0.00	20.00	10.58	9.42	2.94			
21-Apr-01	108	20	1.50	24.80	11.50	13.30	4.16			
22-Apr-01	109	20	1.50	24.80	14.40	10.40	3.25			
23-Apr-01	110	20	0.00	20.00	12.70	7.30	2.28			
24-Apr-01	111	20	0.00	20.00	14.42	5.58	1.74			
25-Apr-01	112	20	0.00	20.00	18.70	1.30	0.41			
26-Apr-01	113	20	6.00	39.20	29.20	10.00	3.13			
27-Apr-01	114	20	2.00	26.40	25.00	1.40	0.44			
28-Apr-01	115	20	0.00	20.00	15.10	4.90	1.53			
29-Apr-01	116	20	0.00	20.00	12.50	7.50	2.34			
30-Apr-01	117	20	4.00	32.80	22.60	10.20	3.19			
Total		540.00	59.50	730.40	510.28	220.12	68.79			
Average		20.00	2.20	27.05	18.90	8.15	2.55			

1-May-01	118	20	3.00	29.60	20.20	9.40	2.94			
2-May-01	119	20	4.00	32.80	25.00	7.80	2.44			
3-May-01	120	20	0.00	20.00	15.20	4.80	1.50			
4-May-01	121	20	0.00	20.00	12.68	7.32	2.29			
5-May-01	122	20	4.00	32.80	21.58	11.22	3.51			
6-May-01	123	20	0.00	20.00	13.86	6.14	1.92			
7-May-01	124	20	0.00	20.00	10.40	9.60	3.00			
8-May-01	125	20	0.00	20.00	7.00	13.00	4.06	20.00	4.16	16.19
9-May-01	126	20	0.00	20.00	11.80	8.20	2.56			
10-May-01	127	20	0.00	20.00	14.00	6.00	1.88			
11-May-01	128	20	0.00	20.00	12.20	7.80	2.44			
12-May-01	129	20	0.00	20.00	12.20	7.80	2.44			
13-May-01	130	20	0.00	20.00	12.30	7.70	2.41			
14-May-01	131	20	0.00	20.00	14.70	5.30	1.66			
15-May-01	132	20	0.00	20.00	12.44	7.56	2.36			
16-May-01	133	20	0.00	20.00	13.60	6.40	2.00			
17-May-01	134	20	0.00	20.00	13.66	6.34	1.98			
18-May-01	135	20	0.00	20.00	13.22	6.78	2.12			
19-May-01	136	20	0.00	20.00	11.82	8.18	2.56			
20-May-01	137	20	0.00	20.00	13.94	6.06	1.89			
21-May-01	138	20	0.00	20.00	9.40	10.60	3.31	20.00	0.00	12.71
22-May-01	139	20	0.00	20.00	14.80	5.20	1.63			
23-May-01	140	20	0.00	20.00	14.00	6.00	1.88			
24-May-01	141	20	0.00	20.00	15.10	4.90	1.53			
25-May-01	142	20	0.00	20.00	14.80	5.20	1.63			
26-May-01	143	20	0.00	20.00	12.60	7.40	2.31			
27-May-01	144	20	0.00	20.00	12.80	7.20	2.25			
28-May-01	145	20	0.00	20.00	16.30	3.70	1.16			
29-May-01	146	20	0.00	20.00	14.46	5.54	1.73			
30-May-01	147	20	0.00	20.00	13.20	6.80	2.13			
31-May-01	148	20	0.00	20.00	13.10	6.90	2.16			
Total		560.00	4.00	572.80	371.96	200.84	62.76			
Average		20.00	0.14	20.46	13.28	7.17	2.24			

1-Jun-01	149	20	0.00	20.00	15.28	4.72	1.48			
2-Jun-01	150	20	0.00	20.00	18.20	1.80	0.56			
3-Jun-01	151	20	0.00	20.00	13.80	6.20	1.94			
4-Jun-01	152	20	0.00	20.00	11.20	8.80	2.75	20.00	0.00	14.26
5-Jun-01	153	20	0.00	20.00	11.64	8.36	2.61			
6-Jun-01	154	20	0.00	20.00	15.20	4.80	1.50			
7-Jun-01	155	20	0.00	20.00	13.70	6.30	1.97			
8-Jun-01	156	20	0.00	20.00	13.48	6.52	2.04			
9-Jun-01	157	20	0.00	20.00	14.68	5.32	1.66			
10-Jun-01	158	20	0.00	20.00	15.60	4.40	1.38			
11-Jun-01	159	20	0.00	20.00	14.84	5.16	1.61			
12-Jun-01	160	20	0.00	20.00	15.42	4.58	1.43			
13-Jun-01	161	20	0.00	20.00	15.86	4.14	1.29			
14-Jun-01	162	20	0.00	20.00	14.32	5.68	1.78			
15-Jun-01	163	20	0.00	20.00	12.80	7.20	2.25			
16-Jun-01	164	20	0.00	20.00	14.44	5.56	1.74			
17-Jun-01	165	20	0.00	20.00	12.70	7.30	2.28			
18-Jun-01	166	20	0.00	20.00	12.96	7.04	2.20	20.00	0.00	14.12
19-Jun-01	167	20	0.00	20.00	12.80	7.20	2.25			
20-Jun-01	168	20	0.00	20.00	13.00	7.00	2.19			
21-Jun-01	169	20	0.00	20.00	15.10	4.90	1.53			
22-Jun-01	170	20	0.00	20.00	14.70	5.30	1.66			
23-Jun-01	171	20	0.00	20.00	15.20	4.80	1.50			
24-Jun-01	172	20	0.00	20.00	14.10	5.90	1.84			
25-Jun-01	173	20	0.00	20.00	13.00	7.00	2.19			
26-Jun-01	174	20	0.00	20.00	14.60	5.40	1.69			
27-Jun-01	175	20	0.00	20.00	16.24	3.76	1.18			
28-Jun-01	176	20	0.00	20.00	15.44	4.56	1.43			
29-Jun-01	177	20	0.00	20.00	15.46	4.54	1.42			
30-Jun-01	178	20	0.00	20.00	16.20	3.80	1.19			
Total		540.00	0.00	540.00	384.68	155.32	48.54			
Average		20.00	0.00	20.00	14.25	5.75	1.80			

1-Jul-01	179	20	0.00	20.00	13.50	6.50	2.03			
2-Jul-01	180	20	0.00	20.00	12.60	7.40	2.31	20.00	0.00	14.42
3-Jul-01	181	20	0.00	20.00	13.94	6.06	1.89			
4-Jul-01	182	20	0.00	20.00	12.84	7.16	2.24			
5-Jul-01	183	20	0.00	20.00	11.80	8.20	2.56			
6-Jul-01	184	20	0.00	20.00	13.34	6.66	2.08			
7-Jul-01	185	20	0.00	20.00	13.84	6.16	1.93			
8-Jul-01	186	20	0.00	20.00	14.80	5.20	1.63			
9-Jul-01	187	20	0.00	20.00	14.50	5.50	1.72			
10-Jul-01	188	20	0.00	20.00	14.70	5.30	1.66			
11-Jul-01	189	20	0.00	20.00	15.00	5.00	1.56			
12-Jul-01	190	20	0.00	20.00	15.04	4.96	1.55			
13-Jul-01	191	20	0.00	20.00	15.60	4.40	1.38			
14-Jul-01	192	20	0.00	20.00	15.64	4.36	1.36			
15-Jul-01	193	20	0.00	20.00	16.30	3.70	1.16			
16-Jul-01	194	20	0.00	20.00	14.30	5.70	1.78	20.00	0.00	14.40
17-Jul-01	195	20	0.00	20.00	16.20	3.80	1.19			
18-Jul-01	196	20	0.00	20.00	16.30	3.70	1.16			
19-Jul-01	197	20	0.00	20.00	14.90	5.10	1.59			
20-Jul-01	198	20	0.00	20.00	12.70	7.30	2.28			
21-Jul-01	199	20	1.00	23.20	13.26	9.94	3.11			
22-Jul-01	200	20	0.00	20.00	14.30	5.70	1.78			
23-Jul-01	201	20	0.00	20.00	14.30	5.70	1.78			
24-Jul-01	202	20	0.00	20.00	12.70	7.30	2.28			
25-Jul-01	203	20	0.00	20.00	13.24	6.76	2.11			
26-Jul-01	204	20	0.00	20.00	15.50	4.50	1.41			
27-Jul-01	205	20	0.00	20.00	15.00	5.00	1.56			
28-Jul-01	206	20	0.00	20.00	15.22	4.78	1.49			
29-Jul-01	207	20	50.00	180.00	170.00	10.00	3.13			
30-Jul-01	208	20	12.00	58.40	48.40	10.00	3.13			
31-Jul-01	209	20	0.00	20.00	14.60	5.40	1.69	20.00	13.44	27.11
Total		560.00	63.00	761.60	594.32	167.28	52.28			
Average		20.00	2.25	27.20	21.23	5.97	1.87			

1-Aug-01	210	20	0.00	20.00	15.00	5.00	1.56			
2-Aug-01	211	20	0.00	20.00	13.10	6.90	2.16			
3-Aug-01	212	20	0.00	20.00	14.60	5.40	1.69			
4-Aug-01	213	20	0.00	20.00	15.30	4.70	1.47			
5-Aug-01	214	20	1.00	23.20	18.40	4.80	1.50			
6-Aug-01	215	20	1.00	23.20	18.40	4.80	1.50			
7-Aug-01	216	20	0.00	20.00	15.40	4.60	1.44			
8-Aug-01	217	20	0.00	20.00	14.40	5.60	1.75			
9-Aug-01	218	20	0.00	20.00	13.40	6.60	2.06			
10-Aug-01	219	20	0.00	20.00	11.90	8.10	2.53			
11-Aug-01	220	20	0.00	20.00	10.20	9.80	3.06			
12-Aug-01	221	20	0.00	20.00	13.30	6.70	2.09			
13-Aug-01	222	20	0.00	20.00	13.40	6.60	2.06	20.00	0.49	14.37
14-Aug-01	223	20	0.00	20.00	13.10	6.90	2.16			
15-Aug-01	224	20	0.00	20.00	17.16	2.84	0.89			
16-Aug-01	225	20	0.00	20.00	17.36	2.64	0.83			
17-Aug-01	226	20	0.00	20.00	17.20	2.80	0.88			
18-Aug-01	227	20	0.00	20.00	13.52	6.48	2.03			
19-Aug-01	228	20	0.00	20.00	15.80	4.20	1.31			
20-Aug-01	229	20	0.00	20.00	14.24	5.76	1.80			
21-Aug-01	230	20	0.00	20.00	14.70	5.30	1.66			
22-Aug-01	231	20	0.00	20.00	14.38	5.62	1.76			
23-Aug-01	232	20	0.00	20.00	10.22	9.78	3.06			
24-Aug-01	233	20	0.00	20.00	11.70	8.30	2.59			
25-Aug-01	234	20	0.00	20.00	13.16	6.84	2.14			
26-Aug-01	235	20	0.00	20.00	11.26	8.74	2.73			
27-Aug-01	236	20	0.00	20.00	12.62	7.38	2.31			
28-Aug-01	237	20	0.00	20.00	15.20	4.80	1.50			
29-Aug-01	238	20	0.00	20.00	12.60	7.40	2.31			
30-Aug-01	239	20	0.00	20.00	9.70	10.30	3.22	20.00	0.00	13.76
31-Aug-01	240	20	0.00	20.00	17.10	2.90	0.91			
Total		560.00	2.00	566.40	395.12	171.28	53.53			
Average		20.00	0.07	20.23	14.11	6.12	1.91			

Table D5: *Phragmites australis* flow data.

DATE	#	INFLUENT (l/d)	RAINFALL (mm/d)	TOTAL INFLUENT (l/d)	EFFLUENT (l/d)	TOTAL LOSS (l/d)	LOSS (mm/d)	AVERAGE INFLUENT (l/d)	AVERAGE RAINFALL (l/d)	AVERAGE EFFLUENT (l/d)
4-Jan-01	1	20	0.00	20.00	12.40	7.60	2.38			
5-Jan-01	2	20	0.00	20.00	10.22	9.78	3.06			
6-Jan-01	3	20	0.00	20.00	9.44	10.56	3.30			
7-Jan-01	4	20	0.00	20.00	9.80	10.20	3.19			
8-Jan-01	5	20	0.00	20.00	10.76	9.24	2.89	20.00	0.00	10.52
9-Jan-01	6	20	0.00	20.00	10.10	9.90	3.09			
10-Jan-01	7	20	0.00	20.00	11.46	8.54	2.67			
11-Jan-01	8	20	0.00	20.00	11.64	8.36	2.61	20.00	0.00	11.07
12-Jan-01	9	20	0.00	20.00	11.52	8.48	2.65			
13-Jan-01	10	20	0.00	20.00	11.42	8.58	2.68			
14-Jan-01	11	20	3.00	29.60	10.00	19.60	6.13			
15-Jan-01	12	20	0.50	21.60	11.14	10.46	3.27	20.00	2.80	11.02
16-Jan-01	13	20	0.00	20.00	10.40	9.60	3.00			
17-Jan-01	14	20	0.00	20.00	10.12	9.88	3.09			
18-Jan-01	15	20	0.00	20.00	8.10	11.90	3.72	20.00	0.00	9.54
19-Jan-01	16	20	0.00	20.00	7.30	12.70	3.97			
20-Jan-01	17	20	0.00	20.00	10.12	9.88	3.09			
21-Jan-01	18	20	1.00	23.20	11.80	11.40	3.56			
22-Jan-01	19	20	0.00	20.00	13.72	6.28	1.96	20.00	0.80	10.74
23-Jan-01	20	20	0.00	20.00	8.20	11.80	3.69			
24-Jan-01	21	20	0.00	20.00	10.10	9.90	3.09			
25-Jan-01	22	20	0.00	20.00	7.30	12.70	3.97	20.00	0.00	8.53
26-Jan-01	23	20	0.00	20.00	8.00	12.00	3.75			
27-Jan-01	24	20	0.00	20.00	8.88	11.12	3.48			
28-Jan-01	25	20	0.00	20.00	10.20	9.80	3.06			
29-Jan-01	26	20	0.00	20.00	6.40	13.60	4.25	20.00	0.00	8.37
30-Jan-01	27	20	12.00	58.40	48.40	10.00	3.13			
31-Jan-01	28	20	0.00	20.00	12.72	7.28	2.28			
Total		560.00	16.50	612.80	321.66	291.14	90.98			
Average		20.00	0.59	21.89	11.49	10.40	3.25			

1-Feb-01	29	20	6.00	39.20	12.90	26.30	8.22	20.00	19.20	24.67
2-Feb-01	30	20	0.00	20.00	13.34	6.66	2.08			
3-Feb-01	31	20	23.00	93.60	83.60	10.00	3.13			
4-Feb-01	32	20	2.00	26.40	13.46	12.94	4.04			
5-Feb-01	33	20	0.00	20.00	7.48	12.52	3.91	20.00	20.00	29.47
6-Feb-01	34	20	0.00	20.00	6.96	13.04	4.08			
7-Feb-01	35	20	0.00	20.00	6.80	13.20	4.13			
8-Feb-01	36	20	0.00	20.00	5.42	14.58	4.56			
9-Feb-01	37	20	2.50	28.00	5.06	22.94	7.17			
10-Feb-01	38	20	0.00	20.00	8.06	11.94	3.73			
11-Feb-01	39	20	0.00	20.00	7.46	12.54	3.92			
12-Feb-01	40	20	0.00	20.00	9.70	10.30	3.22	20.00	1.14	7.07
13-Feb-01	41	20	10.00	52.00	42.00	10.00	3.13			
14-Feb-01	42	20	8.00	45.60	35.60	10.00	3.13			
15-Feb-01	43	20	0.00	20.00	6.90	13.10	4.09			
16-Feb-01	44	20	0.00	20.00	4.38	15.62	4.88			
17-Feb-01	45	20	0.00	20.00	8.62	11.38	3.56			
18-Feb-01	46	20	18.00	77.60	67.60	10.00	3.13			
19-Feb-01	47	20	0.00	20.00	16.80	3.20	1.00			
20-Feb-01	48	20	0.00	20.00	10.26	9.74	3.04			
21-Feb-01	49	20	0.00	20.00	8.00	12.00	3.75			
22-Feb-01	50	20	0.00	20.00	5.30	14.70	4.59	20.00	11.52	20.55
23-Feb-01	51	20	0.00	20.00	7.42	12.58	3.93			
24-Feb-01	52	20	2.00	26.40	15.16	11.24	3.51			
25-Feb-01	53	20	0.00	20.00	9.10	10.90	3.41			
26-Feb-01	54	20	0.00	20.00	8.30	11.70	3.66	20.00	1.60	10.00
27-Feb-01	55	20	0.00	20.00	7.20	12.80	4.00			
28-Feb-01	56	20	0.00	20.00	7.10	12.90	4.03			
Total		560.00	71.50	788.80	439.98	348.82	109.01			
Average		20.00	2.55	28.17	15.71	12.46	3.89			
1-Mar-01	57	20	0.00	20.00	6.00	14.00	4.38			

2-Mar-01	58	20	0.00	20.00	5.54	14.46	4.52			
3-Mar-01	59	20	0.00	20.00	5.57	14.43	4.51			
4-Mar-01	60	20	0.00	20.00	4.80	15.20	4.75			
5-Mar-01	61	20	0.00	20.00	3.02	16.98	5.31			
6-Mar-01	62	20	0.00	20.00	4.64	15.36	4.80	20.00	0.00	5.48
7-Mar-01	63	20	0.00	20.00	3.12	16.88	5.28			
8-Mar-01	64	20	0.00	20.00	6.40	13.60	4.25			
9-Mar-01	65	20	0.00	20.00	4.74	15.26	4.77			
10-Mar-01	66	20	0.00	20.00	6.50	13.50	4.22			
11-Mar-01	67	20	0.00	20.00	3.72	16.28	5.09			
12-Mar-01	68	20	4.50	34.40	7.08	27.32	8.54			
13-Mar-01	69	20	0.00	20.00	10.52	9.48	2.96			
14-Mar-01	70	20	0.00	20.00	8.38	11.62	3.63			
15-Mar-01	71	20	0.00	20.00	5.32	14.68	4.59			
16-Mar-01	72	20	0.00	20.00	5.84	14.16	4.43			
17-Mar-01	73	20	6.50	40.80	30.80	10.00	3.13			
18-Mar-01	74	20	18.00	77.60	67.60	10.00	3.13			
19-Mar-01	75	20	3.00	29.60	24.38	5.22	1.63			
20-Mar-01	76	20	0.00	20.00	10.60	9.40	2.94			
21-Mar-01	77	20	1.00	23.20	7.32	15.88	4.96			
22-Mar-01	78	20	0.00	20.00	10.90	9.10	2.84			
23-Mar-01	79	20	2.00	26.40	11.60	14.80	4.63			
24-Mar-01	80	20	4.00	32.80	18.30	14.50	4.53			
25-Mar-01	81	20	0.00	20.00	8.24	11.76	3.68			
26-Mar-01	82	20	0.00	20.00	10.00	10.00	3.13	20.00	6.24	13.07
27-Mar-01	83	20	0.00	20.00	9.06	10.94	3.42			
28-Mar-01	84	20	0.00	20.00	9.80	10.20	3.19			
29-Mar-01	85	20	0.00	20.00	6.52	13.48	4.21			
30-Mar-01	86	20	0.00	20.00	3.14	16.86	5.27			
31-Mar-01	87	20	0.00	20.00	3.54	16.46	5.14			
Total		560.00	39.00	684.80	305.88	378.92	118.41			
Average		20.00	1.39	24.46	10.92	13.53	4.23			
1-Apr-01	88	20	16.00	71.20	61.20	10.00	3.13			

2-Apr-01	89	20	3.00	29.60	15.18	14.42	4.51			
3-Apr-01	90	20	1.50	24.80	2.82	21.98	6.87			
4-Apr-01	91	20	13.00	61.60	51.60	10.00	3.13			
5-Apr-01	92	20	5.00	36.00	19.52	16.48	5.15			
6-Apr-01	93	20	0.00	20.00	13.34	6.66	2.08			
7-Apr-01	94	20	0.00	20.00	3.48	16.52	5.16			
8-Apr-01	95	20	7.00	42.40	32.40	10.00	3.13			
9-Apr-01	96	20	15.00	68.00	58.00	10.00	3.13			
10-Apr-01	97	20	4.00	32.80	18.22	14.58	4.56			
11-Apr-01	98	20	0.00	20.00	7.04	12.96	4.05			
12-Apr-01	99	20	0.00	20.00	3.20	16.80	5.25			
13-Apr-01	100	20	0.00	20.00	3.80	16.20	5.06			
14-Apr-01	101	20	0.50	21.60	1.12	20.48	6.40			
15-Apr-01	102	20	0.00	20.00	2.60	17.40	5.44			
16-Apr-01	103	20	0.00	20.00	8.60	11.40	3.56			
17-Apr-01	104	20	0.00	20.00	6.40	13.60	4.25			
18-Apr-01	105	20	0.00	20.00	2.00	18.00	5.63	20.00	9.04	14.89
19-Apr-01	106	20	0.00	20.00	2.92	17.08	5.34			
20-Apr-01	107	20	0.00	20.00	1.92	18.08	5.65			
21-Apr-01	108	20	1.50	24.80	1.40	23.40	7.31			
22-Apr-01	109	20	1.50	24.80	4.20	20.60	6.44			
23-Apr-01	110	20	0.00	20.00	2.40	17.60	5.50			
24-Apr-01	111	20	0.00	20.00	3.80	16.20	5.06			
25-Apr-01	112	20	0.00	20.00	4.40	15.60	4.88			
26-Apr-01	113	20	6.00	39.20	29.20	10.00	3.13			
27-Apr-01	114	20	2.00	26.40	25.00	1.40	0.44			
28-Apr-01	115	20	0.00	20.00	11.60	8.40	2.63			
29-Apr-01	116	20	0.00	20.00	5.20	14.80	4.63			
30-Apr-01	117	20	4.00	32.80	7.84	24.96	7.80			
Total		540.00	59.50	730.40	331.20	399.20	124.75			
Average		20.00	2.20	27.05	12.27	14.79	4.62			
1-May-01	118	20	3.00	29.60	15.30	14.30	4.47			
2-May-01	119	20	4.00	32.80	18.60	14.20	4.44			

3-May-01	120	20	0.00	20.00	12.20	7.80	2.44				
4-May-01	121	20	0.00	20.00	5.42	14.58	4.56				
5-May-01	122	20	4.00	32.80	5.14	27.66	8.64				
6-May-01	123	20	0.00	20.00	7.00	13.00	4.06				
7-May-01	124	20	0.00	20.00	0.60	19.40	6.06				
8-May-01	125	20	0.00	20.00	0.10	19.90	6.22	20.00	4.16	8.21	
9-May-01	126	20	0.00	20.00	0.30	19.70	6.16				
10-May-01	127	20	0.00	20.00	0.50	19.50	6.09				
11-May-01	128	20	0.00	20.00	2.80	17.20	5.38				
12-May-01	129	20	0.00	20.00	3.80	16.20	5.06				
13-May-01	130	20	0.00	20.00	1.80	18.20	5.69				
14-May-01	131	20	0.00	20.00	5.22	14.78	4.62				
15-May-01	132	20	0.00	20.00	3.44	16.56	5.18				
16-May-01	133	20	0.00	20.00	3.34	16.66	5.21				
17-May-01	134	20	0.00	20.00	8.84	11.16	3.49				
18-May-01	135	20	0.00	20.00	5.00	15.00	4.69				
19-May-01	136	20	0.00	20.00	4.40	15.60	4.88				
20-May-01	137	20	0.00	20.00	3.10	16.90	5.28				
21-May-01	138	20	0.00	20.00	0.20	19.80	6.19	20.00	0.00	3.29	
22-May-01	139	20	0.00	20.00	0.30	19.70	6.16				
23-May-01	140	20	0.00	20.00	0.10	19.90	6.22				
24-May-01	141	20	0.00	20.00	0.74	19.26	6.02				
25-May-01	142	20	0.00	20.00	2.10	17.90	5.59				
26-May-01	143	20	0.00	20.00	0.10	19.90	6.22				
27-May-01	144	20	0.00	20.00	0.30	19.70	6.16				
28-May-01	145	20	0.00	20.00	0.60	19.40	6.06				
29-May-01	146	20	0.00	20.00	2.40	17.60	5.50				
30-May-01	147	20	0.00	20.00	1.40	18.60	5.81				
31-May-01	148	20	0.00	20.00	0.58	19.42	6.07				
Total		560.00	4.00	572.80	69.62	503.18	157.24				
Average		20.00	0.14	20.46	2.49	17.97	5.62				
1-Jun-01	149	20	0.00	20.00	3.46	16.54	5.17				
2-Jun-01	150	20	0.00	20.00	0.90	19.10	5.97				

3-Jun-01	151	20	0.00	20.00	1.80	18.20	5.69			
4-Jun-01	152	20	0.00	20.00	1.86	18.14	5.67	20.00	0.00	1.19
5-Jun-01	153	20	0.00	20.00	0.30	19.70	6.16			
6-Jun-01	154	20	0.00	20.00	2.00	18.00	5.63			
7-Jun-01	155	20	0.00	20.00	3.02	16.98	5.31			
8-Jun-01	156	20	0.00	20.00	1.56	18.44	5.76			
9-Jun-01	157	20	0.00	20.00	2.00	18.00	5.63			
10-Jun-01	158	20	0.00	20.00	2.70	17.30	5.41			
11-Jun-01	159	20	0.00	20.00	3.84	16.16	5.05			
12-Jun-01	160	20	0.00	20.00	3.26	16.74	5.23			
13-Jun-01	161	20	0.00	20.00	6.00	14.00	4.38			
14-Jun-01	162	20	0.00	20.00	4.76	15.24	4.76			
15-Jun-01	163	20	0.00	20.00	0.94	19.06	5.96			
16-Jun-01	164	20	0.00	20.00	1.30	18.70	5.84			
17-Jun-01	165	20	0.00	20.00	0.10	19.90	6.22			
18-Jun-01	166	20	0.00	20.00	0.30	19.70	6.16	20.00	0.00	2.29
19-Jun-01	167	20	0.00	20.00	1.80	18.20	5.69			
20-Jun-01	168	20	0.00	20.00	2.42	17.58	5.49			
21-Jun-01	169	20	0.00	20.00	0.60	19.40	6.06			
22-Jun-01	170	20	0.00	20.00	0.80	19.20	6.00			
23-Jun-01	171	20	0.00	20.00	1.20	18.80	5.88			
24-Jun-01	172	20	0.00	20.00	0.20	19.80	6.19			
25-Jun-01	173	20	0.00	20.00	1.90	18.10	5.66			
26-Jun-01	174	20	0.00	20.00	2.90	17.10	5.34			
27-Jun-01	175	20	0.00	20.00	3.70	16.30	5.09			
28-Jun-01	176	20	0.00	20.00	5.62	14.38	4.49			
29-Jun-01	177	20	0.00	20.00	6.70	13.30	4.16			
30-Jun-01	178	20	0.00	20.00	7.24	12.76	3.99			
Total		540.00	0.00	540.00	69.02	470.98	147.18			
Average		20.00	0.00	20.00	2.56	17.44	5.45			
1-Jul-01	179	20	0.00	20.00	2.40	17.60	5.50			
2-Jul-01	180	20	0.00	20.00	2.00	18.00	5.63	20.00	0.00	2.82
3-Jul-01	181	20	0.00	20.00	1.22	18.78	5.87			

4-Jul-01	182	20	0.00	20.00	0.88	19.12	5.98			
5-Jul-01	183	20	0.00	20.00	0.10	19.90	6.22			
6-Jul-01	184	20	0.00	20.00	0.26	19.74	6.17			
7-Jul-01	185	20	0.00	20.00	1.22	18.78	5.87			
8-Jul-01	186	20	0.00	20.00	3.60	16.40	5.13			
9-Jul-01	187	20	0.00	20.00	4.60	15.40	4.81			
10-Jul-01	188	20	0.00	20.00	4.80	15.20	4.75			
11-Jul-01	189	20	0.00	20.00	6.38	13.62	4.26			
12-Jul-01	190	20	0.00	20.00	9.70	10.30	3.22			
13-Jul-01	191	20	0.00	20.00	12.34	7.66	2.39			
14-Jul-01	192	20	0.00	20.00	13.20	6.80	2.13			
15-Jul-01	193	20	0.00	20.00	14.40	5.60	1.75			
16-Jul-01	194	20	0.00	20.00	12.30	7.70	2.41	20.00	0.00	6.07
17-Jul-01	195	20	0.00	20.00	13.10	6.90	2.16			
18-Jul-01	196	20	0.00	20.00	13.60	6.40	2.00			
19-Jul-01	197	20	0.00	20.00	13.20	6.80	2.13			
20-Jul-01	198	20	0.00	20.00	11.67	8.33	2.60			
21-Jul-01	199	20	1.00	23.20	11.48	11.72	3.66			
22-Jul-01	200	20	0.00	20.00	11.26	8.74	2.73			
23-Jul-01	201	20	0.00	20.00	11.00	9.00	2.81			
24-Jul-01	202	20	0.00	20.00	11.00	9.00	2.81			
25-Jul-01	203	20	0.00	20.00	11.90	8.10	2.53			
26-Jul-01	204	20	0.00	20.00	13.30	6.70	2.09			
27-Jul-01	205	20	0.00	20.00	14.00	6.00	1.88			
28-Jul-01	206	20	0.00	20.00	14.26	5.74	1.79			
29-Jul-01	207	20	50.00	180.00	170.00	10.00	3.13			
30-Jul-01	208	20	12.00	58.40	48.40	10.00	3.13			
31-Jul-01	209	20	0.00	20.00	11.00	9.00	2.81	20.00	13.44	25.28
Total		560.00	63.00	761.60	462.95	298.65	93.33			
Average		20.00	2.25	27.20	16.53	10.67	3.33			
1-Aug-01	210	20	0.00	20.00	11.60	8.40	2.63			
2-Aug-01	211	20	0.00	20.00	13.10	6.90	2.16			
3-Aug-01	212	20	0.00	20.00	13.60	6.40	2.00			

4-Aug-01	213	20	0.00	20.00	12.70	7.30	2.28			
5-Aug-01	214	20	1.00	23.20	17.00	6.20	1.94			
6-Aug-01	215	20	1.00	23.20	14.90	8.30	2.59			
7-Aug-01	216	20	0.00	20.00	13.30	6.70	2.09			
8-Aug-01	217	20	0.00	20.00	12.40	7.60	2.38			
9-Aug-01	218	20	0.00	20.00	11.60	8.40	2.63			
10-Aug-01	219	20	0.00	20.00	9.60	10.40	3.25			
11-Aug-01	220	20	0.00	20.00	7.60	12.40	3.88			
12-Aug-01	221	20	0.00	20.00	8.70	11.30	3.53			
13-Aug-01	222	20	0.00	20.00	9.10	10.90	3.41	20.00	0.49	11.94
14-Aug-01	223	20	0.00	20.00	11.84	8.16	2.55			
15-Aug-01	224	20	0.00	20.00	12.00	8.00	2.50			
16-Aug-01	225	20	0.00	20.00	15.06	4.94	1.54			
17-Aug-01	226	20	0.00	20.00	15.22	4.78	1.49			
18-Aug-01	227	20	0.00	20.00	11.68	8.32	2.60			
19-Aug-01	228	20	0.00	20.00	11.50	8.50	2.66			
20-Aug-01	229	20	0.00	20.00	12.48	7.52	2.35			
21-Aug-01	230	20	0.00	20.00	11.68	8.32	2.60			
22-Aug-01	231	20	0.00	20.00	11.68	8.32	2.60			
23-Aug-01	232	20	0.00	20.00	11.34	8.66	2.71			
24-Aug-01	233	20	0.00	20.00	12.40	7.60	2.38			
25-Aug-01	234	20	0.00	20.00	10.66	9.34	2.92			
26-Aug-01	235	20	0.00	20.00	10.28	9.72	3.04			
27-Aug-01	236	20	0.00	20.00	11.54	8.46	2.64			
28-Aug-01	237	20	0.00	20.00	11.70	8.30	2.59			
29-Aug-01	238	20	0.00	20.00	9.20	10.80	3.38			
30-Aug-01	239	20	0.00	20.00	7.20	12.80	4.00	20.00	0.00	11.62
31-Aug-01	240	20	0.00	20.00	16.00	4.00	1.25			
Total		560.00	2.00	566.40	330.36	236.04	73.76			
Average		20.00	0.07	20.23	11.80	8.43	2.63			

APPENDIX D 3

Table D6: Main storage tank data.

Day	COD (mgO_2/l)	BOD_5 (mgO_2/l)	BOD_{20} (mgO_2/l)	Chlorides ($mgCl^-/l$)	pH	Alkalinity ($mgCaCO_3/l$)	Ammoniacal Nitrogen ($mgNH_3$-N/l)	TSS (mg/l)	Conductivity (mS/m)
5	495	18.0	31.6	1869	8.41	1325			
8	490			1759	8.21	1285			
12	490			1739	8.36	1240	1.2		
15	484			1799	8.28	1225		112	731
19	490			1729	8.36	1245			
22	502			1769	8.30	1205			
26	504			1739	8.45	1190			
29	504			1759	8.41	1175			
33	502			1789	8.43	1165			
40	504	16.8	25.7	1839	8.47	1075			
50	484			1799	8.54	1070			726
54	518			1839	8.38	1085	1.4	50	727
62	476	15.9	23.9	1819	8.25	1225			705
82	530	10.0	31.6	1820	8.40	1240	1.1	113	710
105	546			1819	8.15	1210		110	716
125	568	13.4	28.2	1819	8.63	1260			706
138	542			1829	8.51	1390	1.5		713
152	612	9.7	23.4	1799	8.70	1460			717
166	536			1779	8.57	1480	2.0	121	734
180	556	5.9	18.6	1879	8.64	1505			735
194	552			1859	8.43	1460	2.6	78	729
209	589	5.7	18.2	1819	8.63	1520			739
222	612			1803	8.51	1480			730
239	550			1819	8.45	1450			730

Table D7: Influent Mix data.

Day	COD (mgO_2/l)	BOD_5 (mgO_2/l)	BOD_{20} (mgO_2/l)	Chlorides ($mgCl^-/l$)	pH	Alkalinity ($mgCaCO_3/l$)	Ammoniacal Nitrogen ($mgNH_3$-N/l)	TSS (mg/l)	Conductivity (mS/m)
5	234	10.0	15.4	930	8.46	735			
8	235			1030	8.25	780			
12	250			1090	8.36	750	0.8		
15	251			980	8.36	755		78	423
19	256			930	8.43	755			
22	247			1000	8.44	745			
26	256			920	8.63	720			
29	259			1050	8.50	730			
33	256			1030	8.56	730			
40	252	12.1	19.8	1070	8.49	660			
50	256			1040	8.45	670			430
54	271			1060	8.61	670	0.8	32	434
62	256	10.9	16.5	1040	8.28	770			423
82	287	6.8	11.5	1040	8.50	730	0.8	10	420
105	299			970	8.21	745		10	419
125	304	5.3	28.2	1050	8.65	795			431
138	271			1030	8.42	825	0.7		409
152	302	4.0	13.2	1000	8.83	890			420
166	302			1030	8.68	890	0.9	66	436
180	257	3.4	10.8	1050	8.76	850			402
194	295			1010	8.40	845	1.1	41	418
209	313	4.0	10.9	970	8.64	850			420
222	302			1055	8.57	890			425
239	304			1040	8.60	850			410

Table D8: Control sample pipe data, sample pipes 1 and 2 respectively.

Day	COD (mgO_2/l)	Chlorides ($mgCl^-/l$)	pH	Alkalinity ($mgCaCO_3/l$)	Conductivity (mS/m)	COD (mgO_2/l)	Chlorides ($mgCl^-/l$)	pH	Alkalinity ($mgCaCO_3/l$)	Conductivity (mS/m)
5	21	140	7.95	195		27	130	8.01	270	
8	25	140	7.92	255		17.5	160	7.93	335	
12	82	290	8.12	350		76	330	8.00	480	
15	68	340	8.02	385	186	60	300	7.90	450	177
19	77	340	8.23	415		69	320	8.14	525	
22	86	440	8.11	455		78	410	7.93	520	
26	119	490	8.16	465		80	440	7.89	525	
29	118	620	8.05	490		89	500	7.86	550	
33	110	580	8.37	465		94	540	8.04	535	
40	117	620	8.26	475		99	540	8.03	495	
50	103	660	8.28	435	275	108	590	7.91	460	259
54	120	640	8.37	420	272	108	590	7.99	440	258
62	122	690	8.38	505	278	118	660	8.04	530	269
82	155	710	8.23	560	312	159	700	8.00	590	297
105	191	750	8.30	630	331	159	730	7.80	625	324
125	210	929	8.42	680	372	202	890	7.92	610	338
138	237	1010	8.32	740	401	181	970	7.77	625	355
152	238	990	8.70	755	409	189	960	8.07	640	377
166	240	1010	8.57	850	430	194	1020	7.83	745	399
180	253	1100	8.64	875	429	208	1140	8.05	765	416
194	274	1170	8.49	875	439	242	1230	7.96	775	412
209	192	610	8.88	535	288	181	870	8.24	605	348
222	271	938	8.56	865	397	243	974	8.36	775	390
239	307	974	8.65	880	412	261	1010	8.30	780	398

Table D9 Control effluent data.

Day	COD (mgO_2/l)	BOD_5 (mgO_2/l)	BOD_{20} (mgO_2/l)	Chlorides ($mgCl^-/l$)	pH	Alkalinity ($mgCaCO_3/l$)	Ammoniacal Nitrogen ($mgNH_3$-N/l)	TSS (mg/l)	Conductivity (mS/m)
5	93	6.8	12.4	440	8.35	510			
8	101			540	8.16	540			
12	115			550	8.27	565	0.4		
15	116			570	8.21	580		79	275
19	127			610	8.32	615			
22	139			700	8.22	640			
26	153			710	8.31	645			
29	141			720	8.18	655			
33	145			650	8.30	645			
40	149	10.9	12.7	770	8.53	675			
50	158			810	8.28	630			350
54	170			830	8.28	630	0.4	0	352
62	174	8.3	12.4	870	8.34	740			359
82	187	6.5	8.0	860	8.25	740		5	364
105	195			840	8.22	785		10	372
125	180	2.1	4.1	850	8.50	580			327
138	203			1000	8.19	710	0.5		370
152	181	0.0	1.4	980	8.52	725			386
166	186			940	8.50	650	0.6	37	387
180	210	0.0	0.4	1120	8.69	710			408
194	233			1130	8.47	735	0.6	5	412
209	260	0.0	0.0	1170	8.34	775			455
222	269			1019	8.62	790			396
239	270			1046	8.55	815			407

Table D10: *Leersia hexandra* sample pipe data, sample pipes 1 and 2 respectively.

Day	COD (mgO_2/l)	Chlorides ($mgCl^-/l$)	pH	Alkalinity ($mgCaCO_3/l$)	Conductivity (mS/m)	COD (mgO_2/l)	Chlorides ($mgCl^-/l$)	pH	Alkalinity ($mgCaCO_3/l$)	Conductivity (mS/m)
5	11	130	7.78	255		15	150	7.76	245	
8	28	150	7.88	290		21	160	7.93	355	
12	84	320	8.00	410		77	290	8.01	535	
15	70	380	7.68	440	206	64	310	7.91	525	199
19	86	400	7.96	495		73	320	7.98	595	
22	102	550	8.13	600		82	430	8.06	600	
26	118	530	7.83	515		102	490	7.72	645	
29	147	670	7.62	600		117	560	7.70	635	
33	150	730	7.89	605		119	610	7.88	645	
40	156	720	7.89	570		122	650	7.97	575	
50	160	840	7.74	580	346	151	720	7.83	550	314
54	167	780	7.74	560	330	156	730	7.85	535	318
62	176	900	7.89	670	363	156	830	7.93	635	328
82	231	880	7.75	720	358	223	830	7.80	785	350
105	215	820	7.50	855	375	215	840	7.59	825	376
125	242	1060	7.53	910	428	214	1020	7.65	850	402
138	271	1150	7.34	938	465	243	1120	7.37	950	433
152	260	1140	7.55	1015	474	234	1150	7.72	995	468
166	264	1120	7.50	1085	482	238	1160	7.52	1065	485
180	266	1300	7.54	1120	496	272	1380	7.60	1090	513
194	293	1360	7.53	1135	503	319	1410	7.57	1115	523
209	271	810	7.54	760	346	242	880	7.56	790	365
222	283	1019	7.57	1095	442	297	1100	7.70	1055	448
239	380	1181	7.57	1135	503	345	1280	7.58	1160	516

Table D11: *Leersia hexandra* effluent data.

Day	COD (mgO_2/l)	BOD_5 (mgO_2/l)	BOD_{20} (mgO_2/l)	Chlorides ($mgCl^-$/l)	pH	Alkalinity ($mgCaCO_3$/l)	Ammoniacal Nitrogen ($mgNH_3$-N/l)	TSS (mg/l)	Conductivity (mS/m)
5	133	7.4	11.8	630	8.26	590			
8	138			600	8.40	610			
12	150			620	8.30	645	0.5		
15	146			630	8.21	655		50	303
19	152			630	8.30	705			
22	157			730	8.30	715			
26	154			750	8.32	735			
29	165			760	8.08	760			
33	161			710	8.33	665			
40	163	11.8	16.2	820	8.48	720			
50	160			790	8.24	710			366
54	168			850	8.32	745	0.6	9	368
62	166	9.4	10.6	900	8.43	900			372
82	205	5.0	6.8	850	8.20	905		6	385
105	221			870	8.30	925		11	391
125	234	1.5	4.3	1020	8.53	810			397
138	241			1120	8.18	900	0.5		432
152	248	0.0	1.5	1170	8.39	965			465
166	252			1230	8.38	1070	0.7	47	495
180	266	0.0	3.5	1370	8.43	1060			514
194	308			1410	8.21	1120	0.7	0	540
209	333	0.0	2.0	1420	7.73	1145			570
222	332			1334	8.43	1135			519
239	318			1289	8.02	1200			524

Table D12: *Vetiver zizanioides* sample pipe data, sample pipes 1 and 2 respectively.

Day	COD (mgO_2/l)	Chlorides ($mgCl^-/l$)	pH	Alkalinity ($mgCaCO_3/l$)	Conductivity (mS/m)	COD (mgO_2/l)	Chlorides ($mgCl^-/l$)	pH	Alkalinity ($mgCaCO_3/l$)	Conductivity (mS/m)
5	96	280	7.22	515		118	300	7.15	520	
8	163	400	7.76	530		124	370	7.35	590	
12	181	610	7.58	640		157	540	7.45	730	
15	168	640	7.51	665	309	151	540	7.52	660	274
19	178	670	7.68	680		156	540	7.55	700	
22	189	720	8.32	700		161	640	7.73	710	
26	217	820	7.45	715		193	680	7.33	690	
29	220	900	7.48	775		202	780	7.08	755	
33	218	920	7.43	785		202	830	7.29	750	
40	228	930	7.55	750		204	840	7.36	715	
50	225	910	7.47	695	385	200	860	6.96	635	359
54	230	940	7.43	690	392	221	930	7.25	630	357
62	308	1030	7.59	795	408	212	950	7.34	770	381
82	257	910	7.50	790	376	267	900	7.10	760	392
105	231	790	7.23	780	367	205	820	7.03	800	363
125	260	990	7.55	800	402	244	950	7.24	785	382
138	269	1050	7.28	875	424	281	1100	7.23	855	425
152	260	1020	7.63	880	437	252	1140	7.59	930	445
166	272	1010	7.38	960	442	258	1060	7.40	975	462
180	307	1170	7.47	985	450	278	1230	7.52	990	463
194	300	1190	7.56	960	448	309	1240	7.56	1010	470
209	267	660	7.50	595	297	260	580	7.66	450	249
222	304	1028	8.56	920	416	288	1028	7.65	875	406
239	314	1019	7.90	965	429	330	1028	7.53	990	438

Table D13: *Vetiver zizanioides* effluent data.

Day	COD (mgO_2/l)	BOD_5 (mgO_2/l)	BOD_{20} (mgO_2/l)	Chlorides ($mgCl^-/l$)	pH	Alkalinity ($mgCaCO_3/l$)	Ammoniacal Nitrogen ($mgNH_3$-N/l)	TSS (mg/l)	Conductivity (mS/m)
5	141	10.9	24.2	600	8.05	1010			
8	135			660	7.82	1040			
12	170			630	8.20	1035	2.2		
15	145			600	8.01	980		101	326
19	146			570	8.19	985			
22	148			660	8.00	960			
26	177			710	8.00	910			
29	189			720	7.80	775			
33	185			780	8.03	675			
40	174	15.1	25.4	860	8.28	815			
50	173			860	7.97	695			371
54	204			900	8.02	750	1.6	19	377
62	204	7.7	15.6	920	8.16	910			387
82	255	6.8	12.7	890	8.20	890		13	378
105	227			850	7.85	860		25	382
125	224	2.7	7.6	960	8.27	775			378
138	253			1100	8.00	850	1.4		412
152	250	0.3	5.3	1240	8.40	910			435
166	262			1120	8.21	970	0.7	60	455
180	276	0.3	6.5	1280	8.34	995			464
194	309			1370	8.17	1000	0.7	24	472
209	344	0.0	4.2	1240	7.53	950			488
222	281			1001	8.13	900			410
239	304			1082	8.17	970			434

Table D14: *Phragmites australis* sample pipe data, sample pipes 1 and 2 respectively.

Day	COD (mgO_2/l)	Chlorides ($mgCl^-/l$)	pH	Alkalinity ($mgCaCO_3/l$)	Conductivity (mS/m)	COD (mgO_2/l)	Chlorides ($mgCl^-/l$)	pH	Alkalinity ($mgCaCO_3/l$)	Conductivity (mS/m)
5	89	290	7.68	495		81	300	7.80	600	
8	104	460	7.80	600		136	410	7.86	755	
12	163	660	8.26	725		172	550	7.88	820	
15	162	730	7.75	740	336	156	600	8.06	815	310
19	179	800	7.94	785		155	600	7.97	815	
22	196	860	7.97	825		154	710	7.84	830	
26	203	870	7.64	835		187	790	7.64	840	
29	232	1000	7.55	800		190	850	7.31	930	
33	237	1060	7.74	785		198	900	7.53	955	
40	242	1050	7.76	785		206	990	7.51	905	
50	216	1040	7.58	740	441	234	1060	7.34	850	443
54	255	1070	7.70	765	446	227	1070	7.44	870	446
62	260	1160	7.72	905	462	232	1190	7.57	1015	474
82	277	1100	7.80	925	460	275	1150	7.35	1085	480
105	261	1020	7.35	950	459	287	1130	7.19	1120	493
125	296	1349	7.44	1150	545	350	1480	7.37	1235	584
138	371	1560	7.20	1275	620	379	1780	7.20	1335	677
152	373	1699	7.35	1530	695	401	1949	7.67	1470	760
166	401	1889	7.46	1640	761	436	2079	7.61	1625	831
180	408	2149	7.50	1570	812	455	2459	7.67	1710	892
194	421	2099	7.45	1630	783	481	2349	7.50	1725	834
209	374	1310	7.70	965	505	359	1200	7.68	830	466
222	327	1181	7.94	1060	480	320	1127	7.60	1070	467
239	359	1145	7.67	1090	504	352	1370	7.70	1170	554

Table D15: *Phragmites australis* effluent data.

Day	COD (mgO_2/l)	BOD_5 (mgO_2/l)	BOD_{20} (mgO_2/l)	Chlorides ($mgCl^-/l$)	pH	Alkalinity ($mgCaCO_3/l$)	Ammoniacal Nitrogen ($mgNH_3$-N/l)	TSS (mg/l)	Conductivity (mS/m)
5	145	10.0	15.1	619	8.24	945			
8	136			650	7.90	965			
12	137			630	7.88	820	0.7		
15	131			620	8.10	915		85	322
19	144			630	8.22	935			
22	158			720	8.17	950			
26	187			780	8.18	925			
29	191			870	8.06	850			
33	191			890	8.27	840			
40	194	10.0	12.7	990	8.62	780			
50	245			1100	8.37	815			440
54	228			1070	8.47	835	0.7	15	444
62	238	7.4	10.9	1130	8.73	960			448
82	287	5.3	10.9	1140	8.56	1095		14	486
105	299			1180	8.48	1125		16	510
125	334	3.1	8.0	1440	8.74	1170			566
138	375			1730	8.55	1235	0.7		641
152	365	0.0	1.1	1959	9.00	1310			722
166	421			2089	7.61	1545	1.0	61	813
180	443	0.0	2.8	2489	8.82	1580			885
194	500			2669	8.15	1710	1.0	1	912
209	458	0.0	1.0	2029	7.74	1460			770
222	402			1839	8.30	1410			723
239	348			1578	8.23	1315			623