The Use of Constructed Wetlands as an Alternative Form of On-site Wastewater Treatment in a Temperate Climate.

by

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Declaration

This thesis contains no material which has been accepted for the award of any other higher degree in any tertiary institution and, to the best of my knowledge and belief, contains no material previously published or written by any other person, except where due reference is made.

Signed: ....

Abstract

It is estimated that between 40% and 60% of on-site systems in Australia are not treating domestic wastewater to acceptable levels. This study investigates the viability of constructed wetlands as an alternative form of on-site treatment of domestic greywater in temperate climates. A case study wetland was constructed to treat domestic greywater from a single household, providing an insight into the design, construction, operational performance, hydraulic flow and reed growth of a small-scale wetland operating in Tasmania. A review of current literature provides an insight into constructed wetlands, emphasising their use as a form of on-site wastewater treatment.

Interviews with officers from five local government councils indicates that knowledge of constructed wetlands is very limited, but a great deal of interest was shown by the interviewed environmental health officers, who suggested that they may be willing to trial wetlands within their municipalities. All the local governments interviewed were experiencing problems with the current and accepted forms of on-site wastewater treatment.

Testing of the case study wetland over a nine-month period showed that pollutant removal processes were occurring across all testing parameters except ammonia and phosphorous. Significant findings from the case study that would improve the quality of the final effluent include the importance of healthy reeds with deep root systems, improving hydraulic flow within the wetland to prevent short-circuiting, and the necessity of additional pre and post wetland treatment. Greywater alone does not have sufficient nutrients and trace elements to sustain vigorous growth for the common reed, *Phragmites australis*. A combined flow of black and greywater led to vigorous and healthy reed growth.

The wetland produced an effluent with an average of 64 mg/L BOD_5 (60% removal), 48.5 mg/L suspended solids (88% removal), 18,427 FCU/100mL faecal coliforms (99.2% removal), 0.008 mg/L nitrite (65% removal), 2.9 mg/L nitrate (6% removal), 2.2 mg/L ammonia, 6.7 mg/L phosphorous, 537 μS/cm conductivity, and a pH of 7.7. It is expected that treatment would improve as the reeds and root systems mature and the design improvements are implemented.

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Table of Contents

List of Figures

List of Tables

List of Plates

Chapter 1. Introduction

> The ancients revered water. In the age of innocence, earth, air, fire and water were regarded as the four constituent elements. Mythology lent them to a richness of character and a range of mysterious attributes. During the intervening millennia, human settlements, human activities and human festivals have all reflected the central role of fresh water.
>
> Today in the age of science, myths and mysteries are not in vogue. Neither, seemingly, is respect for water. Human indifference, human ignorance, and human greed combine globally to waste it, foul it, and divert it..... There is probably no other commodity so treasured by some while regarded with such indifference by others (IDRC 1989: 3).

Water has sustained the world's population for thousands of years in a complex interaction with the rest of the natural environment. Water is the basis of life on earth; without it plants and animals would not survive and fertile soils would become wastelands. In western societies it is hard to acknowledge the importance of water in a spiritual and cultural sense, as the ancients did and some strong aboriginal cultures still do today.

Humans are the largest users and contaminators of water. Rarely is our 'waste water' treated to satisfactory levels before it is discharged into the environment, where it can adversely affect all inhabitants of local ecosystems (plants, animals, invertebrates, microorganisms etc.). Ultimately humans, as a part of nature and highest on the food chain, will feel these adverse effects in one form or another. Such is the complexity of modern society that water is utilised in thousands of different ways, from industry and agriculture down to domestic chores such as drinking, washing, cleaning and waste disposal. It is the use of water in domestic chores that this thesis will focus upon. In urban areas household wastewater is transported through sewers to a centralised treatment plant before being discharged to the environment. While there are many issues associated with centralised treatment systems, the focus of this thesis will be upon the use of constructed wetlands to treat domestic wastewater in areas without the convenience of centralised systems. In these areas all wastewater produced must be treated and disposed of on-site.

The most common method of on-site treatment in Australia is by septic tank and it is estimated that 40% to 60% of septic tank systems fail, that is they are not treating wastewater to acceptable levels (Petrozzi and Martens 1995: 22). Pollution of land, waterways and groundwater is common due to the failure rate of current forms of on-site wastewater treatment. This pollution problem in areas dependent upon on-site wastewater treatment is further exacerbated by the current trend of population movement into these areas (Wood and White 1982, Wood 1989). A number of factors can help to explain this trend: acceptable commuting distances have expanded; people are buying holiday homes with the view of retiring in them; cheaper land is available; rural-residential subdivisions are increasing; and these locations appeal to people seeking a non-conventional lifestyle. With an increase in population comes the added pressures that people place on the environment. Sewage disposal is one of these pressures that are increasingly becoming a major concern for health and environmental reasons. Homeowners must take responsibility of their wastewater and ensure that their systems are installed, operated and maintained in order to prevent their daily activities contributing to the overall problem in their localised area or waterway.

While other treatment systems have entered the on-site wastewater treatment market in recent years, such as the aerated wastewater treatment system, there has been little research into the feasibility and performance of constructed wetlands for this use. Large-scale constructed wetlands are used throughout the world and have become an accepted form of treating municipal wastewater and agricultural and industrial effluent. For these treatment purposes constructed wetlands have been proven to be reliable and robust performers. While little research has been conducted on the use of small-scale constructed wetlands to treat wastewater on-site, they have the potential to perform as reliably and consistently as medium to large-scale systems. If the reliability of small-scale constructed wetlands is proven then they may provide an alternative form of on-site wastewater treatment, which could help alleviate the health and environmental concerns associated with wastewater in areas without centralised treatment.

1.1. Aim and Objectives of this Study

The aim of this thesis is to address the research question: are constructed wetlands a viable option for the on-site treatment of domestic greywater in temperate climates?

In order to achieve this specific aim a number of objectives must be met. These objectives are:

- to establish that there are problems (ie. health, environmental, water conservation etc.) associated with domestic greywater and that it must be treated adequately;
- to investigate whether current and accepted forms of on-site systems are treating wastewater adequately;
- to establish whether constructed wetlands
    - are an adequate and acceptable form of wastewater treatment,
    - are capable of operating on-site in temperate regions and
    - are economical, in terms of costs and maintenance; and
- to determine the amount of knowledge or information on constructed wetlands possessed by local authorities and their level of acceptance of constructed wetlands as a form of on-site wastewater treatment.

1.2. Perceived Barriers to Constructed Wetlands

As with any new technology, constructed wetlands face many barriers in becoming an recognised and accepted form of wastewater treatment. Listed below are six perceived barriers to constructed wetlands becoming an accepted alternative in the on-site treatment of wastewater.

1. *Technical*: Constructed wetlands are unable to treat wastewater to sufficiently low pollution levels.
2. *Establishment costs*: Constructed wetlands are an uneconomical alternative to current on-site systems.
3. *Maintenance requirements*: The levels of maintenance required by constructed wetlands are too high for domestic use.
4. *Existing systems*: Current on-site systems operate adequately and provide no opening for constructed wetlands.

5. *Lack of information*: There is insufficient information regarding constructed wetlands and as a result they are not being considered as an alternative to on-site wastewater treatment.

6. *Current attitudes*: Local government and community possess negative attitudes towards constructed wetlands.

By fulfilling the objectives outlined, the author will be able to determine if any of these barriers are substantial or are merely temporary impediments to the acceptance of constructed wetlands as a form of on-site wastewater treatment.

The first perceived barrier, 'Technical', is the most critical to the success of constructed wetlands. If the technical aspects of constructed wetlands cannot achieve satisfactory performance levels then all the other barriers become irrelevant, as there is little point in pursuing constructed wetlands for domestic treatment purposes. If barriers 5 and 6, information and attitudes, are proven to exist they can be seen as temporary impediments that can be removed if it is shown that constructed wetlands are a feasible form of on-site wastewater treatment. These barriers can be removed through the release of information and data, promotion/marketing and accessible demonstration sites. 'Establishment costs' and 'Maintenance requirements', perceived barriers 2 and 3, must be kept relatively low, when compared with other on-site systems, if constructed wetlands are to become a viable alternative form of on-site wastewater treatment.

1.3. Methodology

In order to achieve the above objectives a range of techniques were used.

- Literature review: The review briefly covers the problems associated with domestic wastewater, the current forms of on-site wastewater treatment, and provides a thorough insight into constructed wetlands, their treatment levels, processes and design.

- Case study: A constructed wetland was set up for a single household to provide an insight into design, costs, maintenance, limitations, treatment rates, etc.

- Interviews: Representatives from five local government councils were interviewed to gain an awareness of their knowledge of constructed wetlands, to gauge their level of acceptance of and enthusiasm to this form of treatment and their thoughts on current forms of on-site treatment.

1.4. Limitations

Research for this thesis was conducted part-time over a two year period. The initial research and construction of the case study wetland was completed in November 1996. The reeds were planted in December 1996 and, therefore, the reeds did not have a full growing season for testing over winter and spring. Ideally testing and monitoring of the constructed wetland would have included a period when the reeds were fully-grown and hence the system would have been operating at its maximum treatment ability. However, even one year later, December 1997, the reeds and their root/rhizome system were not fully-grown. Consequently, testing was completed before the system had reached its full potential.

The residents of the site provided some money for the materials used in the construction of the wetland as well as valuable time, advice and use of their equipment. The project received no additional external funding and a small budget from the University Department was used for the testing and monitoring of the wetland's inflow and outflow.

These fiscal and time constraints resulted in a less detailed and shortened investigation into the case study's performance than the author would have otherwise desired. These limitations are discussed where applicable throughout the thesis and an indication is given of areas where further studies would be useful in the future.

1.5. Thesis Structure

The structure of this thesis is such that the aim can be achieved through a logical procession of the objectives stated above.

Chapter 1 sets out the aim and objectives of this thesis, briefly describes the subject of on-site wastewater treatment and the perceived barriers associated with constructed wetlands becoming an accepted form of treating greywater.

Chapter 2 focuses on the characteristics of household wastewater and the current methods of treatment. The issues associated with the re-use of wastewater are also briefly discussed.

Chapter 3 introduces constructed wetlands, detailing their components, uses, aspects of design and provides a summary of results from past research.

Chapter 4 focuses on the case study of a constructed wetland operating in Tasmania. It covers the practical issues of design, construction, costs, maintenance and monitoring. Results of the case study are provided and discussed.

The local government interviews are discussed and summarised in Chapter 5.

Chapter 6 draws together all the information to form a conclusion of the possible future for constructed wetlands treating wastewater on-site in a temperate climate.

Chapter 2. Household Wastewater

Households are large consumers of water and consequently, as a whole, large generators of wastewater. Within every household water is essential for many activities associated with everyday living. Water is needed for drinking, preparation and cooking of food, cleaning ourselves, our clothes and the house, and also in the garden. The aim of this chapter is to define household wastewater, outline the sources and various contaminates associated with greywater, and to provide an overview of the current methods of treating wastewater on-site. Some issues connected with greywater re-use will also briefky be touched upon. For a more detailed summary of household wastewater, including composting toilet effluent, refer to *On-site Management of Greywater and Human Wastes* (Marshall 1995).

2.1. Overview of Household Wastewater

Wastewater is split into two categories, blackwater and greywater. Blackwater is all water that passes through the toilet system and becomes contaminated from human bodily wastes. Greywater is the term used for all the wastewater produced that does not originate from the toilet system. Greywater originates from a variety of different sources and therefore has a wide range of physical, chemical and biological contaminants. It is important to stress that greywater does contain human faecal indicator bacteria in concentrations high enough to pose a health risk from the potential presence of pathogenic microorganisms (Jeppeson 1996: ii).

Sources of greywater include the kitchen, laundry and bathroom. The types of contaminates that can be found in greywater include various chemicals, oils, soaps, hair, dead skin, detergents, bacteria and viruses. Greywater composition is not only variable within a household, depending upon its source, but also very variable between households. Households that are conscious of water use and use 'green' products produce less greywater with lower levels of chemical contaminants than households with an apathetic approach do. Education and a conservation ethic are therefore also powerful weapons in decreasing the effects of wastewater on society and the environment.

Table 2.1 shows the breakdown of domestic water use for an average household in Australia. The data in this table are compiled from four studies of coastal cities, Melbourne, Perth, Adelaide and Sydney. Of all the water used in an average household 41% exits as greywater, 17% as blackwater and 42% is used for outdoor purposes such as watering the garden (White 1994 in Marshall 1995: 4). Blackwater can be almost entirely eliminated from a household through the use of composting toilets. Further water savings can be achieved through the re-use of greywater for outdoor purposes such as watering the garden. Theoretically it could be possible for a household to save up to 59% (17% 'Blackwater' and 42% 'Outdoor Use') on water consumption through the use of composting toilets and greywater re-use. Both composting toilets and greywater re-use are briefly discussed below.

Table 2.1: Average[1] Household Water Use in Australian Residences

Type of Wastewater	Domestic Water Use		
	kL/house/yr	**L/house/day**	**%**
Greywater	124	340	41
Blackwater	51	140	17
Outdoor Use	125	342	42
Total	**300**	**822**	**100**

Note: 1- average is 2.9 occupants per household.
Source: White 1994 in Marshall 1995: 4

2.1.1. Greywater Source and Volumes

Greywater not only varies from household to household but also significantly within the period of a day. Mornings, evenings and periods of clothes washing usually produce the peak flows for greywater. Table 2.2 provides the breakdown of different greywater sources and the average volume of greywater that each produces from an average Australian household. Data from this table show that an average Australian household of 2.9 occupants generates 340 L/day of greywater, an average of 117 litres of greywater per person per day.

Table 2.2: Breakdown of Average Greywater Use in Australian Residences

Greywater Source	Average Domestic Greywater Use[1]			
	kL/house/yr	L/house/day	% of Greywater	% of Total Household Water Used
Bathroom	65	178	53	21
Laundry	39	107	31	13
Kitchen	20	55	16	7
Total Greywater	**124**	**340**	**100**	**41**

Note: 1- average is 2.9 occupants per household.
Source: White 1995 in Marshall 1995: 5

2.1.1.1. The Bathroom

On average the bathroom generates 178 litres of greywater per day, 53% of total greywater (Table 2.2). Bathroom greywater is potentially the least chemically contaminated of all greywater sources but has poorer microbial quality than laundry water as discussed below. This is confirmed by Rose et al. (1991, as quoted in Jeppesen and Solley 1994: 13) who state that "total coliform and faecal coliform numbers [are] approximately ten times greater in bathing water than in laundry water". The major contaminants from the bathroom include shampoo, soap, toothpaste, shaving cream, bathroom cleaners, dirt, hair, dead skin, urine and minor faecal matter.

2.1.1.2. The Kitchen

As shown in Table 2.2, kitchen greywater represents 16% of total household greywater and is considered the poorest quality greywater (Jeppesen and Solley 1994: 12). Kitchen sink and dishwasher wastewaters are heavily contaminated with food particles, cooking oils, grease, detergents and cleaning agents. Kitchen greywater promotes the presence and growth of bacteria and other pathogens and Sherman (1991, as quoted in Jeppesen and Solley 1994: 12) states that "the kitchen sink produces wastes of sufficient strength to be considered blackwater". Studies conducted by Brandes (1978) and Karpiscak et al. (1992; both in Jeppesen and Solley 1994: 19) recorded faecal coliform levels per 100 mL of 9×10^5 and 2×10^9 respectively. Greywater from the kitchen is often excluded from greywater re-use systems due to the high levels of faecal coliforms and suspended solids.

Jeppesen and Solley (1994: 12) state that dishwasher water is not the best source of greywater to re-use directly for irrigation purposes. Problems can occur from re-use of dishwasher water because it is too alkaline, the temperature can damage plants and dissolve solidified grease, cause blockages of irrigation systems, and may cause problems with soil hydrology.

2.1.1.3. The Laundry

Laundry greywater from washing machines and laundry tubs/sinks is the second highest source of greywater, 107 litres/day or 31% of total greywater (Table 2.2). Of all greywater sources the laundry has the highest chemical contamination and potentially has the highest microbial quality (Jeppesen and Solley 1994: 14). The level of microbial quality in laundry greywater is dependent upon household composition; if babies are in the household then the method of nappy disposal will greatly influence the quality of the greywater. Other laundry contaminants include detergents, lint, hair, bleach, fabric softeners and dirt. Unfortunately laundry tubs are sometimes used illegally to dispose of chemical substances such as paints, solvents, herbicides and pesticides, which greatly increase the levels of chemical contamination.

2.1.2. Greywater Contamination

If greywater is to be treated or re-used then it is essential to understand its composition to ensure that it is treated to satisfactory levels and thus to minimise the health and environmental risks. Greywater composition involves identifying the contaminants and their concentration levels along with the implications or risks associated with their presence. Greywater contaminants can be split into two categories, microbial contamination and chemical contamination. It is important to note that results from any studies are highly variable due to the greywater source, household composition and habits, socioeconomic factors, climate, and types of cleaners and detergents used.

2.1.2.1. Microbial Contamination

There is a health risk when dealing with wastewater, black or greywater, as water is an ideal medium for the transport of human pathogens. Traditionally, the microbial quality of greywater has been determined by the presence or absence of total and/or faecal coliforms. Coliforms are used as indicators that water has faecal contamination and therefore the potential for the presence of pathogenic microorganisms (Jeppesen and

Solley 1994: 16). Some of the potential sources of pathogens in greywater include (Christova-Boal et al. 1995: 43):

- human body secretions and skin cells emitted during bathing/showering;
- the washing of clothes after an active sport or activity;
- soiled nappies washed in a trough or washing machine;
- clothing or linen washed in a trough or machine that contains vomit or excretions from an infected person;
- family pets washed in the bath or laundry trough; and
- soiled or faecal contaminated shoes (e.g. dog faeces) washed in the bath or laundry trough.

The microorganisms of greatest concern in wastewater are enteric or intestinal pathogens, which include bacteria, viruses (e.g. poliovirus, Hepatitis A, rotavirus, adenovirus), protozoa (e.g. giardia, cryptosporidium) and helminths (worms). The transfer of these pathogens through greywater is by direct contact, ingestion or inhalation of infectious water vapour or droplets, or indirectly through contact with a media (e.g. soil) previously contaminated. There are many factors that influence whether specific doses will be infectious. For example, in the case of enteric viruses several outcomes are possible depending upon a person's existing immunity, age, nutrition, ability to elicit an immune response, and other non-specific host factors (Christova-Boal et al. 1995: 43). Not all individuals who become infected will develop clinical illness. Table 2.3 summarises the infectious doses for various microorganisms.

Table 2.3: Infectious Dose for Some Enteric Pathogens

Pathogen	Probability of Infection from Exposure to 1 Organism	Dose to cause an incidence of 1%
Poliovirus 1	$1.49x10^{-2}$	0.67
Poliovirus 3	$3.1x10^{-2}$	0.32
Echvirus 12	$1.7x10^{-2}$	0.59
Rotavirus	$3.1x10^{-1}$	0.03
Salmonella species	$2.3x10^{-3}$	4.3
Salmonella typhi	$3.8x10^{-5}$	263
Shigella dysenteriae	$4.97x10^{-4}$	20
Shigella flexneri	$1x10^{-4}$	100
Campylobacter	$7x10^{-3}$	1.14
Entamoeba histolytica	$2.8x10^{-1}$	0.04
Giardia lamblia	$1.98x10^{-2}$	0.5

Source: Christova-Boal et al. 1995: 43

Siegrist (1977, in Jeppesen and Solley 1994: 18) also draws attention to the potential health risk of non-enteric organisms which can be discharged into greywater through saliva or skin from someone suffering respiratory or epidermal infection. However, Siegrist concludes that "transmissions of non-enteric organisms in household greywater is not of major concern" due to the very low probability of infection.

Few studies have directly tested for pathogens in greywater. The City of Los Angeles (1992, in Marshall 1995:8) conducted a greywater pilot study where eight greywater sites were tested regularly for 12 months for faecal coliforms and four 'common' disease organisms, *Salmonella, Shigella, Entamoeba hystolitica* and *Ascaris lumbricoides*. Faecal coliforms were undetected in only 7 samples of 95 tested, the remaining samples had a range of 17 to >1.6 x 10^5 and an average of >3 x 10^4 CFU/100mL. None of the diseases were detected in any of the samples. The study concluded that this may have been due to "(1) none of the residents in any of the test sites shed any of these organisms, or (2) disease organisms that may have been present were deactivated in the detergent-laden environment of the storage tank" (City of Los Angeles 1992, in Marshall 1995:8). This report provided inconclusive evidence that pathogens are never present in greywater or that there is no potential for greywater to contain pathogens because there was no proof that any of the residents carried any of the organisms being tested.

At times it is necessary to hold greywater in sullage or storage tanks for later re-use or treatment. In the first 48 hours of storage Rose et al. (1991, in Jeppesen and Solley 1994: 21) found that faecal coliforms increased by a factor of 10 to 100, then decreased slowly after this period. Even after 12 days of storage faecal coliform numbers remained higher than initially tested. The conclusions drawn from these tests are that the physical and chemical properties of greywater, such as high levels of phosphates, ammonia and turbidity, may actually promote the growth of microorganisms. Rose et al. also tested for the survival of *Salmonella typhimurium*, *Shigella dysenteriae* and Poliovirus in greywater. Unlike faecal coliforms there was no regrowth of any of the pathogens. The shigella bacteria declined in numbers immediately, while the salmonella bacteria and poliovirus remained stable for 2-4 days before decreasing. After 8 days there were still significant levels of all pathogens. Rose et al. (1991, in Jeppesen and Solley 1994: 21) conclude that while there was no regrowth for the three pathogens tested this may not be the case for all possible pathogens in greywater and they state that "due to the low infectious dose of viruses, even low concentrations would be of concern".

As stated above faecal coliforms are used as an indicator for the presence of pathogenic microorganisms. However, faecal coliforms are not a perfect indicator of pathogenic contamination in greywater and they are increasingly being questioned for this use because the relationship between faecal coliforms and many human pathogens is not reliable (Grohmann 1995; Logan 1994, both in Marshall 1995: 9; Jeppesen and Solley 1994: 18). However, Jeppesen and Solley (1994: 18) state that faecal coliforms may not be a perfect indicator of pathogenic contamination but they are most likely the best indicator available at the current time. Direct testing for pathogens in water is generally not conducted due to the difficulty and expense of testing (Tchobanoglous and Burton 1991, in Marshall 1995: 9). Therefore, faecal coliforms are the primary and most used indicator of pathogenic contamination for greywater.

The data in Table 2.4 show faecal coliform levels recorded in several studies of greywater and indicate that faecal contamination of greywater does occur and that there is a potential for pathogens to be present in the greywater. The figures in Table 2.4 highlight the variability of greywater, not only from the differing source within studies but also from the same source across the studies.

Table 2.4: Faecal Coliform Concentrations in Greywater Prior to Storage

Source	Faecal Coliforms/100mL			
	Rose et al. (1991)	**State of Calif. (1990)**	**Brandes (1978)**	**Karpiscak et al. (1992)**
Bathing/Shower Water	$6x10^3$	$4x10^5$ (MPN)	<10 to $2x10^{8c}$	
Laundry Wash Water	126	$2x10^3$ to 10^7 (MPN)		
Laundry Rinse Water	25			
Kitchen Wastewater			<10 to $4x10^{6c}$ $9x10^5$	$2x10^9$
Combined Greywater	6 to 80 [a] $1.5x10^3$ [b] $1.8x10^4$ to $8x10^6$		$8.8x10^{5cd}$ $13x10^{6d}$	$1.73x10^5$

Notes: All values are cfu/100mL (colony forming units) except MPN= most probable number

a- families without children

b- families with children

c- other study quoted

d- kitchen and bath only

Source: Jeppesen and Solley 1994: 19

2.1.2.2. Chemical Contamination and Physical Quality of Greywater

The physical and chemical qualities of greywater can have many environmental consequences if not identified and dealt with or treated sufficiently. Greywater can cause a general decrease in the diversity of biota in river and soil systems due to increased levels of pH, turbidity, salts, suspended solids, boron and BOD_5, accompanied with low levels of dissolved oxygen. The high levels of nutrients associated with greywater can cause problems of eutrophication in waterways and have a detrimental effect on certain species of native flora (Christova-Boal et al. 1995: 45). Greywater also has the potential to leach into and pollute groundwater. The chemical quality data for greywater from various studies is shown in Table 2.5.

Table 2.5: Chemical and Physical Quality of Greywater

Parameter	Unit	Value												
		BCC[e]	Rose et al. (1991)		Brandes (1978)		Enferandi[a]	Boyle[b]	Sherman[b]	Siegrist (1977)	Karpiscak (1992)	City of L.A (1992)		Tap Water
		mean	mean	range	mean	range	range	range	mean	mean	mean	mean	range	d
BOD_5	mg/L	175			149	35-245	40-620	125-291	33	260	229, 1489[c]			
COD	mg/L				366	119-870	60-1610	242-622	52		539, 597[c]			
TOC	mg/L				125	30-375								
Suspended Solids	mg/L	120			162	25-510	20-1500	36-160		155	90, 150[c]			
Turbidity	NTU	90	76.3	20-140				42-67			56, 63[c]			<1
Total Dissolved Solids	mg/L	350			528	284-854	420-1700	686-925				861	140-5960	1000
pH	–	7.4	6.54	5-7	6.8	6.5-7.3		7.1-8.7				7.5	5.7-9.9	6.5-8.5
Alkalinity	mg/L		158	149-198	148	125-169		382						131
Hardness	mg/L		144	112-152	39	26-54								142
Electrical Conductivity	:mho/cm	580			443	330-510								
Ammonia	mg/L	5.5	0.74	0.15-3.2	1.7	0.1-8.1		0.6						0
Total Kjeldhal N	mg/L	12	1.7	0.6-5.2	11.3	5.5-18	2-50	5.7-18.4	1.9	17	1.16, 6.68[c]			
Nitrite Nitrogen	mg/L	<0.2			0.04	0.01-0.24								1.0
Nitrate Nitrogen	mg/L	0.3	0.98	0-4.9	0.12	<0.1-0.2		0.1-0.6						10
Sulphate	mg/L	30	22.9	12-40	11	4-19								28.3
Phosphorous	mg/L	8.0	9.3	4-35	1.4	0.8-3.2		0.3-11.9	3.4	23				3.1
Potassium	mg/L				8.9	4.5-13								
Aluminium	:g/L	670			120	20-270								200
Barium	:g/L	45												1000
Boron	:g/L											n.d	n.d	5000
Cadmium	:g/L	<10												5
Calcium	mg/L	30			9	4-18						67	20-824	
Chromium	:g/L	<10												50
Chloride	mg/L		9.0	3.1-12	48	20-88						81	6-136	250
Copper	:g/L	150												1000
Iron	:g/L	790			17500	11000-28000								300
Lead	:g/L	<50												50
Magnesium	mg/L	15			4	1-6								
Manganese	:g/L	40										24	8-235	100
Nickel	:g/L	<15												
Sodium	mg/L	70			76	59-90						120	32-1090	200
Zinc	:g/L	380												5000

a- Source Rose et al (1991)
b- Source City of Los Angeles (1992)
c- Kitchen sink wastewater only
d- A compilation of measured values and various standards/guidelines
e- Source Jeppesen and Solley 1994
n.d- Not detected

Source: Jeppesen and Solley 1994: 23-4

2.1.2.3. Definition of Main Chemical and Physical Qualities

BOD_5 is a five-day test to measure of the quantity of oxygen used by microorganisms in aerobic oxidation of organic matter. Higher BOD_5 levels result in greater oxygen consumption from the aerobic microorganisms, which lower dissolved oxygen levels that other aquatic and soil organisms require to survive (such as fish, insects, microflora etc.).

Turbidity is a result of suspended solids in the water. High levels of turbidity provide a means of protection for microorganisms from disinfection and are therefore a concern in the treatment of wastewater. Both chlorine and ultraviolet light disinfection become less effective as turbidity levels rise in wastewater (Jeppesen and Solley 1994: 22). In natural water systems higher levels of turbidity decrease water's ability to support a diversity of aquatic organisms as suspended particles in turbid waters absorb more heat, increasing the temperature of the water and decreasing dissolved oxygen levels.

Dissolved oxygen (DO) is a fundamental factor in the maintenance of healthy water systems. Most aquatic plants and animals require oxygen dissolved in water to survive (Mitchell and Stapp 1988: 17). However, different aquatic organisms survive or function better at certain levels of DO, for example pike and trout require medium to high levels of DO to live while carp and catfish flourish in waters of low DO. Waters of consistently high DO are usually considered healthy and stable aquatic ecosystems that are capable of supporting many different kinds of aquatic organisms.

Sudden or gradual depletions in DO can cause major shifts in the diversity and abundance of an aquatic ecosystem make-up (Mitchell and Stapp 1988: 17). Organisms may shift from pollution intolerant species to pollution tolerant species. Even slight drops in DO or increases in pollution levels may cause large changes in the make-up of invertebrate and microorganisms, which may then have a rippling affect through the food chain. Nuisance algae and anaerobic organisms may also become abundant in waters of low DO.

Some natural factors that affect DO levels are temperature, flow, season and the physical structure of the river (Mitchell and Stapp 1988: 18). Cold water can hold more oxygen than warmer water because gases are more soluble in cold water. The primary factor that depletes DO is the accumulation of organic wastes. These may enter water systems naturally (e.g. leaves) or through human activities (eg. sewage, industry etc.).

Aerobic bacteria consume oxygen in the process of decomposing or breaking-down these organic wastes.

Water (H_2O) consists of both H^+ ions and OH^- ions. The pH value of water provides a measure of hydrogen ion (H^+) concentration (Mitchell and Stapp 1988: 33). The pH of natural water is usually between 6.5-8.5 and it is in this range that the largest variety of aquatic organisms are found in the range of 6.5-8.5. Organisms are less likely to survive in acidic water than basic water.

Total phosphorous includes organic phosphorous and inorganic phosphate (Kadlec and Knight 1996: 443). Organic phosphorous is attached to particulate organic matter composed of once-living plants and animals. Inorganic phosphates comprise ions (H_2PO_4, HPO_4 and PO_4) bonded to soil particles and phosphates that are present in laundry detergents (polyphosphates). Phosphorous is an essential element of life as it is a nutrient needed for plant growth and a fundamental element in the metabolic reactions of plants and animals. In most waters phosphorous is usually present in very low concentrations. Any 'unattached' or free phosphorous in the form of inorganic phosphates is rapidly taken up by algae and larger aquatic plants, and therefore excess phosphorous can cause eutrophication because algae only require small amounts of phosphorous to live.

Nitrogen is another nutrient that can cause eutrophication problems in natural waterways. In its molecular form nitrogen is one of the most common elements in the atmosphere and dissolves easily into aqueous systems.

2.1.2.4. Sources of Chemical and Physical Qualities

Food scraps, cooking oils and grease from the kitchen raise the levels of BOD_5, suspended solids, nutrients, faecal coliforms and odours in greywater (Tchobanoglous and Burton 1991, in Marshall 1995: 12). Hair, lint and dirt add suspended solids to greywater and can be quite slow to break down.

Urine consists of 15-19% nitrogen and 2.5-5% phosphorous and can contribute to the high nutrient levels in greywater (Gotaas 1956, in Marshall 1995: 12). Faecal matter is only present in small quantities but is the greatest potential source of human pathogens.

Detergents, especially laundry detergents, are the primary source of chemical contamination in greywater and greatly increase turbidity (Marshall 1995: 12).

Detergents are significant contributors of phosphorus, sodium (salts), boron, chlorine and other chemicals. The composition of the detergent will greatly affect the quality of greywater. Christova-Boal et al. (1995: 24) used three different commercial detergents in a greywater irrigation study and found that greywater quality parameters varied greatly between brands; the important parameters are summarised in Table 3.6. The products used were:

1. "Cold Power"- a powder detergent;
2. "Bio-Z"- another powder detergent that contains enzymes and zeolite and is claimed by the manufacturer to be phosphate free and fully biodegradable; and
3. "Pure Laundry Detergent"- a liquid based detergent based on 0.5% Potassium Citrate which was supplied as a specially designed product to be tested for greywater re-use; it is referred to as Potassium Based Detergent (PBD).

Table 2.6: Detergents and their Influence on Greywater Quality

Detergents	Cold Power	Bio-Z	PBD
Parameters	Range	Range	Range
pH (units)	7.4 - 10	7.2 - 9.4	6.3 - 7
EC 25C (:S/cm)	320 - 1400	190 - 480	83 – 380
TDS, (mg/L)	204.8 - 896	121.6 - 307.2	53.12 – 243.2
Sodium, (mg/L)	65 - 480	49 - 150	12 - 61
SAR	7.22 - 37.3	4.4 - 9.27	1.33 - 5.07
Phosphorous (mg/L)	3 - 42	0.062 - 4.4	0.1 - 0.63
Boron (mg/L)	<0.1 - 4.4	<0.1 - 0.1	<0.1 - 0.3
Aluminium (mg/L)	<1.0 - 1.2	14 - 96	<0.1 - 9.4

Source: Christova-Boal et al. 1995: 25

Christova-Boal et al. (1995: 25) does not recommend irrigation with greywater generated from Cold Power because it was too strongly alkaline, high in sodium, saline, boron, and phosphorous. Greywater produced with Bio-Z also had relatively high pH levels, but had lower and comparatively safe levels of salinity, sodium, phosphorous and boron. As seen in Table 3.6, the PBD provided better test outcomes to virtually all the greywater quality parameters and is considered the most suitable of the three detergents for greywater irrigation re-use. This study also shows how the choice of detergents (and other cleaning agents) can greatly affect the quality of greywater and how 'green' products should be the first choice for consumers who wish to make less of an environmental impact.

Sodium salts are used in laundry powder detergents for bulking agents and are seen to have little value in the overall wash performance (Patterson 1994, in Marshall 1995: 13). Sodium salts increase the sodium absorption ratio (SAR) which has a detrimental effect on soil structure.

Soaps, shampoos and toothpaste are all highly diluted by shower and basin water and do not add significant pollutants to greywater (Marshall 1995: 14).

Household cleaners, especially in the form of bleach and chlorine cleaners, contain highly toxic chemicals that can kill beneficial bacteria in treatment systems, and adversely affect plant growth if greywater is re-used for irrigation (Marshall 1995: 14).

As mentioned above other chemicals can be dumped illegally down drains and can be highly toxic and poisonous.

2.2. On-site Wastewater Treatment

As seen above household wastewater is of a variable nature, dependent upon the source of the wastewater (i.e. kitchen versus bathroom) and the make-up and philosophies of the household. Many areas, especially rural and rural-residential zoned areas, are not connected to centralised sewerage systems and must therefore treat wastewater on-site. On-site wastewater treatment systems can be defined as technology that treats and disposes of wastewater, generated by a household, entirely within the confines of the owner's immediate land. It is important that on-site systems are designed, installed, operated and maintained to ensure that the following objectives are met on a sustainable basis (NSW Dept. of Health et al. 1996: 10):

- "*Prevention of Public health Risk:* Unacceptable public health risks must not occur.
- *Protection of Lands:* Land quality deterioration through chemical or biological contamination, or degradation of soil structure must not occur.
- *Protection of Surface Waters:* Surface waters must not be contaminated by any flow emanating from the site, including first flush run-off, contaminated surface or sub-surface flow or contaminated groundwater.
- *Protection of Groundwaters:* Groundwaters must not be contaminated.

- *Resource Utilisation:* The useful resources in domestic wastewater, including nutrients, organic matter and water should be identified and utilised to the maximum extent possible within the bounds posed by the other performance objectives.

- *Community Amenity*: Unreasonable interference and nuisance to the public, due to odour, dust, insects, and noise above normal background levels must be avoided."

An ideal on-site system could be seen to have the following characteristics:

- Low capital (set-up) costs;

- Easy and practical construction and installation;

- Low operation and maintenance demands, in terms of labour/time and costs;

- Provision of effective wastewater treatment i.e. is capable of achieving the required health standards; and

- Adaption to a range of climates and conditions possible.

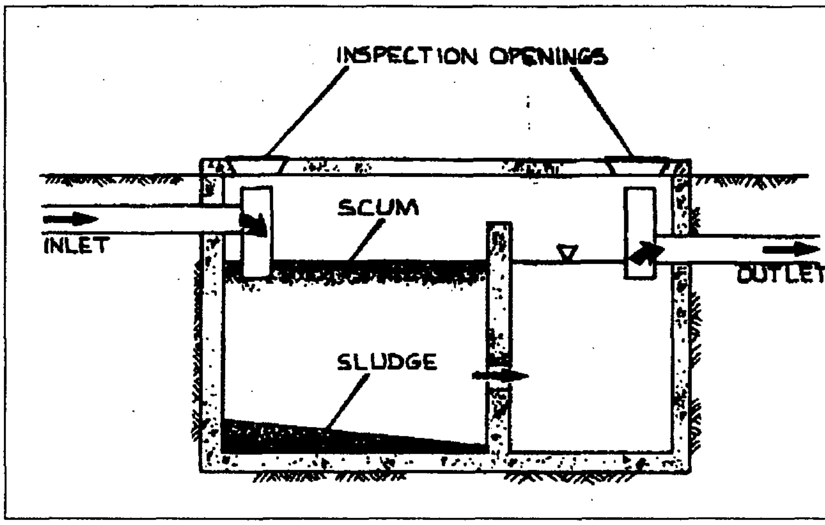
2.3. Current Forms of On-site Wastewater and Waste Treatment

The three most common forms of on-site wastewater treatment, septic tanks, AWTS and composting toilets, are discussed below in regard to operating and maintenance procedures, costs and advantages and disadvantages.

2.3.1. Septic Tanks and Soil Absorption Systems

The most conventional form of on-site wastewater treatment in Australia is the combined use of a septic tank and soil absorption trenches (ST/SAS). Septic tanks are designed to treat wastewater, both greywater and blackwater, to a primary level. Septic tanks are generally divided into two chambers, an initial settling chamber (two-thirds of the overall length) and a final settling chamber (Figure 2.1). Single chamber septic tanks do not provide the same level of treatment due to poor settling arising from turbulence and water oscillation created by incoming wastewater (Petrozzi and Martens 1995: 14). Household wastewater enters the first chamber of the septic tank where floatable solids form a scum layer and heavier solids sink to the bottom forming a sludge. It is in this chamber that a constant anaerobic process partially breakdowns the contaminates associated with wastewater, sludge/solids, nutrients, pathogens and viruses. As the solids settle and the scum layer is separated, effluent flows into the second chamber where further settling occurs. The effluent exits the second chamber and is disposed of through soil absorption trenches.

Figure 2.1: Septic Tank System



Source: Australian Water Resources Council 1983b: 24

Septic tanks can achieve the removal of approximately 40-60% BOD, 50-70% suspended solids, 10-20% nitrogen, and 30% phosphorous (Laak 1986 in Geary and Gardner 1996: 2). Table 2.7 contains average values for treated septic tank effluent from two different studies. The data indicate that septic tank effluent contains reasonably high levels of suspended solids, BOD_5, total phosphorous, total nitrogen and ammonia concentrations, and high faecal coliform counts.

Table 2.7: Septic Tank Effluent Quality

Parameter	Average Value	
	Canter and Knox[1] (No. of Samples)	Laak[2]
Suspended Solids	49 (148)	200
BOD_5	138 (150)	200
COD	327 (152)	500
Ammonia-N	31 (108)	-
Nitrate-N	0.4 (114)	-
Total-N	45 (99)	40
Ortho-phosphates	11 (89)	-
Total-P	13 (99)	20
pH	-	7.2
Faecal Coliforms cfu/100mL)	5.0×10^6 (7)	10^8
All concentrations in mg/L except for faecal coliforms.		

Source: 1- Canter and Knox 1988: 53-6
2- Laak 1986 (in Rawlinson 1994: 20)

Septic tank effluent is disposed of and further treated by soil absorption trenches. Soil absorption trenches have the processes of "infiltration; dilution; filtration; biological treatment through the action of plants and bacteria; contaminant absorption; precipitation and evapo-transpiration" to further treat wastewater (Petrozzi and Martens 1995: 19). Depending upon the soil type, level of effluent contamination and hydraulic loading, it is possible that contaminants can be removed from wastewater to required levels in a soil depth of 0.6-0.9 metres (Australian Water Resources Council 1983a: 246). This removal of contaminants is not instant, in temperate climates, such as Tasmania; with ideal conditions it is estimated that bacterial pathogens are limited to 2-3 months survival (Gerba et al. 1975, in Australian Water Resources Council 1983a: 245). An important component of trenches is the "slime layer" or "biological mat" that forms on the trench bottom and walls. This mat offers hydraulic resistance to effluent infiltration and Perkins (1989, in Petrozzi and Martens 1995: 20) argues that the filtration in this layer accounts for 95% of the contaminant and pathogen attenuation.

A site assessment should be made in order to determine whether or not the physical characteristics of the land are suitable for the installation of a ST/SAS. Unfavourable sites include: shallow coarse soils or clay based soils; a high level of ground water; excessive rainfall; and a steep slope. These poor site conditions either result in the pooling of wastewater on the soil surface or the wastewater rapidly passing through an ineffective soil absorption and filtration process, which does not adequately treat the effluent, causing ground and surface water pollution. Even if growing townships do possess soil systems that are adequate for septic tanks, a point of saturation can be reached where the land will be unable to adequately filter all the wastes. Poor siting and saturated soils result in the contamination of ground and surface water by pathogens, viruses and nutrients. All are obvious health risks but the nutrients have the added concern in that they encourage eutrophication. Several variations to the trench systems have been devised to increase reliability or overcome a site's limitations. These variations include: mound systems where high ground water exists; evapotranspiration systems used where soils have low permeability; absorption beds rather than trenches where land is limited; serial distribution on sloping land; and the attachment of sand filters before effluent is released into the soil. These variations provide alternative designs for the disposal of septic tank effluent at sites with unfavourable characteristics.

The cost of a septic tank is dependent upon the site to be installed. In an ideal site which has no or little slope and good soil the cost for a ST/SAS is about $2,000-$2,500, while an unfavourable site may cost up to $5,000 (pers. com. Robertson 1997). The actual septic tank costs around $700 and absorption trenches cost approximately $50/metre.

2.3.1.1. Maintenance of Septic Tanks

Septic tank systems should be inspected annually for sludge and scum build-up and pumpouts should be undertaken every two to three years (Crennan 1992: 63). Pumpouts cost between $110-130. The wastes from the pumpout should be disposed of in an ecologically sound manner. If pumpouts are not regularly performed then the sludge and scum will exit the chamber and enter the absorption trench causing clogging and reduced effluent infiltration. Theoretically, if the trenches are of sufficient length, the clogging layer in a trench will reach a state of equilibrium between build up and break down and allow long-term soil absorption. Equilibrium will remain undisturbed provided pumpouts are regular and that hydraulic overloading does not occur from excessive use of household water appliances (e.g. washing machines or lengthy showers). Other maintenance and operation taboos are the use of household bleaches and other chemicals that can slow or stop the anaerobic process.

2.3.1.2. System Performance

A well maintained and appropriately designed and sited ST/SAS is an inexpensive, viable and valuable option for many on-site treatment areas. Septic tanks are a valuable on-site system but have been known to fail due to unsuitable site conditions, poor design and construction, and inadequate operation and maintenance. Failure of ST/SAS refers to the inability of systems to: contain effluent within property boundaries; limit human health risks; prevent the degradation of environmental; and water resources (Petrozzi and Martens 1995: 22). It is estimated that between 40% and 60% of septic tank systems fail, that is they do not function in the proper manner (Canter & Knox 1988: 54; Henery et al. 1987, in Petrozzi and Martens 1995: 22). Binnie and Partners (1988, in Rawlinson 1994: 21) state that the failure of ST/SAS systems can usually be attributed to:

- "inappropriate site condition (i.e. poor soil permeability, excessive rainfall, steep slope);
- excessive hydraulic loading, either because the system is undersized, or because of excessive water use;
- premature clogging- due to carryover of solids from improperly maintained septic tanks;
- poor design and/or careless installation."

A comprehensive study on septic tank performance was conducted by O'Neill et al. (1993, in Rawlinson 1994: 21) in NSW. One aspect of the study was the assessment of system failure by visible surface flow from the trench; the results are shown in Table 2.8. Of 200 septic tank systems surveyed 93 (47%) had visible surface flow and were therefore considered failing. It must be noted that these are only visible failings and do not include failures that cannot be seen, such as groundwater contamination. In another study of septic tank performance a total of 118 systems were surveyed in two different South Australian communities (Rawlinson 1994: 22). This study found that 73% of the septic tanks were performing unsatisfactorily. The results of both these studies are of concern due to the high level of system failures. In both of these studies the authors state that overloading was the major cause of system failure.

Table 2.8: Survey of Trench Failures in NSW

Location in NSW	No. of Systems Surveyed	Trench Failure with Visible Surface Flow (%)
Wollondibby, Cobbin & Mowamba Catchments, Jindabyne	35	45
East Jindabyne	68	41
Ammerdown & Clifton Grove, Orange	22	59
White Gate & Mt Pleasant, Bathurst	15	53
Perthville	9	22
Valley View Close & Glanmire, Evans Shire	6	100
Hill End	9	78
Nundle	8	38
Kingswood Estate, Tamworth	8	38
Moruya	12	42
Fairhaven	3	66
Beauty Point	5	20

Source: O'Neill et al. 1993, in Rawlinson 1994: 21

Poor design, construction, operation and/or maintenance can also lead to the failure of septic tanks. The construction of soil absorption trenches at appropriate lengths is essential in order to provide adequate unsaturated effluent filtration. The average length of a trench that is required for long-term performance is around 20 metres for sandy soils and 55 metres in clayey soils. These lengths make it hard for owners with limited land to comply, but research by the Australian Water Resources Council (1983b: 20) discovered that many trenches in Australia (they provided no figures) are less than the required lengths. The soil's ability to remedy wastewater contaminants is decreased because less soil must filter greater quantities of effluent.

Although there can be a lot of problems associated with septic tanks the proper siting, design, construction, operation and maintenance can provide a more than satisfactory and a relatively cheap on-site option.

2.3.2. Aerated Wastewater Treatment Systems (AWTS)

AWTS are small-scale treatment systems that treat household wastewater to a secondary level and, usually, provide disinfection of the resulting effluent. While there are significant generic differences in the design and operation of available AWTS, Petrozzi and Martens (1995: 26) state that all systems include the following treatment stages:

- Primary treatment;

- An aeration phase to produce bio-chemical oxidation and consumption of organic matter;
- Clarification and chlorination; and
- Disposal of resulting effluent through irrigation.

The first stage of treatment occurs in the primary sedimentation tank where floatable solids and oils form a scum layer and heavier solids form a sludge, essentially the same process as occurring in a septic tank. The effluent then flows into an aeration tank where air is forced through the wastewater. Aeration encourages further breakdown of the sewage by aerobic bacteria. After aeration the effluent flows into the clarification chamber where sludge, suspended solids and scum are further removed by sedimentation. In order to reduce a build up of solids in this chamber all the accumulated sludge and scum are returned by pump to the primary chamber. The final chamber chlorinates the effluent to give a free or residual chlorine concentration of 0.5 to 2.0 mg/L in order to provide disinfection of any organisms still surviving in the effluent. Martens and Warner (1991, in Petrozzi and Martens 1995: 27) state that a contact time of 30 minutes is necessary for effective disinfection. The disinfected effluent is then disposed of through sub-surface or surface irrigation.

Figure 2.2: Aerated Wastewater Treatment System

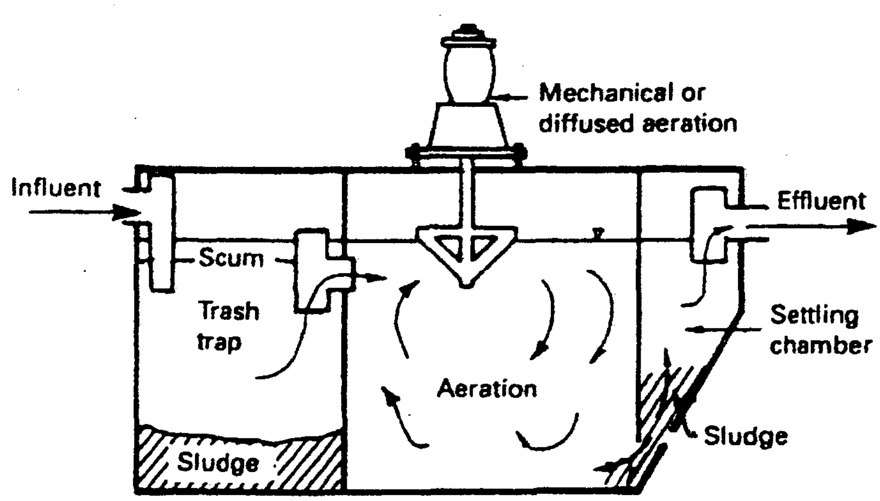


Table 2.9 compares the characteristics of effluent from AWTS with the Australian Standard's permitted levels for effluent re-use through surface irrigation. As can be seen there is quite a range within each of the parameters for AWTS effluent. This range will

depend upon how well the AWTS is operating and maintained and the loading rates. Standard AWTS are not specifically designed to reduce nutrients so this makes the effluent ideal to re-use through irrigation so that the nutrients can be utilised by vegetation.

Table 2.9: Effluent Quality- from AWTS and Required Levels for Irrigation

Parameter	AWTS (mg/L)[1]	Quality for Effluent Surface Irrigation: AS 1547 (1994)
BOD_5	5-80	20
Suspended Solids	5-100	30
Faecal Coliforms (FCU/100mL)	$10\text{-}10^3$	10
Free Residual Chlorine		>= 0.5 ppm
Total Nitrogen as N	50-60	-
Total Phosphorous as P	7-12	-

Note: 1- Beavers 1993, in Rawlinson 1994: 32

The cost of installing an AWTS and the necessary irrigation equipment is relatively high, approximately $8,000 with annual maintenance bills between $450-$550 (pers. com. Robertson 1997). AWTS require a reliable power supply and electricity must also be factored into the systems costs. AWTS are relatively compact and reduces household water consumption by re-using the wastewater for irrigation.

2.3.2.1. Maintenance of AWTS

In Tasmania quarterly maintenance is required for most AWTS in order to ensure that the wastewater is being treated to required levels. Maintenance involves the checking of all moving parts and the replacement of chlorine tablets. Failure to adequately maintain an AWTS can result in turbid wastewater with high faecal coliform counts, electrical faults, pump failure, and irrigation line blockages (Crennan 1992: 65). If a malfunction occurs most units have a capacity to operate for a few days but the quality of the effluent deteriorates rapidly.

As the primary tank operates like a septic tank regular pumpouts, one to two times per year, are required to ensure that no sludge or scum enters the aerobic chamber. Observation of irrigated areas is also required to ensure that sections of the garden do not become saturated and cause plants to die and destroy soil structure.

2.3.2.2. System Performance

AWTS do not cope well with intermittent use and 'shock loading', which result in reduced retention times of wastewater within the system (Petrozzi and Martens 1995: 33; Rawlinson 1994: 30; Geary and Gardner 1996: 5). Reduced retention time produces a poor quality effluent characterised by high levels of BOD_5, nitrogen and faecal coliforms. Toxic loads can also cause problems within AWTS due to the death of microbial populations, which do a great deal of the decontamination work.

Surveys of the disinfection performance of AWTS show that a high percentage of systems failed to meet the faecal coliform and residual chlorine requirements. The results of a study by Roser (1992, in Rawlinson 1994: 32), Table 2.10, show that of 90 AWTS tested 54% exceeded 30 CFU/100mL (20 CFU above the Australian Standard for effluent re-use) and 70% did not achieve the chlorine disinfection level of 0.5 ppm. Another study that randomly selected 27 AWTS (out of a total of 127 operating in the Campbelltown area at the time) had similar results. Only 50% achieved the faecal coliform limit of 30 cfu/100mL and only 1 of the 27 achieved the residual chlorine level (Khalifé and Dharmappa 1996: 26-7). This study also tested AWTS effluent for BOD_5 and SS levels and found that very few satisfied the required limit. These results are of concern as there is a reasonably high percentage of systems failing, which raises questions as to the reliability of AWTS.

Table 2.10: Disinfection Performance of AWTS

Parameter	Wollondilly and Blue M'tain Shires	Wyong Shire	Glenning Valley, Wyong	Totals
No. of Systems Tested	42 (100%)	36 (100%)	12 (100%)	90 (100%)
Samples Exceeding 30 CFU/100 mL	29 (69%)	13 (36%)	7 (58%)	46 (54%)
Samples NOT Achieving 0.5 ppm residual chlorine	31 (74%)	27 (74%)	5 (42%)	63 (70%)

Source: Roser 1992, in Rawlinson 1994: 32

There have also been failings in the disposal of effluent through irrigation. The effluent should be irrigated over a sufficient area and take into consideration soil type, slope and vegetation. Rawlinson (1994: 30) states that it is important to check the irrigation area regularly for evidence of ponding, run-off and the health of vegetation. An example of

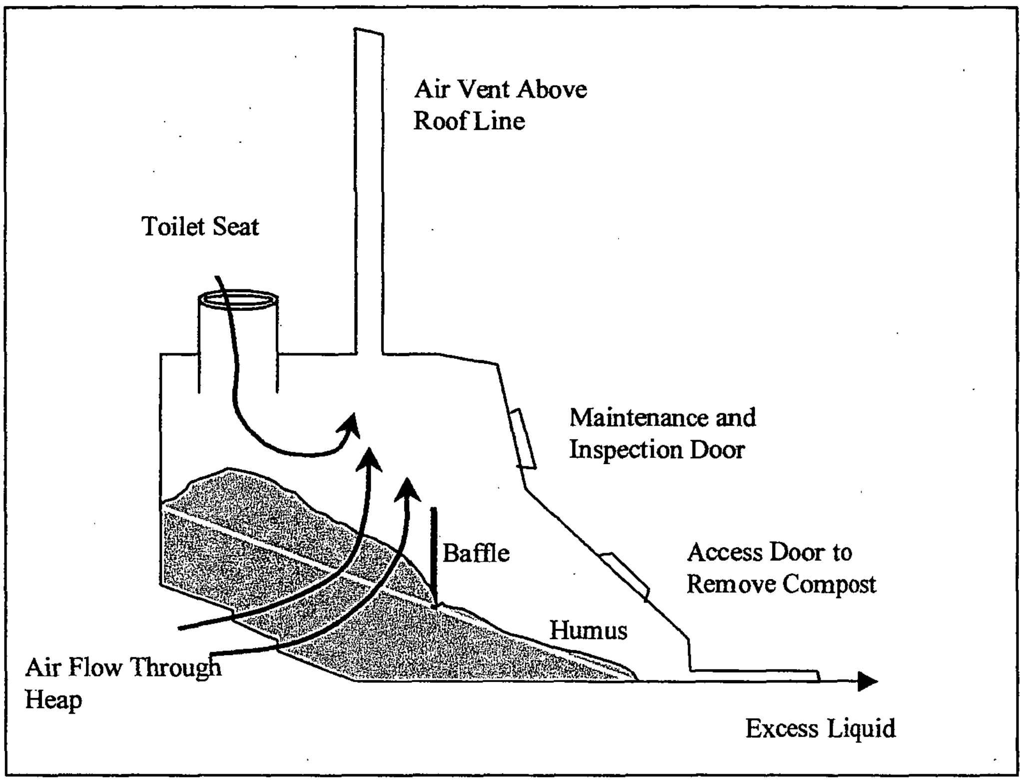
AWTS failure with irrigation was highlighted in the Mercury (Bester 1997: 11) where a family in the Derwent Valley paid about $8,000 to comply with council regulations only to have had large parts of their garden "swamped", killing dozens of plants. The article goes on to state that a neighbour is having a similar problem with their AWTS. Saturated soil also has the potential to contaminate ground and surface water. The council believes that these saturation problems caused by irrigation can be remedied through the use of water loving plants.

2.3.3. Composting Toilets

Composting toilets are a waterless alternative to treating raw human excrement. They, as the name suggests, use a composting process to breakdown faeces into humus through the activity of microorganisms. Users defecate and urinate in a chamber that is designed to minimise contact of the contents between humans and the environment (Crennan 1995: 11). As a composting process is required for the treatment of excrement it is essential to add a bulking agent or carbonaceous material and to maintain sufficient aeration and moisture.

The two main types of composting toilets are a continuous system and a batch system. The continuous system involves only one chamber and fresh excrement is added at the top end of the chamber and the end product, compost, is removed from the bottom end of the chamber (Figure 2.3). Theoretically there is a continual process of addition and removal from the chamber. The batch system makes use of two or more chambers or containers that are used alternatively. As one container is filled it is removed from use and replaced by an empty one. The full container is allowed to compost without any further additions of excrement, eliminating the risk of further contamination.

Figure 2.3: Continuous Flow Composting Toilet Design



The composting process has been shown to be very effective at eliminating viruses, pathogenic bacteria and protozoan cysts. Safton (1996: 3) undertook a study aimed at determining if intestinal parasites and commensal (non-pathogenic) organisms could be recovered from the humus (i.e. the end product of the composting toilet system). The study investigated seven composting systems over a 16-month period, four commercial units and three home built units. Faecal samples were taken from the humus, the end product. Sample were also taken from toilet users and from the top of the chamber pile in order to determine whether or not parasites and/or commensal organisms were being introduced into the toilet systems. Safton established that both parasites and commensal organisms were being introduced into some of the units Some of the microorganisms include *Blastocystis hominis*, *Dientamoeba fragilis*, *Endolimax nana*, *Entamoeba coli*, *Entamoeba histolytica* and *Enterobius vermicularis* (pin worm). The results of Safton's study, shown in Table 2.11, found that of 118 humus samples analysed no parasites or commensal organisms were found in any of the seven composting toilets.

Table 2.11: Summary of Analysis of Parasites and Commensal Organisms in Composting Toilets

Composting Unit	No. of User and Top of Chamber Sample Taken	No. of Positive in User and Chamber	Positives Tests in Humus
Unit A	57	38 (67%)	0
Unit B	44	27 (61%)	0
Unit C	43	17 (40%)	0
Unit D	22	4 (18%)	0
Unit E	9	0	0
Units F & G	-	-	0
Totals	194	86	0 (n=118)

Source: Safton 1996: 4

Because composting toilets are waterless the household consumption of water is dramatically reduced as flush toilets account for 35-41% of water use in an average house (Crennan 1992: 69). Not only does this represent a substantial water saving it also results in the separation of excrement from wastewater, as only greywater is produced. This allows for a better treatment of greywater as a smaller volume has to be treated and it is not as contaminated as if the two streams of wastewater were mixed. Composting toilets produce moisture from urine and the composting process, and this liquid is collected from the base of the chamber to avoid moisture build-up and anaerobic conditions. Only a small volume of liquid is produced and can easily be treated with greywater or separately, via a liquid absorption area.

Commercial composting toilet units cost between $2,500-$3,000 (pers. com. Robertson 1997).

2.3.3.1. Maintenance of Composting Toilets

Compared to the conventional 'flush and forget' toilets, the maintenance requirements of composting toilets are relatively high but once users become familiar with the 'needs' of composting toilets maintenance levels are quite low. A composting toilets 'needs' comprise of maintaining an appropriate carbon: nitrogen ratio and water: aeration ratio. For the composting process to work these ratios must be kept in check otherwise the pile may become either anaerobic and smelly or dry with little moisture and nitrogen for the microorganisms to feed off. Bulking agents, such as woodchips, sawdust, paper or cardboard, need to be added to the chamber on a regular basis to maintain carbon levels and provide aeration.

After 12 months the humus can be removed and used in the garden or added to a compost pile for further breakdown (Crennan 1992: 72). The humus can be used to benefit the garden by supporting beneficial soil microbial populations, balancing soil pH, through the slow release of nutrients to plants and by improving the soil structure with the addition of organic matter. On many systems a fan is used to enhance the flow of air through the compost pile. This should also be checked on a regular basis to ensure that it is operating properly.

2.3.3.2. System Performance

Problems that can occur with composting toilets are foul odours and fly and vermin infestation (Petrozzi and Martens 1995: 39; Rawlinson 1994: 35). These problems occur due to poor design and/or insufficient maintenance of the system. Toxic loads of cleaning agents and insecticides will kill organisms responsible for the composting process. During these periods, before the microorganisms build-up again, wastes accumulate and foul smells are experienced. User education is essential for the proper operation of composting toilets.

Composting toilets are widely used in remote pristine environments, such as Tasmania's World Heritage Area, because they provide a potentially environmentally sound method of dealing with sanitation in sensitive areas. Crennan (1995: 28, 89-90) surveyed nine composting toilets on Tasmania's Overland Track in the World Heritage Area and found that seven were "malfunctioning" and had an end product that consisted of a "wet dense mass of predominantly undecomposed excrement, toilet paper and bulking agent". Crennan (1995: 31, 91) states that the problems occurred from infrequent maintenance and poor design. The problems associated with infrequent maintenance arose from the area's remoteness and the use of the toilets by bushwalkers without an awareness of composting toilet requirements. Systems that are installed in Tasmania's World Heritage Area need careful and thoughtful design to take into account the technical, maintenance and climatic demands associated with these areas in order to minimise the possibility of system failure.

As with all on-site systems, if composting toilets are well designed and maintained they provide a very effective form of treatment with the added benefits of being waterless.

2.4. Greywater Re-use

The principle behind greywater re-use is that water that has been used within a household is used again, or recycled, for other activities around the household. In the Guidelines for Re-use of Wastewater in Tasmania the Department of Environment and Land Management stress that effluent re-use should not be viewed as "disposal" but rather it should seen "as the use of a secondary resource which would otherwise be wasted" (DELM 1994: 5). In these guidelines DELM states that the aim should be to "economically treat and re-use wastewater without endangering public health, cause pollution of groundwater or surface waters, or contribute to long-term land degradation."

The guidelines (DELM 1994: 10) mainly refer to the secondary treated effluent from municipal treatment sites but the effluent uses and quality can be used as a guide for on-site re-use of wastewater. Table 2.12 outlines the different applications, irrigation methods and faecal coliform levels for effluent re-use. For an on-site system and user the most common types of greywater re-use from Table 2.12 would be for the irrigating of landscaped areas and potentially fruit and vegetable crops. Australian Standard 1547 sets out the requirements for disposal systems of effluent for domestic premises. The permissible levels set out by the Standard for surface irrigation are summarised in Table 2.9.

Greywater re-use has the benefits of decreasing the potable water demands and associated costs to the household, and decreases the hydraulic load into the household's on-site treatment system. As discussed above, excessive hydraulic loading is one of the main reasons for septic tank and AWTS failure. By decreasing the wastewater inflow of the household's on-site treatment system less pressure will be placed upon that system, thereby increasing the system's performance levels and hydraulic retention time, lengthening the lifespan and decreasing maintenance costs and demands.

Table 2.12: Applications, Irrigation Methods and Requirements for the Use of Treated Wastewaters

Applications	Acceptable Irrigation Method	Faecal Coliform Level[1]	Provisos
Crops for human consumption which will be commercially processed [2]	Furrow or trickle	Geometric Mean <300 Upper Limit <2000	Processing system approved by the CG [3]
Crops for human consumption which will be cooked before being eaten [2]	Furrow or trickle	Geometric Mean <300 Upper Limit <2000	Processing system approved by the CG. *Irrigation of foods to be consumed raw is not permitted.*
Grasses and Landscaped areas (includes golf and race courses etc.)	Spray	Geometric Mean <750 Upper Limit <5000	Public excluded during any spraying operation
Pasture lands for growth of fodder crops	Spray	Geometric Mean <3000 Upper Limit <14000	Public excluded during any spraying operation and crops not to be harvested within 10 days
Pasture for sheep, cattle, horses and other grazing animals (excluding dairy animals)	Spray	Geometric Mean <3000 Upper Limit <14000	Public excluded during any spraying operation and animal excluded for 10 days
Orchard and vineyard crops for human consumption	Furrow or trickle	Geometric Mean <3000 Upper Limit <14000	Dropped fruit not to be harvested for consumption
Forest areas and areas being rehabilitated after mining or quarrying	Any	Geometric Mean <3000 Upper Limit <14000	Public excluded during any spraying operation
Dust suppression on roads or on coal stockpiles in isolated areas	Spray	Geometric Mean <3000 Upper Limit <14000	Public excluded during any spraying operation
Make-up water for sewer flushing or coal washeries	N/A	Geometric Mean <3000 Upper Limit <14000	None

Notes: 1- Measured in faecal coliform organisms per 100mL. The geometric mean and upper limit are to be calculated from the results of five samples collected at half-hourly intervals.

2- For cereal crops such as wheat, which would not normally be irrigated prior to harvest, Level B applies.

3- CG= Coordinating Group set to foster a whole-of-government approach to re-use.

Source: Adapted from DELM 1994: 10-11

Jeppesen (1996: 96) states other benefits from greywater re-use as:

- reduced costs through the delay of capital works (such as additional dams, reservoirs, pumping stations etc. if dealing with reticulated water systems) as well as reducing costs for treatment, pumping and maintenance;
- reduced wastewater volumes discharged to sewerage system (if on reticulated system), providing associated savings in capital works, treatment, pumping and maintenance;
- decreases in nutrient levels flowing into water ways;

- decreases in peak potable water demand due to irrigation demand being met by greywater re-use;
- potentially greener and more lush environment due to improved irrigation and nutrients in greywater; and
- householders' perception that they are contributing towards helping the environment.

Greywater may be treated or untreated before re-use. As stated above, greywater contains human faecal indicator bacteria in concentrations high enough to pose a health risk. The best way to minimise the chance of disease is to remove or destroy these micro-organisms but if the level of treatment is not satisfactory, as can occur in AWTS, or if the greywater is untreated then it is necessary to prevent any human contact with the greywater. Surveys in the USA and Australia have shown that 60-80% of on-site domestic wastewater treatment systems are not adequately maintained (Jeppesen 1996: 108). Jeppesen and Solley (1996: 51) believes that these unmaintained treatment systems may pose a potentially worse health hazard than untreated wastewater because users expect the effluent from the unmaintained systems to be treated to a satisfactorly standard and can therefore be less cautious in their use of it. The safest method of greywater re-use is one that prevents contact between greywater and humans. Subsurface irrigation is considered the safest method as there is a protective layer of soil stopping human contact and aerosols from escaping into the air.

Other major concerns of greywater re-use are the potential to damage soil structure from the many chemicals and salts found in greywater and the contamination of groundwater and waterways due to leaching and runoff (Jeppesen 1996: 109).

Chapter 3. Wetlands

Chapter 2 highlighted the characteristics and problems associated with wastewater and the current methods of treating wastewater on-site. The aim of this chapter is to introduce a new form of on-site wastewater treatment, constructed wetlands. This chapter will provide an insight into constructed wetlands by outlining the components associated with wetlands, the types of constructed wetlands, treatment processes, and general design considerations.

3.1. Natural Wetlands

Natural wetlands consist of very complex and productive ecosystems that sustain a wide variety of organisms. Natural wetlands include various stages in the food chain, ranging from bacteria, algae, zooplankton, crustacea through to fish and bird life. Wetlands are difficult to define as they are basically a transition zone between permanently wet and permanently dry ecosystems. The Hawkesbury Nepean Catchment Management Trust (Mitchell 1996: 1) defines wetlands as areas where:

- the surface soil or artificial substrata are periodically saturated or flooded to a shallow depth;
- saturation or flooding may be cyclic, intermittent, or permanent;
- water may be fresh, brackish, or saline; and
- saturation or flooding determines soil/substrata conditions and the types of plants and animal communities.

Natural wetlands have traditionally been viewed as a hindrance to 'progress and development' and consequently many have been drained and filled in. This is illustrated in a study by Oates (1994: in Mitchell 1996: 1) which found that more than 70% of Victoria's shallow and deep water wetlands have disappeared. The remaining natural wetlands cover just 2% of Victoria but support more than 75% of the state's amphibians, 12 species of reptiles, 11 species of rare and threatened waterbirds, 108 species of birds which require wetlands for their life cycle and more than 30% of the state's rare, endangered, and vulnerable plants species.

In their struggle between terrestrial and aquatic environments, wetland plants have developed unique qualities which make them particularly useful in treating wastewater. An important aspect of natural wetland systems is that they are pulse driven with daily, seasonal and/or sporadic variations. Due to these inherent fluctuations, wetlands possess an internal ecology that is characterised by robustness, resilience, and a resounding agility and vigour, enabling them to cope with a huge variety of changes and variables. Technology and knowledge now allow the artificial replication of wetland systems.

3.2. Function of Constructed Wetlands

In the past natural wetlands have been used as a form of treatment for wastewaters. Nowdays this practice is rarely used because natural wetlands are rare habitats, and by receiving wastewater their whole ecosystems can become adversely affected. Through the construction of artificial wetlands, natural and diverse wetland habitats are preserved and generators of wastewater are able to avoid the strict regulatory and environmental requirements associated the discharging effluent into natural wetlands. A carefully designed wetland may be able to treat wastewater to a much higher standard than natural wetlands.

Research on constructed wetlands was first initiated in Germany by Siedel and Kickuth in the 1960s and focused on sewage treatment (Cooper 1993: 203). Since the 1960s constructed wetlands have gradually captured the imagination of many scientists, engineers and authorities and are seen as a real alternative to conventional, centralised wastewater treatment systems. Unlike the conventional systems constructed wetlands require no chemicals and have low energy, labour and maintenance requirements. Along with sewage, constructed wetlands are now capable of providing treatment of municipal, industrial, and agricultural wastewaters, agricultural and urban runoff, landfill leachate, acid mine drainage waters, and domestic wastewater on-site.

3.3. Wetland Components

The five principal components of natural and constructed wetlands are (Hammer and Bastian 1989: 14):

1. water column;
2. plants;
3. substrates;
4. an aerobic and anaerobic microbial population; and
5. invertebrates and vertebrates.

3.3.1. *Water Column*

The water column is the medium in which wastewater pollutants are carried. These pollutants may be carried in a dissolved, solid or colloidal form. An important consideration in the design process is to ensure that a plug-flow occurs through the wetland, so that none of the wastewater is short-circuiting the treatment processes. A well designed wastewater flow will maximise pollutant contact with all of the treatment processes involved within the wetland, achieving the highest level of treatment possible.

3.3.2. *Plants*

Wetland plants are very robust and have developed many unique characteristics that enable them to survive in water saturated soils and tolerate wide fluctuations in nutrient levels. Wetland plants that are to be used for wastewater treatment must be able to provide an effective and efficient means of pollutant removal. Some categories and species of wetland plants are better suited for wastewater treatment than others. The three categories of wetland plants are:

1. floating plants- the photosynthetic parts of the plant are at, or above, the water surface and the roots extend down into the water column (e.g. water hyacinth);
2. submerged plants- these plants are rooted in the bottom sediments and their photosynthetic parts extend upward in the water column (e.g. pondweed); and
3. emergent plants- firmly rooted in the bottom sediments and the photosynthetic portions protrude above the water surface (e.g. reeds, bulrushes).

Submerged plants are not widely used in wastewater treatment because they become shaded by algal blooms, which can cover the surface of a treatment area due to the high nutrients found in wastewater. The shading by the algae prevents the growth of submerged plants and therefore the treatment process become less effective (Griggs and Koosterman 1988: 3). Floating plants, on the other hand, receive plenty of light as they are located on or above the water surface and are fast growers provided that they have sufficient room, adequate nutrients and an optimal temperature. Under ideal conditions floating plants are rapid growers and very effective at stripping wastewater of its pollutants. For example, the water hyacinth (*Eichhornia crassipes*) doubles every 6.2 days in sewage ponds in Florida while salvinia (*Salvinia molesta*) has been reported to double every 36 hours in nutrient rich conditions at Mt. Isa (Brett 1989: 18).

There are, however, three main problems associated with floating plants (Brett 1989: 18). Firstly, as prolific growers they can cause huge environmental and social problems if they find their way into other waterways. As a result many floating plants in Australia are declared noxious weeds. The second problem associated with floating plants is that their rapid growth creates large quantities of biomass, which must be harvested regularly in order to maintain the constructed wetlands treatment capabilities. Regular harvesting greatly increases operating and maintenance costs, but it may be possible to obtain some benefits from the resulting biomass as compost material or fuel. Lastly, most floaters are tropical plants and need certain climatic conditions to grow and are unsuitable for use in temperate and seasonal climates.

Emergent species are by far the most commonly used plant species in constructed wetlands throughout the world. They are the only species of wetland plant that can be planted in subsurface flow constructed wetlands (outlined below in 3.4 Types of Wetlands), which are the most widely utilised type of constructed wetland used for wastewater treatment. Emergent species also provide the greatest stabilisation of ground sediment, prevent erosion and clogging, and provide insulation in colder climates (Brix 1994: 76-7).

A major advantage of emergent species is their ability to transfer oxygen to their rhizosphere, creating an environment essential for sustaining aerobic mirco-organisms, which are essential in the wastewater treatment process. Water saturated soils are depleted of oxygen, unlike well drained terrestrial soils that have porous spaces allowing microorganisms and plant's root systems to gain their oxygen requirements. Pore spaces in water logged soils are filled with water rather than oxygen and therefore become anaerobic except for a few millimeters at the surface. Oxygen supply to the root zone for emergent species of wetland plants in water saturated soils is delivered internally from the plants' aerial organs (Brix 1994: 72).

The oxidisation process produces aerobic, anoxic and anaerobic zones in close proximity to one another within the substrate. The aerobic zone creates an environment in which micro-organisms flourish within the plants' rhizosphere. These micro-organisms play a vital role in the breakdown of pollutants found in wastewater by stimulating both aerobic composition of organic matter and the growth of nitrifying bacteria (Brix 1994: 73).

Oxygen and other atmospheric gases enter the aerial shoots via numerous stomata/lenticels on living leaf sheaths and to a smaller extent culm nodes. Internal transportation of oxygen may occur by 'passive molecular diffusion', where oxygen is dispersed by temperature and humidity gradients between the plant's aerial parts and its root system; or by 'convective flow', bulk flow of air through the internal gas spaces, lacunal system, of the plant (Brix 1994: 73). The internal lacunal system can occupy up to 70% of the total plant volume for some emergent species (Guntenspergen et al. 1989: 73). Oxygen transport into the root zone through lenticels and the lacunal system has been measured between 2.08 g $O_2/m^2/d$ (Brix and Schierup 1990; 56) and 5 to 12 g $O_2/m^2/d$ (Armstrong et al. 1990: 41) in *Phragmites australis* grown in gravel beds. For oxygen to enter the rhizosphere the oxygen levels transported must be greater than the oxygen required for root metabolism, these 'luxury' levels of oxygen are then released to the microbial populations surrounding the root system.

In oxygen studies of *Phragmites australis* Armstrong and Armstrong (1990: 532) found that in established constructed wetlands, where some old dead culms existed, there was greater transportation of oxygen than in newly established beds. They also found that in winter, in periods of dormancy or senescence, the transfer of oxygen through humidity-induced convection virtually ceases but the presence of broken culms "should considerably augment ventilation of the rhizome...", maintaining some oxygen supplies. It is essential to maintain oxygen supplies to the roots and rhizomes in the winter period in order to maintain population levels of aerobic microorganisms, resulting in a more consistent year round treatment process.

As the aerated rhizosphere plays a vital role in the treatment process, the ideal plants to be used are those which develop a large network of roots and rhizomes. The larger the area that the root system covers results in a greater contact area between the root zone and wastewater, increasing the efficiency of constructed wetland in treating wastewater. The effect of root penetration on operational performance in SSF wetlands is illustrated in Table 3.1. It is clear from the data in Table 3.1 that the removal efficiency for BOD and ammonia nitrogen (NH_3) is directly related to the depth of root penetration. Emergent species with greater root depth provide greater distribution of oxygen throughout the substrate, thereby allowing aerobic microorganisms to populate levels of the substrate that would otherwise be anaerobic. These results also clearly demonstrate the necesity of vegetation in SSF wetlands as the performance of the unvegetated beds for BOD and ammonia were relatively poor.

Table 3.1: Performance Comparison of Root Penetration on Water Quality in SSF wetlands at Santee, California.

Bed Condition[1]	Root Penetration (cm)	Effluent Quality, mg/L (Removal Efficiancy)		
		BOD	TSS	NH_3
Scirpus	76	5.3 (96%)	3.7 (94%)	1.5 (94%)
Phragmites	>60	22.3 (81%)	7.9 (86%)	5.4 (78%)
Typha	30	30.4 (74%)	5.5 (90%)	17.7 (29%)
No vegetation	0	36.4 (69%)	5.6 (90%)	22.1 (12%)

Note: 1- Q= 3.04 m^3/d; HRT= 6 days; bed dimensions, L= 18.5 m, W= 3.5 m, y= 0.76 m; primary wastewater effluent, BOD= 118 mg/L, TSS= 57 mg/L, NH_3= 25 mg/L.

Source: Reed et al. 1995: 229.

In summary emergent species perform five critical functions in constructed wetlands for wastewater treatment, most of which relate to their underground structure and growth (Verhoeven 1996: 4):

1. the root and rhizome mass form extensive aerobic attachment sites for microorganisms, which are essential in the treatment process;
2. the roots of some species produce "luxury" amounts of oxygen that leaks into the rhizosphere supporting further aerobic activity;
3. the roots and rhizomes modify the substrate texture, hydraulic conductivity and chemistry, and therefore sustain sediment structure and promote sedimentation and pollutant transformation process;
4. the plants are able to accumulate nutrients and toxins in their standing biomass, this uptake is minimal compared to the work done by the microorganisms and once the plant stand has matured subsequent uptake will be minimal unless harvesting is done; and
5. appropriately planted emergent macrophytes maintain a plug-flow of wastewater across the wetland providing a more consistent treatment performance.

Wetland plants are capable of reproduction through sexual and asexual means. Sexual reproduction occurs through the pollination of female flowers from male flowers and results in the formation of seeds. Many species produce seed that is viable for years but remains dormant until conditions are favourable for germination (Kadlec and Knight 1996: 141). Asexual reproduction occurs through vegetative growth when new shoots emerge from the roots or rhizomes.

3.3.2.1. Plant Establishment

A healthy stand of aquatic vegetation is an essential feature for a consistent wetland treatment performance. Kadlec and Knight (1996: 686) state that vegetation establishment should be done in spring to early summer, which is the start of the growing season. Plant establishment can be done in three ways:

1. Seedlings: young plants are established from fertile seeds collected in the field or from stock. Seedlings are easily transplanted into the wetland and provide for quicker establishment than for seed germination on-site. The survival rate for seedlings is greater (usually >80%) than for seed germination at the site.

2. Seeds: On-site germination of seeds at the wetland site has a lower cost than seedlings but the time period for plant establishment is a lot slower because seedlings can be grown under more favourable and controlled conditions in a hothouse. The survival rate is more variable because conditions cannot be controlled as well.

3. Field harvested plants: this involves the collection of plants or parts of the plants where they are already established. The best part of the plant to get are the roots and rhizomes as nutrients are already stored in these part of the plant, once the growing season starts many shoots can be produced from one cutting in a few weeks. Plants established from this manner grow faster and produce greater quantity of plants resulting in the earlier treatment of wastewater. The cuttings should be from the local area as the plants will be more adapted to the local conditions and climate.

3.3.3. Substrate

The substrate of a constructed wetland has three main functions. The first is to provide physical support for the vegetation in the wetland. Secondly, the substrate provides important attachment surfaces for the microbial populations. Lastly, it provides important removal processes of pollutants through the filtration of solids and by providing a considerable reactive surface area for complexing ions, anions and other compounds (Hammer and Bastian 1989: 15). The reactive surface area is important in the reduction of chemical and nutrient compounds such as phosphorous.

The most commonly used substrates in constructed wetlands are in the form of soil, sand or gravel. Plastic has also been trialled and tested as a substrate (Burgoon et al. 1989: 536). It was thought that the plastic could be designed to have a specific surface area two to four times that of gravel and would therefore provide larger colonies of microorganisms. Unfortunately, the plastic was not seen as a good substitute for gravel because plant establishment was not nearly as good and pollutant removal suffered as a result.

Soil is the favoured substrate for free water wetlands and gravel tends to be the most favoured substrate material for subsurface flow wetlands (both described in 3.4 Types of Wetlands) due to higher levels of hydraulic conductivity.

3.3.4. Aerobic and Anaerobic Microorganisms

The capacity of wetlands to treat wastewater by transforming pollutants into non-toxic forms is based on their high biological activity (Meney 1996: 3). Microbial populations in wetlands include the diverse flora of bacteria, algae, and fungi which are important for nutrient cycling and pollutant transformation. Due to the huge diversity of microorganisms, which are capable of functioning in a wide range of physical and chemical conditions, wetlands are a robust and consistent form of wastewater treatment. Microorganisms are able to convert pollutants in organic and inorganic forms to other substances, such as water, carbon dioxide and biomass, through a enzymatic or a biological chemical process (Portier and Palmer 1989: 90). The diversity and population size of microorganisms in a wetland are dependent upon the quantity of attachment sites and differing levels of oxygen available at these sites, i.e. whether aerobic, anoxic or anaerobic conditions exist (Meney 1996: 3). Under favourable conditions, which can be designed into constructed wetlands, most microbial species are capable of reproducing extremely rapidly, some doubling in population every couple of hours. These rapid population increases are vital in maintaining a stable treatment quality when the level of pollutants and inflow volumes are extremely variable. Microorganisms also help provide the robustness of constructed wetlands through their ability to produce strains quickly through genetic modification enabling them to survive hostile or toxic environments (Portier and Palmer 1989: 99). Constructed wetlands are capable of being utilised as mining leachate or tailings dams because certain metals are required by all microorganisms for normal cell functioning.

The overall productivity of wetlands to efficiently treate wastewater is dependent on two main categories of microorganisms (May et al. 1990: 34). The first is the heterotrophic bacteria which oxidise organic matter and release ammonia, and the second is the autotrophic nitrifying bacteria which oxidise ammonia to nitrite and nitrate. It is also important to note that many naturally occurring microbial groups are predatory and will feed on pathogenic organisms (Hammer and Bastian 1989: 14).

3.3.5. Invertebrates and vertebrates

Invertebrates and vertebrates are higher links in the food chain and provide integral parts of any ecosystem. Cetain species of fish and crustaceans are filter feeders and can be used in wastewater treatment to reduce solids found in the water column, further decreasing levels of BOD and suspended solids (Reed et al. 1995: 167). Many on-site designs avoid this aspect of constructed wetlands as a larger treatment area is required to support them.

3.4. Types of Wetlands

Constructed wetlands can be classified into two main categories; free water surface systems (FWS) and sub-surface flow wetlands systems (SSF). The two systems operate in different ways, each having different flow patterns, different pollutant removal mechanisms, and differing plant species and planting regimes (Mitchell 1996: 2). These two types of systems are outlined below with their distinct advantages and disadvantages.

3.4.1. Free Water Surface Wetlands

In FWS the wastewater flows between the wetland floor and the atmosphere. Wastewater is required to flow through a plant matrix which can consist of floating, emergent or submerged species. The great majority of microbial activity occurs in the peat accumulation zone at the floor/water interface, and in the microbial films which coat the stems and roots of the plant sections in the water column (Mitchell 1996: 4). Filtration and sedimentation in FWS systems occur with the natural settling of solids and by the plant matrix acting as a filter in the water column. BOD removal in FWS is achieved by the filtration/sedimentation process followed by microbial activity.

Aeration of the wastewater in FWS occurs through gas exchange between the atmosphere and water surface, reed movement mixing oxygen into the water, and oxygen release through photosynthesis or from the root zone if emergent species are used.

It is possible to introduce crustaceans, fish and other vertebraes at the concluding stages of FWS wetland to produce a higher quality effluent.

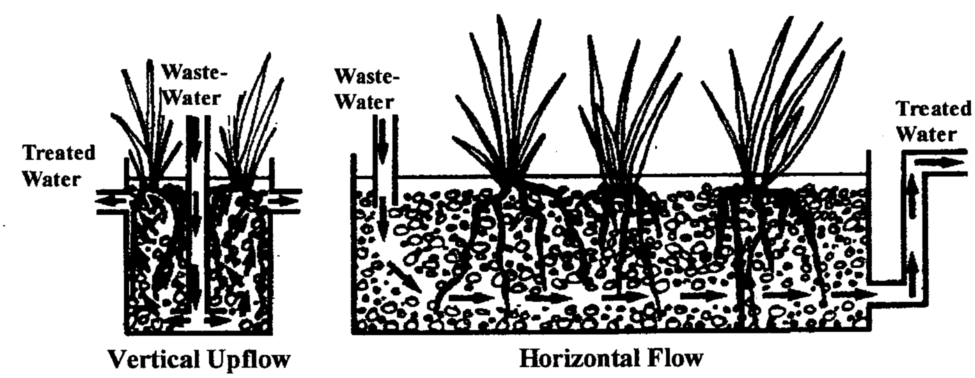
3.4.2. *Subsurface Flow Wetlands*

SSF systems can only use emergent plant species because the wastewater flows below the substrate in which the plants are rooted. One of the principal design aims of SSF wetlands is to maximise the wastewater root-zone contact, thereby providing greater treatment efficiencies as the root-zone accommodates the main population of microorganisms. SSF systems provide a greater potential for microbial activity as there is a greater number of attachment sites in the form of substrate material and the root and rhizome system of the plants. Essentially the wastewater is treated in three ways, filtered through the substrate, uptake from the plants, and by the microorganisms present in the root zone and substrate.

The two main categories of SSF wetlands are the horizontal and the vertical flow system shown in Figure 3.1. In horizontal systems the wastewater flows horizontally through the substrate until it reaches the outflow pipe at the other end. A water depth of 30-50cm can be maintained in the bed/trench (Grant et al. 1996: 50). The outlet pipe on some horizontal flow systems can be raised or lowered, affecting the water level in the substrate. This is an important feature as the water level can be lowered to encourage the root and rhizome systems to grow deeper into the substrate. However, it must be noted that root densities of aquatic plants decrease with depth, so unfortunately a volume of wastewater travels along the bottom of the trench, bypassing the rootzone (Brett 1989: 21). Vertical flow systems overcome this problem by further maximising the wastewater rootzone contact as the effluent must pass vertically through the rootzone on its journey to the outflow pipe (Breen 1989: 168).

Vertical flow wetlands can be divided into upflow or down-flow systems. Upflow systems, as in Figure 3.1, require the wastewater to enter through a central pipe into the base of the reed bed. The effluent is forced upward through the root-zone until the outflow pipe is reached just below the surface of the substrate. The vertical upflow system was developed and patented through CSIRO's Division of Water Resources by Mr Peter Breen, Dr David Mitchell and Mr Alan Chick in the 1980s. In downflow systems the wastewater flows vertically downward through the substrate. There are two variations on the downflow system. The first is free draining where each reed bed resembles a percolating filter planted with aquatic plants (Grant et al. 1996: 48). The wastewater is fed directly onto the substrate surface, filtering vertically downwards through the substrate. The wastewater passes through the rootzone to the bottom of the trench where it is collected by under drains and directed to an elimination trench or storage pond. The second is not free draining but maintains a constant effluent depth within the reed bed (Rogers et al. 1990: 588). Effluent outlets are located at the bottom of the bed and rise outside of the planted reed bed to the desired water level.

Figure 3.1: Vertical and Horizontal Flow SSF Systems



3.4.3. Advantages and Disadvantages

Due to the distinctive design of FWS and SSF each has their own advantages and disadvantages. Two main advantages of SSF over FWS systems are that there is no free or exposed water surface and that there is a greater surface area for microbial growth (Wood 1995: 24-5; Mitchell 1995: 900). Constructed wetlands that have less wastewater exposed decrease the risk of public exposure, thereby decreasing the heath risks associated with wastewaters. Also, by maintaining the wastewater within the substrate there are no or minimal odours and less scope for pests (e.g. mosquitos) to breed or be attracted to. There is greater microbial activity in SSF wetlands due to the greater surface area from the substrate and the root/rhizome system, which promotes a better treatment efficiency per unit area of land. Therefore, SSF constructed wetlands require less land area to treat the same volume of wastewater.

Algae and floating macrophytes are capable of removing greater quantities of nutrients and being significantly more photosynthetically active than emergent plant species used in SSF systems (Wood 1995: 24). FWS constructed wetlands permit algae and floating macrophyte communities to develop in free water whereas SSF cannot support floating macrophytes and algae will only appear on the substrate surface if sufficient sunlight is available. Therefore, in FWS, algae and floating macrophytes are more capable of generating greater levels of oxygen and indirectly removing CO_2 from the water, resulting in an increase in water pH towards alkaline, which assists phosphate precipitation and ammonia volatilisation (Wood 1995: 24). However, because algae and floating macrophytes are more active in nutrient uptake and photosynthesis they generate greater biomass, which has two disadvantages. If left unchecked they can adversely affect the water flow and the hydraulic retention time. Secondly, if plant uptake is the main form of nutrient removal harvesting is necessary because once a plant is fully grown nutrient uptake is minimal. Therefore, FWS systems usually require harvesting, dramatically increasing the operating and maintenance costs. SSF systems require no or minimal harvesting as these systems essentially rely on the microorganisms to take charge of the purification processes

Nitrogen removal can be more efficient in FWS systems as the sediments and plant litter that forms on the soil surface of FWS systems provide a supplemental carbon source for denitrification (Wood 1995: 25). In SSF systems nitrogen removal is limited by the ability of oxygen to get to the subsurface water flow through the root/rhizome network.

A poor design or an inadequate substrate with low hydraulic conductivity has a tendency to clog SSF constructed wetlands. FWS systems do not have clogging problems as the wastewater does not flow through a substrate, but flows freely through a plant matrix. Many studies conducted on SSF wetlands with soil substrates comment on the poor hydraulic conductivity of soil for a substrate medium (Cooper 1993: 207; Davies and Hart 1990: 518; Schierup et al. 1990: 503). Poor conductivity or clogging of a substrate can cause flooding and surface flow, essentially creating a FWS, and results in a short-circuiting of the treatment process and increases maintenance costs. This problem has been overcome through the use of gravel in SSF systems, which has a greater hydraulic conductivity and a lower potential for clogging.

SSF constructed wetlands are more costly to install than FWS systems. This additional expense is due to the high costs associated with the purchase of a substrate material, usually gravel. For greater details on costs relating to constructed wetlands refer to 3.9 Costs.

Further advantages and disadvantages that directly relate to on-site constructed wetlands in a temperate climate are discussed in the design setion below.

3.5. Constructed Wetland Design

Kadlec and Knight (1996: 14) estimate that there are over 1,000 constructed wetlands worldwide with highly individualistic and variable designs. This variability in design is due to the wide range of specific purposes that constructed wetlands are used for and the lack of cohesion around the world regarding information on design and construction of wetlands. With this information scattered throughout many locations around the globe it is difficult for scientists, engineers, and public officials to access it. Consequently, research and design procedures are being repeated unnecessarily, resulting in wasted time and money or systems failures, which eventually discourages owners and authorities from utilising this technology.

The variable nature of wetland design and construction is further compounded by the way certain professions view them in isolation with regard to a specific discipline (i.e. engineers vs hydrologists vs ecologists vs botanists etc.). Only recently has there been any attempt to gather the vast amount of information and data regarding design, construction, their treatment capabilities, costs etc. Three databases have been set up to establish exactly what the current position is with respect to constructed wetlands technology and the scope of treatment capabilities, and thus to avoid the unnecessary duplication of research and work. One database focuses on the North American experiences, another on European systems and the last and smallest concentrates on constructed wetlands in Great Britain (Kadlec and Knight 1996: 717, 736).

The collation of all the data described above has led to a large array of design formulae and models for constructed wetlands, most relating to their physical aspects. However, due to the complexity of wetland ecosystems there are no formulae or models available for the biological components of constructed wetlands. There is growing support for understanding the actual biological processes at work within constructed wetlands (Meney 1996: 5; Mitchell 1996: 5; Kadlec and Knight 1996: 109). This knowledge should lead to a more holistic approach in the designing of a constructed wetland whereby engineering, hydrological and biological design components are jointly addressed to achieve the treatment goals more effectively and efficiently.

3.5.1. General Design Considerations

As there have been numerous design manuals released with regard to constructed wetlands (Kadlec and Knight 1996; Reed et al. 1995; USEPA 1988, 1993) this thesis will not delve into any of the models or formulae. However, it is important to outline some principles that should be incorporated into the design process. Meney (1996: 5-6) outlines three broad principles of biological design that can apply to constructed wetlands of any scale, from small on-site systems to large-scale industrial or urban wetlands. Meney summarises these as:

- "Design must be focused on establishing a sustainable biological system. This requires understanding species-species interactions and interactions with substrate, hydrological regime, pollutant types and concentrations, and the overall physical and chemical balances;"
- "The biosystem must be structurally and functionally optimal for the type and character of wastewater being treated. Plant architecture, pollutant tolerances, oxygen demands, growth rates and seasonal dormancy characteristics vary between species and growth forms;" and
- "The efficiency of the biosystem is strongly dictated by the appropriateness of the physical design. For this reason, the biological design should be considered in the preliminary design phase."

It is also important to be aware of other factors which must be considered in the designing stage of constructed wetlands. The following design factors are drawn from a variety of sources (Mitchell 1996, Kadlec and Knight 1996, Meney 1996, Marshall 1995, Reed et al. 1995, Griggs and Koosterman 1988, and own experiences[1]) in order to gain an understanding of the types of decisions required at the design stage.

Wastewater Characteristics and Establishment of Treatment Objectives

For a constructed wetland to operate successfully it is crucial that the influent characteristics, loading rates and treatment objectives are known. A major consideration in establishing the treatment objectives is the legislative requirements that are applicable

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[1] The author has set-up an on-site constructed wetland as a case study for this thesis and is including any design aspects that are considered to be of value.

to the situation. Different pollutants require different treatment processes, so identification of contaminants and their concentrations/levels must be known in order to be incorporated into the design. In some instances it may be necessary for the wastewater to require pretreatment before entering a constructed wetlands (e.g. blackwater pretreated in a septic tank to settle the solids).

Site Selection

Factors to be considered in site selection are climate (rainfall, temperature ranges, seasons, wind, solar radiation), geography, soil, groundwater etc. In many instances on-site treatment offers little choice in site selection and any weaknesses or disadvantages in the site must be known so that they can be accounted for in the design stage.

Substrate and Hydraulics

A substrate should be chosen that is suited to the type of plant species to be used in the wetland. Some species are not capable of growing in some soils or find that root and rhizome penetration is difficult in gravel substrate. In horizontal flow SSF systems it is essential that the substrate depth does not exceed the maximum rooting depth of the selected plant species, otherwise the wastewater will flow under the root zone and exit the system relatively untreated. Certain substrates induce flooding and surface flow in SSF systems, which is undesirable as the treatment is effectively being short-circuited. In FWS systems the water flow should be gentle so the plants are not bullied or stressed and the water level should not be so high as to cause death or stress to emergent and submerged species.

Hydraulic Retention Time (HRT)

It is necessary for wastewater to stay within a constructed wetland for a certain period of time to ensure that adequate treatment occurs. If wastewater passes too quickly through a wetland then all the treatment processes at work will have been unable to treat the wastewater to desired levels and the effluent may still pose potential health and environmental risks. HRT is derived by dividing the volume of wastewater that the constructed wetlands can hold by the hydraulic loading per day (ie. influent per day).

Inlet and Outlet Distribution

Good distribution at both the inlet and outlet generally ensures good system performance and unnecessary short-circuiting. Perforated pipes and coarse substrate material, such as large gravel, will promotes good distribution and decreases the chances of clogging at the inlet on SSF wetlands.

Bed Configuration

It is best if the beds are compartmentalised into a series of individual or parallel beds and are capable of being drained. This allows flexibility in operation, maintenance, and long-term management, and short-circuiting is minimised.

Plant Selection and Establishment

Ideally local species should be selected as they are most likely adapted to the local environment and if they spread into local waterways are less likely to become a pest or nuisance plant. Species should be selected that are most appropriate to the wastewater characteristics that it will be treating. The plants must be planted in sufficient numbers to ensure adequate treatment will occur once fully grown. An outlet control that allows the user to dictate the water level is important in FWS to allow faster plant establishment and in SSF systems to encourage root depth.

3.5.2. On-site Design Considerations

To date, the majority of constructed wetlands used for wastewater treatment focus on medium to large-scale systems. Constructed wetlands have the potential to be an alternative to traditional on-site wastewater treatment and disposal methods, such as septic tanks and absorption trenches. But for constructed wetlands to become a recognized form of on-site treatment they must be accepted by both users and authorities as safe, reliable and cost effective. Below are some considerations which should be considered when designing constructed wetlands for small-scale on-site use.

Users of on-site treatment systems do not want ugly structures visually polluting their property. Therefore, aesthetics need to be considered if constructed wetlands are to become a more accepted form of treatment in on-site domestic situations. Constructed wetlands should be intergrated into the surrounding landscape/garden to become aesthetically pleasing. However, Mitchell (1995: 906) stresses that while aesthetics are important in domestic wetlands they must not be designed in at the expense of operational requirements. By compartmentalising a wetland system's beds there is greater flexibility in enhancing the aesthetics of the situation while maintaining the required operational levels (Mitchell 1996: 7).

For greater acceptance in domestic situations it is vital to design a system that is failsafe, flexible and requires minimal maintenance. To accomplish this, initial preparation and design is vital so that an adequate buffer capacity is inherent, health risks from exposure to wastewater are non-existent or minimal, and any inspection outlets and filters are easily accessible and changeable.

It is important to design a wetland for household wastewater treatment so that it does not drain freely but retains a water column within the substrate. By retaining a water column the wetland plants will be able to survive periods with no wastewater inflow, such as periods when the inhabitants are on holiday (Mitchell 1995: 901). Due to this, free draining downflow SSF systems should not be used in a domestic situation.

To ensure vigorous and healthy reed growth the wastewater must have adequate nutrients and trace elements. Marshall (1995: 89) and Mitchell (1995: 903) both recommend mixing blackwater with the greywater to guarantee strong reed growth. Both Marshall's and Mitchell's papers state that while reed growth was occurring in constructed wetlands receiving greywater only it was not as vigorous as in wetlands receiving a mixed effluent. Blackwater can be sourced from a septic tank or from excess liquid from a composting toilet.

A big advantage of constructed wetlands is that they can be installed into any soil type provided an appropriate liner is used (Mitchell 1995: 900). This means that constructed wetlands can be installed into locations where other traditional wastewater treatment and/or disposal methods are failing, such as absorption trenches in clay soils or where the groundwater is close to the surface.

SSF wetlands are the best systems available for domestic on-site situations because the wastewater flows below the substrate minimising health risks and insect breeding sites. The additional cost of SSF systems over FWS due to expensive substrate is worthwhile for these two factors alone. However, the price differential of SSF systems as compared to FWS system in small-scale on-site situations is expected to be minimal. One other advantage in using SSF systems for on-site wastewater treatment is that they require less land requirements than FWS systems due to their better treatment efficiency.

3.5.3. Cold Climate Design

Considerations must be made to constructed wetlands that are to operate in temperate climates with distinct seasons. Winter periods with their associated colder temperatures affect microbial activity, wetland plants (i.e. no growth, dormancy or senescence), flow of water (i.e. ice) and chemical precipitation. All of these variables are capable of decreasing a wetlands ability to treat wastewater to adequate or required levels and consequentially, have the potential to deter many authorities from using constructed wetlands in temperate climates.

Emergent species are the best plants to be used in temperate climates because they are the least affected of all wetland plants in cold conditions and provide some insulation, providing protection against frosts and the freezing of water. It is important to note that most reeds that are used in temperate climates will experience a period of dormancy or senescence over the winter period. Therefore, consideration must be given to the relative fate of the plant biomass during these periods because there is a potential for the release of assimilated nutrients into the water column. Wood (1995: 24) states that this period of die-off in the plants is more of a disadvantage to FWS systems than to SSF systems. This is a greater risk in FWS systems as nutrient release is directly into the water column where as the nutrients are more likely to remain immobilised in the substrate of SSF constructed wetlands.

FWS systems are also very dependent upon the plant matrix for filtration, microbial attachment and pollutant degradation and transformation. With plant die-off or partial shedding of leaf matter during senescence or dormancy there is less filtration, fewer microbial attachment sites, and the flow paths and hydraulic retention time can be adversely affected (Wood 1995: 24). Wood stresses that these periods can be particularly detrimental in performance of FWS systems as plant senescence and dormancy also “coincides with reduced temperatures and associated biological reaction kinetics”, the combined effect can greatly diminish the treatment capability.

SSF systems not only have a greater amount of microbial activity but also have more stable attachment sites than FWS system throughout the year (Wood 1995: 24). This stability of attachment sites is due to insulation from the surrounding substrate and the large quantity of attachment sites available to microorganisms. The insulating factor diminishes the impact of colder temperatures on the metabolic activity of the microorganisms, providing a more balanced performance of pollutant removal throughout the year. Without insulation FWS systems are prone to freezing or ice cover in very cold areas, resulting in a smaller treatment area, less aeration of the wastewater and minimal microbial activity. Jenssen et al. (1997: 248), who have experience with constructed wetlands in Norway, also stress the importance of insulation in winter to prevent hydraulic failure due to freezing. They also recommend a deeper system in order to gain a sufficient hydraulic retention time to compensate for the decline in microbial activity.

There has been some success in FWS constructed wetlands operating in the cold climate of Platteville, Colorado, USA, where summer algal problems were causing high levels of BOD_5, TSS and pH (Thorson et al. 1994: 26). The reeds were able to control the algal growth by preventing sunlight from penetrating the algae. During winter, when the wetland plants die-off, algal growth is not such a problem because the climatic conditions are not optimal for its growth. In this situation, where the main cause of effluent problems is algae, the FWS constructed wetlands provided an ideal solution. However, the Colorado Department of Public Health and Environment (CDPHE) is concerned about the cold weather performance of FWS constructed wetlands at future Colorado sites where ammonia or nutrients are required to be removed to certain levels. Their main concerns are the freezing of surface water, decreases in microbial metabolism and die-off of wetland plants during winter, all adversely affecting most of the pollutant removal mechanisms for part of the year.

These are genuine, realistic and practical concerns expressed by the CDPHE and are a good indication of the limitations facing FWS constructed wetlands operating in colder climates. For these reasons SSF constructed wetlands are the favoured systems used in these climates.

3.6. Purification Processes

A prime objective of constructed wetlands is to eliminate or lower levels of contaminants found in wastewater. The main contaminants are; Biochemical Oxygen Demand (BOD), nitrogen, phosphorous, faecal coliform and suspended solids (SS). These contaminants were described in Chapter 2, but to provide an understanding of the mechanics at work within a constructed wetland the treatment processes required to break down these contaminants are outlined below.

3.6.1. Biochemical Oxygen Demand (BOD)

To recap, BOD is a measure of oxygen consumption by microorganisms in the aerobic oxidation of organic matter (i.e. carbon). There are many carbon processes occurring in wetlands, some raising BOD levels and others which reduce it. In wetlands, BOD removal associated with settleable solids in the wastewater usually occurs within the first part of the system through sedimentation and filtration (Griggs and Koosterman 1988: 8; Reed et al. 1995: 187). The remaining colloidal and soluble BOD is primarily removed as a result of the metabolic activity of micro-organisms.

3.6.2. *Suspended Solids*

Suspended solids are particulate matter that are found within the water column. Suspended solids increase turbidity and BOD levels and are capable of retaining other pollutants, such as chemicals and metals. Suspended solids are removed from wastewater in a similar way to BOD, by sedimentation and filtration of the solids within the substrate media. Any non-settling colloidal solids are removed by decay, microorganisms, plant uptake or through absorption to other solids (Griggs and Koosterman 1988: 8).

3.6.3. *Pathogens*

As discussed in Chapter 2, pathogens have the potential to be present in domestic wastewater situations. It is therefore imperative that any wastewater treatment process eliminates this potential risk. Conventional treatment decreases pathogens through chlorination, ozonisation and ultra-violet disinfection. Chlorination is the most widely used treatment due to its low cost and effectiveness. However, residual free chlorine harms a variety of aquatic organisms and causes chronic and acute toxicity to microorganisms and fish, and has the potential to become carcinogenic when it comes into contact with organic compounds in wastewater or naturally occurring compounds found in receiving waters (Kadlec and Knight 1996: 533). Wetlands are a hostile environment for pathogens and provide a climate which is capable of eliminating or reducing pathogens through natural die-off, temperature, ultra-violet light, unfavourable water chemistry, biological deactivation, predation, filtration, and sedimentation (Kadlec and Knight 1996: 535; Griggs and Koosterman 1988: 11).

3.6.4. *Nitrogen Cycle*

Nitrogen compounds are one of the principal pollutants in wastewater that must be addressed due to their role in eutrophication, their effect on the oxygen content of receiving waters and their toxicity to invertebrate and vertebrate species. It is important to note that nitrogen compounds can play a beneficial role in augmenting plant growth which in turn stimulates the production of wildlife. Nitrogen compounds include a variety of inorganic and organic forms (Kadlec and Knight 1996: 373). The most important inorganic forms of nitrogen are ammonia (NH_4^+, NH_3), nitrite (NO_2^-), nitrate (NO_3^-), nitrous oxide (N_2O), and nitrogen gas (N_2). The organic forms of nitrogen include urea, amino acids and purines.

Nitrogen is removed from wastewater during wetland treatment through a number of mechanisms. The main ones are (Griggs and Koosterman 1988: p.8):

1. Bacterial nitrification/denitrification,
2. Volatilisation of ammonia, and
3. Uptake and subsequent harvesting of plants.

Bacterial nitrification/denitrification provides the greatest overall potential for nitrogen removal and is the main nitrogen removal process at work in constructed wetlands. This is achieved through natural biological transformation by bacteria living within the wetland system. Organic nitrogen is biologically transformed to ammonia through ammonification and is the first step in the mineralisation of organic nitrogen (Kadlec and Knight 1996: 380). Ammonia nitrogen is then converted to nitrite and nitrate through nitrification. Aerobic conditions are necessary for nitrification to occur, although the process will still occur at low levels of dissolved oxygen. Denitrification occurs under anaerobic conditions whereby nitrite and nitrate nitrogen are reduced to nitrogen gas, nitrous oxide, or nitric oxide. For the denitrifying process to occur there must be an adequate supply of carbon.

Volatilisation of ammonia is a process where ionised ammonia in solution (NH_4^+) is converted to unionised ammonia gas (NH_3). Volatilisation is a pH dependant process, at a pH below 8 volatilisation is insignificant but becomes more significant as pH increase to 9 and above (Griggs and Koosterman 1988: 10). Wetland pH levels must therefore be quite high for any significant removal of nitrogen in this manner.

For significant nitrogen removal through plant uptake to occur it is necessary to incorporate a harvesting regime into the wetlands operations. Once wetland plants reach maturity nutrient uptake diminishes substantially. Relying on plant uptake in temperate climates can also be a problem as many of the plants utilised in wetlands systems go dormant for the colder period of the year.

3.6.5. *Phosphorous Cycle*

Phosphorous is a difficult pollutant to remove in any water treatment technology. In constructed wetlands the primary phosphorous removal mechanisms are biological and chemical storage with some plant uptake (Griggs and Koosterman 1988: 10). Adsorption of phosphorous into substrate sediments accounts for the majority of phosphorous removal, but has a limited storage capacity. The phosphorous fixation or storage capacity of a wetland depends on phosphorous loading, sediment composition and the sediment-wastewater contact area. The rate of adsorption is directly dependant upon the chemical and physical properties of the substrate. Iron and aluminium rich materials, limestone media, and specially prepared clays are found to have high phosphorous storage capacities as they are able to immobilise and store the phosphorous in their sediments (Kadlec and Knight 1996: 451). Hence, phosphorous removal is much more successful in SSF systems than in FWS systems due to wastewater passing through the substrate bed, allowing greater potential for phosphorous absorption. Substrates, however, only have a limited storage capacity and once the substrate is saturated with phosphorous the removal rate will slow dramatically and the prime removal mechanism will be through plant uptake. If phosphorous removal is a prime treatment objectives of a wetland it is necessary to replace the substrate once saturation occurs.

Phosphorous removal through the harvesting of biomass has not yet proved feasible for large-scale constructed wetlands as it is labour intensive and costly, which Kadlec and Knight (1996: 445) believe is "antithetical to the passive character of wetlands technology". But harvesting of plants for small-scale on-site constructed wetlands may be feasible as the planted area is only much smaller and will take only a fraction of the time to harvest. Disposal of the resulting biomass would not be a problem as it could be composted and added to the owner's garden. There has been some success with the harvesting floating aquatic plants, such as water hyacinths, which are easier to harvest, have rapid growth and account for levels of phosphorous removal of around 20% (Fisher and Reddy 1987 quoted in Kadlec and Knight 1996: 445) as opposed to 2.5% removal for emergent species (Herskowitz 1986 in Kadlec and Knight 1996: 445). Floating aquatic plants can only be grown in tropical climates and would be of no use in temperate climates as they would struggle to survive.

3.7. Results of Constructed Wetlands in Operation

Constructed wetlands have been in operation internationally for over 20 years now. As stated in section 3.5 three databases have been set-up from three different regions of the globe. These databases have allowed an analysis of many systems enabling a greater understanding of constructed wetlands and their performance levels, design guidelines, operational and maintenance requirements and other characteristics. This aim of this section is to provide an insight into pollution removal rates and effluent quality of medium to large-scale constructed wetlands, small-scale on-site systems, and systems operating in cold climates.

3.7.1. Medium to Large-Scale Wetlands

A performance summary of constructed wetlands operating in three regions, North America, Denmark and Great Britain, are shown in Table 3.2 and Table 3.3. It must be noted that all wetlands represented in these regional summaries are medium to large-scale systems, performance values of small-scale on-site constructed wetlands are discussed below.

The Danish systems summarised in Table 3.2 have an average area of about 2037 m^2 and are able to provide an effective level of secondary and advanced treatment (Kadlec and Knight 1996: 735). They reduce average BOD_5 from 128 to 18 mg/L, ammonia nitrogen from 21 to 14.1 mg/L, nitrate + nitrite nitrogen from 4.1 to 2.0 mg/L, TSS from 163 to 27 mg/L, total nitrogen from 36.7 to 21 mg/L, and total phosphorous from 9.1 to 5.8 mg/L.

Denmark, being situated north of 55^0N and with winter temperatures averaging 0°C, can be classed as a temperate country (World Book Encyclopedia 1981). In a summary paper, the authors of the Danish database found that there was minimal seasonal variability for constructed wetlands operating within Denmark and the above results suggest that wetlands are capable of performing adequately in cooler and more seasonal climates (Schierup et al. 1990: 499).

The results from the British wetlands, Table 3.2, also suggest that climate is not a problem with removal efficiency of wastewater pollutants. The five British systems summarised here have an average area of 719.2 m^3 and produce an effluent with average BOD_5 and TSS values of 2.8 mg/L and 5.1 mg/L respectively. These effluent values are better than the Danish wetlands because the influent received by the British wetlands receives higher levels of pretreatment (Kadlec and Knight 1996: 737). The pretreatment consists of secondary treatment with trickling filters or rotating biological contactors and removes BOD_5 and TSS very effectively, hence the large difference of inflow values for these parameters (average inflow values of 15.35 mg/L and 32.57 mg/L for BOD_5 and TSS respectively for British wetlands and 128 mg/L and 163 mg/L for the Danish systems). Greater levels of pretreatment mean that a wetland can receive greater rates of hydraulic loading.

Nitrification and nutrient removal rates have not been as great in the British systems as the Danish; the average removal efficiency for nitrate + nitrite nitrogen, total nitrogen and total phosphorous in the Brithish systems is 36%, 30.6% and 20.9% respectively while the Danish systems have removal rates of 51%, 42.7% and 35.6% respectively.

An operational performance summary of SSF and FWS constructed wetlands that are listed on the North American datadase are shown in Table 3.3. Also shown in this table are typical effluent characteristics for SSF and FWS wetlands as set out by the USEPA. It is interesting to note that in North America there has been no favoured design philosophy, with SSF, FWS and hybrid systems developed, while the SSF design has dominated constructed wetland technology in Europe and Great Britain (Kadlec and Knight 1996: 734).

Table 3.3 shows that both SSF and FWS designs are achieving better quality effluent than the maximum typical performance levels stated by the USEPA. The removal efficiency of SSF systems is better than FWS wetlands on five of eight listed parameters. The three parameters that FWS wetlands have a better removal rate over SSF systems are; BOD_5 (74% to 69%), ammonia (54% to 25%) and total phosphorous (57% to 32%). The differential in removal rates for ammonia and total phosphorous is relatively large, 29 percentage points for ammonia and 25 percentage points for total phosphorous. The largest difference of SSF systems outperforming FWS systems is 14 percentage points which is for organic nitrogen; the rest are all under 10 percentage points. This may suggest that the differing designs have an impact upon the effectiveness of a wetland's ability to treat certain pollutants.

It is important to note the figures in Table 3.2 and Table 3.3 are averages only and that constructed wetlands can experience large variability between one another. Every wetland should be viewed as an ecosystem and any changes or processes that occur within that ecosystem can produce variations within or between wetlands (Kadlec and

Knight 1996: 611). For example, seasons, algal blooms, seasonal growth rates, insect or disease attack, loading rates, influent quality and design may all affect the final effluent quality of a constructed wetland.

Table 3.2: Operational Performance Summary of Danish and British Wetlands

		Concentration mg/L					
Parameter[1]	Ave and Range	Danish Constructed Wetlands[2]			British Constructed Wetlands[3]		
		In	Out	Eff.%	In	Out	Eff.%
BOD_5	Ave.	128	18	86	15.35	2.84	81
	Min.	3	2	-67	12	2	73
	Max.	661	149	99	27	4	89
TSS	Ave.	163	27	83	32.57	5.18	83
	Min.	4	2	-157	20	4	77
	Max.	1960	750	99.5	61	8	87.6
NH_4-N	Ave.	21	14.1	33	8.71	4.71	47
	Min.	0.1	0.1	-600	6.1	1.8	20.3
	Max.	79.1	46.6	50	12.6	8.6	80.6
$NO_2 + NO_3$-N	Ave.	4.1	2.0	51	4.78	3.04	36
	Min.	0.0	0.0	-1500	1.4	0.5	-28.1
	Max.	50.3	8.4	99.8	8.3	7.0	90.2
TN	Ave.	36.7	21.0	42.7	20.96	15.46	30.61
	Min.	4.1	4.7	-104.9	6.0	0.2	6.7
	Max.	142.3	69.4	85.6	33.0	25.7	96.7
TP	Ave.	9.1	5.8	35.6	9.44	7.78	20.87
	Min.	0.5	0.2	-111.4	6.6	3.9	3.0
	Max.	35.0	14.6	97.9	13.2	12.8	40.9

Notes: 1 – BOD_5= 5 day biochemical oxygen demand, TSS= total suspended solids, NH_4-N= ammonia N, NO_2+NO_3-N= nitrite + nitrate, TN= total nitrogen, TP= total phosphorous.

2 – Data from 71 constructed wetland systems in Denmark. Source: Kadlec and Knight 1996: 734-5.

3 – Data from 5 constructed wetlands systems in Great Britain. Source: Kadlec and Knight 1996: 734-5.

Table 3.3: Operational Performance Summary of North American Database and USEPA Expected Effluent Levels

		Concentration (mg/L)				
Parameter[1]	Type[2]	Nth Americcan Database[3]				Typical Perf. (USEPA)[4]
		In	Out	Eff.%		Out
BOD_5	FWS	30.3	8	74		<20
	SSF	27.5	8.6	69		<25
	All	29.8	8.1	73		-
TSS	FWS	45.6	13.5	70		<20
	SSF	48.2	10.3	79		<15
	All	46	13.0	72		-
NH_4-N	FWS	4.88	2.23	54		
	SSF	5.98	4.51	25		
	All	4.97	2.41	52		
$NO_2 + NO_3$-N	FWS	5.56	2.15	61		
	SSF	4.40	1.35	69		
	All	5.49	2.10	62		
ORG-N	FWS	3.45	1.85	46		
	SSF	10.11	4.03	60		
	All	4.01	2.03	49		
TKN	FWS	7.60	4.31	43		
	SSF	14.21	7.16	50		
	All	8.11	4.53	44		
TN	FWS	9.03	4.27	53		<15
	SSF	18.92	8.41	56		<12
	All	9.67	4.53	53		-
Ortho-P	FWS	1.75	1.11	37		
	SSF	nd	nd	nd		
	All	1.75	1.11	37		
TP	FWS	3.78	1.62	57		<6
	SSF	4.41	2.97	32		<1-4
	All	3.80	1.68	56		-

Notes: 1- BOD_5= 5 day biochemical oxygen demand, TSS= total suspended solids, NH_4-N= ammonia N, NO_2+NO_3-N= nitrite + nitrate, ORG-N=organic N, TKN= total Kjeldahl N, TN= total nitrogen, Ortho-P= ortho phosphorous, TP= total phosphorous, nd= no data.

2–FWS = free water suface flow, SSF = subsurface flow, All = FWS+SSF+hybrid systems.

3 – Source: Kadlec and Knight 1996: 731

4 – Source: U.S.EPA in Etnier and Guterstam 1997: 81

While SSF wtlands are usually the preferred design in cooler climate, FWS systems have had some luck in certain circumstances. A new wastewater treatment plant was required for Platteville, Colorado, USA, as the town's initial wastewater treatment plant, two aerated lagoons and quiescent lagoon, was discharging effluent that was over twice the town's legislated discharge limit of 75 mg/L for TSS and 30 mg/L for BOD, and exceeding the required pH level of 9 (Thorson et al. 1994: 26). The main cause for these excessive levels was ideal summer conditions for algal growth resulting in high readings of TSS, BOD and pH. A FWS constructed wetlands, planted with cattails, replaced the old system and results throughout the year showed good TSS and BOD removal, producing an effluent of 18 mg/L and 19 mg/L respectively. In summer, the cattails solved the algal problem by preventing sunlight from penetrating the algae, controlling its growth. During winter, when the wetland plants die off, algal growth is not such a problem because the climatic conditions are not optimal for its growth. In this situation, where the main cause of effluent problems is algae, the FWS constructed wetlands provided an ideal solution.

3.7.2. On-site Constructed wetland

Studies with detailed results of small-scale on-site constructed wetlands treating domestic wastewater are very limited. The most detailed study that the author could find was an Honours thesis conducted by Glenn Marshall (1995) at Southern Cross University, Lismore, NSW. Summary papers by Mitchell (1995) and Perfler and Haberl (1993) provide a further, but less detailed, insight into the use and efficiencies of constructed wetlands to treat household wastewater. All are reviewed below.

Marshall's thesis, On-site Management of Greywater and Human Wastes (1995), is an examination of effective on-site management and re-use of greywater and human wastes in Australia, with a particular focus on constructed wetlands in domestic situations. His report details the design, installation, and monitoring of four small-scale SSF wetland systems in northern NSW. All the systems received household greywater and systems 1 and 4 also received excess liquid from the composting toilet. The average performance of the first three systems are shown in Table 3.4. No water quality data were obtained from the fourth system but observations were made. System 1 was the the most mature system (12 months) and received the most extensive testing (23 inflow samples and 38 outflow samples). As systems 2 and 3 had no more than 5 effluent samples taken for testing the focus of this section will be based around system 1.

System 1 had a grease trap treating all household greywater prior to its entry into the constructed wetland and final effluent exited to a holding pond. Test results of system 1 show a general trend in a reduction of most parameters from inflow to outflow, with the notable exception of phosphate, ammonia and conductivity (increasing by 254%, 22% and 60% respectively). The system produced an average effluent with 24 mg/L BOD_5 (95% removal), 15 mg/L total nitrogen (58% removal), 5.1 mg/L orthophosphate (254% increase), 8 mg/L suspended solids (90% removal), 1,400 CFU/100mL faecal coliforms (99.9% removal), 630 μS/cm (60% increase) and a pH of 7.1. It must be noted that in system 1 only the first two, of four, wetland cells had fully grown *Phragmites australis*, the last two cells were planted with clumps and rhizomes in the last month of testing (August 1995). Marshall expects treatment to improve and that effluent quality would meet current regulatory standards for on-site sub-surface irrigation re-use once the reeds are fully established over the entire wetland.

Table 3.4: Operational Performance of Three On-Site Constructed Wetlands

Parameters	System 1[1]			System 2[2]		System 3[2,3]	
	In	Out	Eff.%	In	Out	In	Out
Ave Flow Volume, L/day	221.6	201	-	480	-	186	-
Wetland Capacity, L	1160			1770		440	
Design HRT, days	5.3			3.7		2.6	
Water Temp., °C	-	14	-	13	14.5	15	15.6
Conductivity, μS/cm	394	630	-60	2900	1137	310	449
pH	7.2	7.1	-	10.5	9.4	5.14	7.0
Suspended Solids, mg/L	81	8	90	24	31.25	188	11
Dis. Oxygen, mg/L	0	0	-	0	0	0	0.5
BOD_5, mg/L	253	24	91	143	142	404	80.4
Faecal Coliforms, CFU/100mL	1,600,000	1400	99.9	<10	1008	451,600	3500
Phosphate, as mg/L P	2.6	9.2	-254	1.1	0.98	1.64	0.78
Total N, as mg/L N	35.5	15	58	25.1	8.8	15.9	6.42
$NO_2 + NO_3$, as mg/L N	8	0.1	99	0.33	0.9	.036	.06
TKN, as mg/L N	27.5	14.9	46	24.8	8.7	15.9	6.36
Ammonia, as mg/L N	9.8	12	-22	20.2	4.6	0.6	4.42
Number of Tests Conducted	23	38		1	4	5	5

Notes: 1 – receives greywater and excess liquid from a composting toilet.
2 – receives greywater only.
3 – a fourth system was included in the study but as the reeds were planted late in the study no water quality data was conducted.

Source: Adapted from Marshall 1995: 76-96

A hydraulic tracer study was also conducted for system 1. The results from the tracer study indicate that the medium hydraulic retentention time for system 1 is 13.1 days, with the first throughflow appearing after 5.4 days and the peak detection occuring at 7.3 days. As the ideal plug-flow indicates a hydraulic retention time of 5.5 days there is no real short-circuiting through the wetland (Marshall 1995: 86-7, 126-7). Marshall believes that short-circuiting of the system would be extremely hard due to the modular setup of the wetland. However, while no dispersion of lithium has occurred ahead of the flow (short-circuiting) there has been dispersion of lithium behind the flow, last detection of lithium occurring at 24.1 days. Marshall believes that this may be due to the stop-start nature of wastewater flow through the wetland.

Reed growth in systems 1 and 4 was vigorous while systems 2 and 3 showed poor growth. Marshall (1995: 89) believes that systems 1 and 4 showed good reed growth because of the nutrient and trace element inputs received from the composting toilet's excess liquid. Greywater has inadequate levels of trace elements to support vigorous reed growth.

Mitchell's (1995: 903) paper also found that macrophytes planted in a constructed wetland receiving greywater only did not grow as quickly or were not as healthy as those also receiving blackwater. Mitchell's paper focuses on the performance of a SSF constructed wetland treating the wastewater from a single household. Initially the greywater and blackwater were to be treated separately but due to the poor reed growth occuring in the greywater section they were mixed and treated together, dramatically improving reed growth and health. Overall, Mitchell found that the wetland was producing secondary quality effluent in terms of BOD_5 and suspended solids as well as some nutrient removal.

The three constructed wetland systems in the third paper received combined black and greywater from farmhouses in Austria (Perfler and Haberl 1993: 142). The influent into all three systems was primary treated in a settling pit and the beds were planted with *Phragmites australis*. The results from the one year study are summarised in Table 3.5. All three systems were SSF, with system 1 being a single stage vertical flow system, system 2 consisting of two vertical flow stages, and system 3 a two stage horizontal flow system.

All three systems achieved reasonable removal rates with system 1 performing the most consistently across all parameters. Systems 1 and 2 respectively achieved 92% and 94%

removal of BOD_5 and greater than 80% removal for COD and TOC. System 3 achieved removal rates of greater than 70% for these three parameters.

The variability of pollutant concentration levels varied greatly within systems and between systems. For example the BOD_5 influent value range for system 1 varied from 28 mg/L to 330 mg/L producing an average influent concentration of 121 mg/L, while the average influent values for systems 2 and 3 were 78 mg/L and 155 mg/L respectively. The average effluent concentrations of BOD_5 for systems 1, 2 and 3 were 9 mg/L, 4 mg/L and 37 mg/L respectively. No ranges were provided on effluent values across the one year testing period for each system. This variability within and between these Austrian systems highlights the unpredictable nature of wastewater as outlined in Chapter 2.

The elimination rates for nitrogen and phosphorous also varied between systems. System 1 was the only wetland achieving removal rates greater than 60% for total nitrogen, total phosphorous and ammonia (62%, 74% and 79% respectively).

Table 3.5: Operational Performance of Three Austrian On-Site Wetlands

Parameters	System 1			System 2			System 3		
	In	Out	Eff.%	In	Out	Eff.%	In	Out	Eff.%
COD	345	49	86	236	36	82	364	99	74
BOD_5	121	9	92	78	4	94	155	37	78
TOC	103	16	85	70	13	82	132	36	74
NH_4-N	50	11	79	50	22	56	74	39	48
N_{TOT}	70	27	62	60	33	43	87	47	49
P_{TOT}	11	3	74	8	3	57	12	2	80

Note: all influent and outflow values are mean concentrations measured in mg/L
Source: Perfler and Haberl 1993: 145-7

Overall system 1 is the constructed wetland operating most effectively across the range of parameters given, with system 2 showing a similar behavior to system 1 but without reaching the higher removal rates. The horizontal flow wetland (system 3) generally had a lower production performance in all parameters with the exception of phosphorous, where it achieved the greatest removal efficiency of 80%.

3.7.3. *Cold Climate Constructed Wetlands*

In Norway an experiment using a multi-stage system to treat wastewater from two households was undertaken. The system consisted of a three chamber septic tank, a vertical flow sand filter and two separate horizontal SSF constructed wetlands (Jenssen et al. 1997: 246). The first constructed wetland was filled with iron rich sand and the second with a light expanded clay aggregate (LECA®), both substrates are capable of high phosphorous storage through adsorption and precipitation. Jenssen et al. concluded from their preliminary results of the first 18 months of operation, Table 3.6, that high nutrient removal can be obtained using SSF constructed wetlands in cold climates, 63% for nitrogen and 96% for phosphorous. The phosphorous removal rate is so high due to the use of substrates with high phosphorous storage capacity. As can be seen in the last column of Table 3.6, removal rates of pollutants can be further enhanced through the use of multistage systems, in this case utilising reactive porous media and a sand filter for aerobic pretreatment.

Table 3.6: Effluent Values (mg/L) and Removal Rates for a Norwegian Multistage Wastewater Treatment System

Pollutant	STE[a]	SFE[b]	CW1[d]	CW2[e]	% Removal from CW's[c]	Total % Removal
COD	250	100	127	55	45	78
BOD_7	210	90	45	20	78	90
TSS	56	27	26	10	63	82
TN	102	95	48	35	63	65
TP	11	8	1.1	0.3	96	97

Notes: a- Septic tank effluent
b- Sand filter effluent
c- Removal rate calculated from SFE values to CW2
d- Outlet of constructed wetland number 1
e- Outlet of constructed wetland number 2
Source: Jenssen et al. 1997: 247

3.8. Maintenance and Operation

Maintenance is essential to ensure that constructed wetlands are functioning to their designed levels and to maximise the systems lifespan. Basically, the management of constructed wetlands consists of four components; operation control, inspection, maintenance and monitoring (Beharrell et al. 1996: 81). Constructed wetlands only require a small percentage of the energy, time, labour and technical skills needed in conventional centralised sewage treatment systems. Williams et al. (1995: 49) believes that the operating skills required in constructed wetlands are more aligned to agriculture and irrigation rather than the technically demanding procedures needed in conventional treatment plants. Obviously larger systems will require more maintenance and operating time than smaller operating systems but the same principals outlined below apply.

Constructed wetlands that have been properly designed and are correctly managed are 'passive' low-maintenance systems; however, due to the complex and dynamic nature of constructed wetlands problems can occur. Beharrell et al. (1996: 81) state that problems will most likely occur when:

- wetlands are poorly designed and constructed;
- when the operator has inadequate understanding of the system;
- the wetland is overloaded, both in terms of hydraulic overload and pollutant overload;
- natural disasters occur;
- the wetland is plagued by weed problems; and
- excessive amounts of sediment and litter accumulate and are not removed from the system.

Operation and maintenance plans save the operator money as early detection problems result in a longer life span for the wetland and cheaper and simpler solutions, rather than later and larger remedial action (Beharrell et al. 1996: 82). By extending the life of a constructed wetland the necessity for a major refit or decommissioning are delayed or not required. Other duties required in operation and maintenance plans include repair and maintenance of pumps, berms and control structures, vegetation management, and the eradication of unwanted species and weeds (Kadlec and Knight 1996: 701).

Long-term monitoring of constructed wetlands is essential in order to increase the knowledge and awareness of the complex biological systems operating within a constructed wetland (Hicks and Stober 1989: 447). This knowledge and experience will help build an information base that will be beneficial in design processes, provide realistic performance expectations, and contribute valuable advice on ways to operate and maintain a healthy biological system within a wetland, further streamlining and minimising the operation and maintenance workload.

It is also important to monitor for nuisance or hazardous conditions to humans or biota within a wetland (Kadlec and Knight 1996: 707). Conditions to impact adversely on humans include breeding zones in the wetland for mosquitos, odours and open water. Open water can be a problem if humans come into contact with it before it has been adequately treated for pathogens or other harmful substances or chemicals. The operational performance of a wetland can be adversely affected by hostile conditions that dramatically affect a wetland's biota. Examples of these hostile conditions include an increase in parasites for vegetation and toxicity levels.

3.9. Costs

One aspect that will help constructed wetlands gain wide public acceptance is if their costs can be kept within the price range of current forms of on-site wastewater treatment. Discussed below are the costs associated with the construction of wetlands fro wastewater treatment.

Marshall's (1995: 138) study of small-scale constructed wetlands in northern NSW treating domestic wastewater on-site provides an insight into the cost of constructing such systems in Australia. Marshall estimates that an average sized wetland would cost \$1,200 excluding labour, a breakdown of these costs is shown in Table 3.7. The cost of a greasetrap and sand filter are \$100 and \$200 respectively. To maximise the benefits of a constructed wetlands it is essential to have a system that allows the re-use of the effluent. Marshall has valued irrigation equipment and a holding pond or holding tank to allow the re-use of wetland effluent at around \$1,550. An on-site wastewater treatment system that incorporates a greasetrap, wetland, sand filter and re-use of effluent would require about \$900 worth of labour, taking the total cost of such a system to approximately \$3,900. Section 4.3.5 contains the costs associated with the construction of the case study wetland.

Table 3.7: Expected Costs of a Small-scale On-site Constructed Wetland

Component	**Cost**
Wetland Modules @ \$100 each	\$600
Gravel for Substrate	\$200
Reed Planting Stock	\$200
Pipes/Fittings	\$200
Total Cost (excl. labour)	**\$1,200**

Source: Marshall 1995: 138.

For reference purposes the costs associated with the construction of medium to large-scale wetlands are shown in Table 3.8. As can be seen from this table there is a huge variability not only between FWS and SSF wetlands but also within each category. The major costs associated with FWS constructed wetlands include the land, an impermeable liner, earthworks, and dykes or berms (Reed et al. 1995: 279). SSF wetlands require the same costs as FWS systems but have the costly addition of a substrate material, usually gravel, which can cost upto 50% of the capital cost (Kadlec and Knight 1996: 725-7). SSF wetlands require a smaller land area than FWS systems to treat the same volume of wastewater, which can be a considerable saving depending on land prices.

Table 3.8: Capital Costs of Constructed Wetlands ($US/hectare)

Type of Wetland	Kadlec and Knight (1996: 725-7)		Tchobanoglous (1997: 82)		Reed et al. (1995: 279)
	Cost Range	**Median Cost**	**Cost Range**	**Median Cost**	**Approximate Cost**
FWS	$10,000-$100,000	$44,600	$5115-$160,550	$44,460	$74,000 /ha
SSF		$358,000	$178,388-$247,000	$216,273	$111,000 /ha

Operating and maintenance costs consist of pumping, monitoring, dyke maintenance, equipment replacement and repairs, and harvesting if incorporated into the operating plan (Kadlec and Knight 1996: 635). No chemical purchases are necessary, there are no significant time requirements for semi-skilled employees and there is no need for highly trained staff. Annual operating and maintenance costs for large-scale constructed wetlands range from between $US5,000 and $US50,000 per year.

3.10. Summary

This chapter has provided an insight into wetland components, treatment processes, types of systems in use and considerations that should be taken into account at the design stage. Constructed wetlands provide a robust, reliable and passive wastewater treatment system. Relatively low costs are involved in a constructed wetlands operation as they require low levels of labour and technical ability and no chemical inputs.

The two main types of constructed wetlands are SSF and FWS systems. Both have different flow patterns and treatment processes that provide each with distinct advantages and disadvantages in certain situations. The most suitable wetland design for domestic wastewater treatment in temperate climates is the SSF system as it minimises public health risks, pest/insect vectors, requires a smaller treatment area, and operates more consistently in winter periods.

Chapter 4. Case Study: On-site Reed Bed System

No constructed wetlands treating domestic wastewater on-site could be found operating in Tasmania. As a result the author decided to set up a single household case study, which would provide a valuable insight and knowledge into the performance of constructed wetlands operating domestically in Tasmania. Knowledge would be gained in the areas of design, construction, maintenance, costs, plant growth, and treatment capabilities. The case study is described in detail below along with testing procedures and parameters and a discussion of results.

4.1. Study Location

The case study wetland is located on a farm 35 kilometres from Hobart. The wetland is intended to treat the greywater from a single household permanently occupied by two adults in their early 50s. The initial wastewater treatment system consisted of a septic tank treating only blackwater and a separate greywater pipe that drained onto a well-fenced paddock. The paddock has a south-facing slope and has been used for the grazing of Angora goats.

4.2. System Summary

The constructed wetland was completed in December 1996 and started receiving greywater at this time. The greywater flows from the house through a basic gravel filter before entering a combined up and downflow SSF constructed wetland. The wetland is planted with *Phragmites australis* in a gravel substrate. Outflow from the wetland runs through a basic gravel filter before flowing into the paddock for soil absorption. The new system has been fenced off. As this study is focusing on the treatment of greywater within constructed wetlands the blackwater was treated in the same way as before, by septic tank and absorption into the ground.

Unfortunately, financial constraints did not allow the 'ideal' constructed wetland set-up. If greater funds had been available a greasetrap would have been purchased to pretreat the greywater before entry into the wetland, a sand filter for post treatment and the construction of a holding pond to allow the re-use of final effluent.

4.3. System Detail

4.3.1. Filters

As discussed in Chapter 2 greywater contains many contaminants, including solids in various forms. These solids can be a contributing factor to the clogging of a SSF constructed wetland, resulting in a short-circuiting of the treatment process. Filters are a reliable method of decreasing solids entering into the wetland, thereby decreasing potential clogging problems. There are two basic free-draining gravel filters operating within the treatment system. The first pre-treats greywater before it enters the wetland and the second provides post-treatment of the effluent exiting the wetland. Both filters used are cheap but effective and consist of 12-17mm gravel in a 55-litre container with an outlet pipe situated at the base.

4.3.2. Constructed Wetland

As stated in Chapter 3, SSF systems appear to be the best type of constructed wetlands to use on-site because they provide a safe and effective method of treatment and present less odour and pest problems. Horizontal SSF systems are not as efficient as upflow or downflow systems because the wastewater/root-zone contact is not maximised. The wetland in this case study is a combination of the downflow and upflow designs outlined in Chapter 3. It consists of five modules and provides a theoretical hydraulic retention time of 6 days (for calculations refer to section 4.3.3 below). Each module has a surface area of 1 m^2 and a depth of 0.85 m. The total wetland comprises an area just under 6 m^2.

Figure 4.1 shows the layout of the five modules. The five modules are split into three sections, modules 1 and 2 make up the first section of the wetland, modules 3 and 4 the second section and module 5 the last section. Within each section the modules are joined at the base with 80mm flexible tubing, allowing the greywater to flow to the next module. At the end of module 2 and 4 the greywater flows from the top of the module, via a 50mm PVC pipe, onto the substrate surface of the next module.

Greywater enters the first section of the wetland at the substrate surface of module 1 and exits the outflow pipe in module 2, 2.5 cm below the substrate level. The greywater is forced to flow down through the root system in module 1 and up through the root system in module 2, resulting in a combined downflow and upflow SSF system as shown diagrammatically in Figure 4.2. The greywater then enters the second section of the wetland, modules 3 and 4, where the same process occurs. The final section consists of module 5 where the downflow process occurs before exiting at the base of the module into a final gravel filter.

To improve plug-flow within the wetland each downflow module has a network of perforated pipes covering the base and each upflow module has an outlet pipe spanning the width of the module. This allows for a more even collection and distribution of the greywater across the modules, decreasing the chances of short-circuiting due to dead pockets within the wetland.

The land where the constructed wetland is situated is below the initial treatment and disposal area, allowing the greywater to flow to and through the wetland by gravity. A mini-excavator was used to prepare the site, which involved digging into the hill, levelling the land and making a step from section 1 to section 2 so that they could remain gravity fed. The reasons for the modular arrangement of the wetland include that it is easier to maintain, easily added to, minimises short-circuiting and aesthetics (although not apparent at this stage in the case study). Each module has the capacity to be drained for maintenance reasons if required and is impervious. Plate 4.1 shows the case study site before construction commenced and Plate 4.2 shows the site after construction. In Plate 4.2 there are seven modules. The case study originally intended to conduct two experiments, the first treating greywater only through a constructed wetland and the second treating only septic tank effluent. The two modules on the bottom right hand side of Plate 4.2 were intended to treat the septic tank effluent but unfortunately, the septic tank was experiencing leakage problems and no effluent was exiting the septic tank outlet. By the time that the septic tank was fixed this study was nearing completion and consequently the second experiment of using a wetland to treat septic tank effluent was abandoned.

The common reed, Phragmites australis, was planted in the wetland. Roots and rhizomes were collected from the Derwent River between the Bridgewater Bridge and New Norfolk. The wetlands were planted in December 1996 with 200-300mm centres.

The substrate material used was bluemetal gravel, sized between 12mm-17mm, which allows good permeability, prevents clogging and provides a sufficient rooting medium for the Phragmites australis. While the author tried to wash the gravel before adding it to the modules, silt and other particulate matter remained within the gravel and to remove it all would have required an excessive amount of water. To account for the settling of particulate matter the outlet pipes were raised 2-3cm above the floor of modules 1,3 and 5 and were monitored for any adverse affects. The author calculated that gravel of size 12mm-17mm has a porosity of about 40% (the ratio of voids to the total volume of the gravel), therefore each module filled with 75cm of gravel and a surface area of 1 m^2 will hold 0.3 m^3 (300 litres) of greywater. The overall constructed wetland, five modules, is capable of holding a total volume of 1.5 m^3.

4.3.3. Hydraulic Retention Time

One of the most important considerations in the design process was to ensure that there was adequate treatment of the greywater by providing sufficient hydraulic retention time (HRT). The 'average' person living in an Australian household generates 117 litres of greywater per day (White 1995, in Marshall 1995: 5). In the calculations of HRT the author has taken a larger figure of 125 litres/person of greywater generated per day. Marshall (1997: 36) states that a HRT of 6 days is adequate for a temperate climate. As a module is capable of holding 300 litres of greywater (for calculations refer to section 4.3.2 Constructed Wetland), five modules would be required to provide a design HRT of 6 days for two people generating 125 litres of greywater each per day.

4.3.4. Greywater Sources

All greywater generated by the household passes through the constructed wetland. Sources include a laundry, bathroom and kitchen (including dishwasher). The occupants have no conscious practices regarding water use within the household and do not use 'green' products for household water activities. In design calculations it is assumed that the volume and characteristics of the greywater generated by the household would be average.

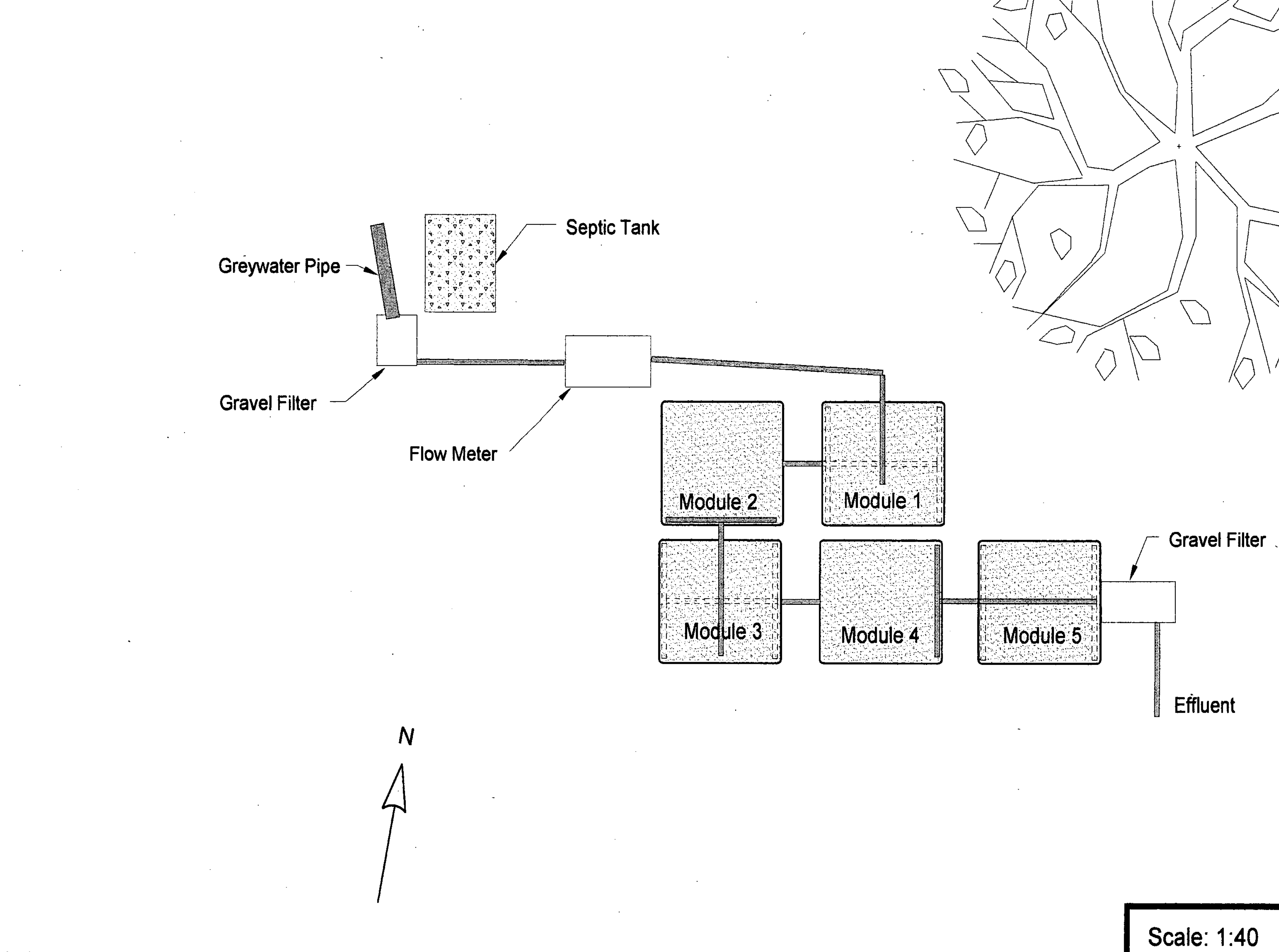


Figure 4.1: Case Study Layout

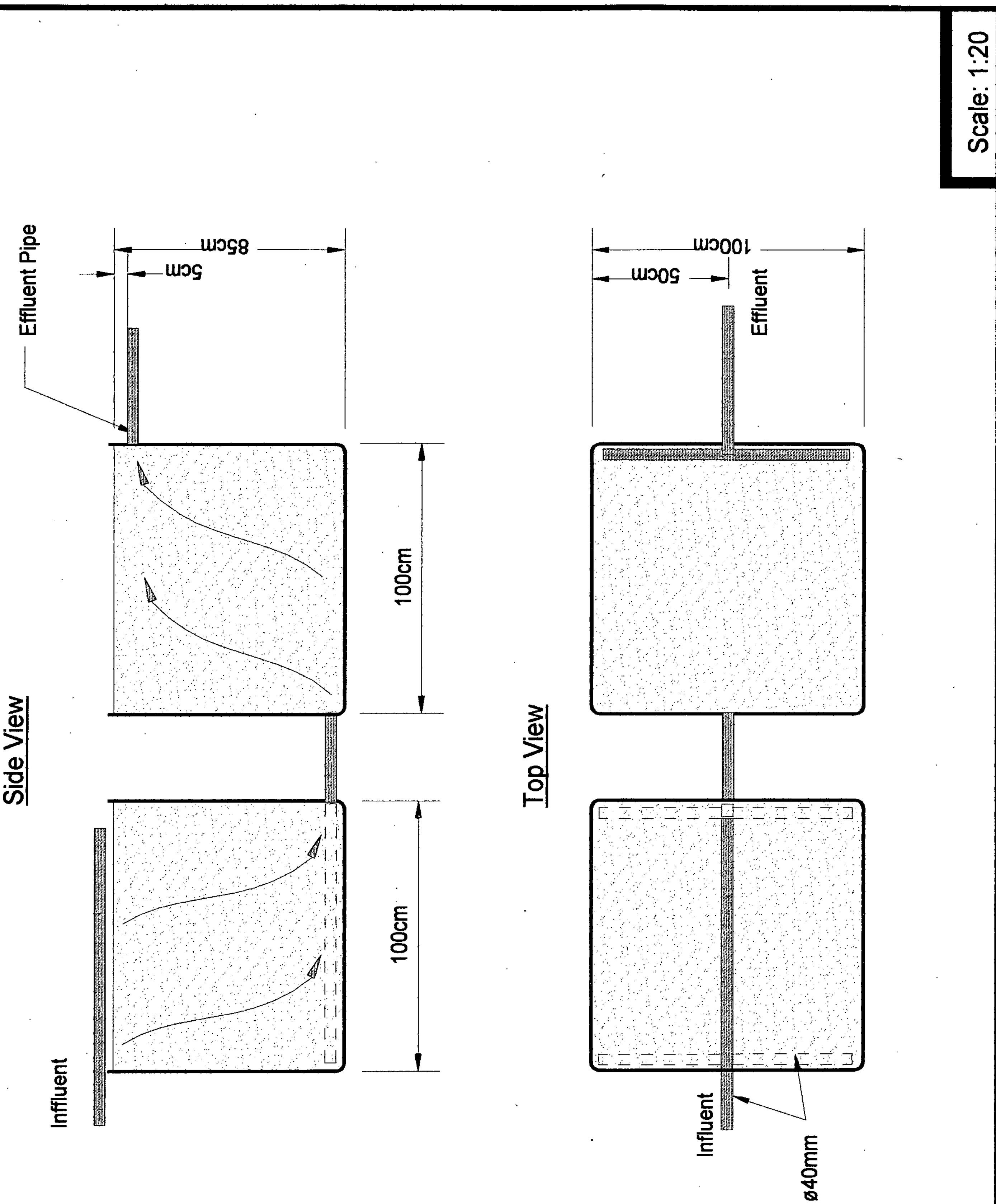


Figure 4.2: Case Study Design

Plate 4.1: Site of Case Study Before Construction

Plate 4.2: Case Study Wetland (October 1997)

Plate 4.3: Flow Meter

Plate 4.4: Dormant Reeds in Module 1 (Winter 1997)

4.3.5. Costs

The total cost of the case study wetland was just under $850. Table 4.1 lists the costs in the different areas associated with the construction of the wetland. In the construction of the wetland all labour, except in the use of the mini-excavator, was done by the author and has not been factored into the costing below.

Table 4.1: Cost of Case Study Constructed Wetland

Wetland Breakdown	Cost
Modules (5 @ $45)	$225.00
Substrate	$230.00
Excavation Equip. and Labour	$110.00
Plumbing	$242.85
Gravel Filters	$33.65
Total	***$841.50***

Marshall (1995: 138) estimates labour cost associated with the construction of an on-site wastewater treatment system that incorporates a greasetrap, wetland, sand filter and the re-use of effluent would be approximately $900. The total cost of the case study wetland incorporating Marshall's estimate on labour costs would be around $1,750.

For comparison a septic tank and soil absorption trench system cost between $2,000 and $2,500 on a favourable site and up to $5,000 on more difficult site. Aerated wastewater treatment systems cost about $8,000 installed (refer Chapter to 2).

4.4. Water Collection and Analysis

4.4.1. Collection Method

Initial monitoring of the case study wetland commenced in March 1997. Monitoring was done throughout the year until November 1997. Due to restricted resources the chemicals and reagents needed in the testing procedures for monitoring were limited and had to be rationed throughout the year, so not all parameters were tested at each monitoring.

Water was collected in glass jars that were initially acid washed in dilute HCl solution. Before a sample was taken the collection jars were rinsed twice with distilled water and then twice again with effluent on-site. Samples were taken after the first gravel filter (but prior to the greywater entering the constructed wetland) and at the outlet of the gravel filter receiving effluent exiting the wetland.

As discussed in Chapter 2 greywater is extremely variable in nature, depending upon the source and characteristics of the household. To overcome this variability a collection bucket was used to take greywater samples over a 20-26 hour period. Clear tubing of a 2cm diameter was inserted into the first gravel filter at the same level as the outflow pipe. The day prior to monitoring the occupants of the house would open the tap on the tubing so that a representative sample of the greywater would enter the collection bucket (a 55-litre rubbish bin). The next day the collection bucket would be gently stirred to ensure it was sufficiently mixed before a sample was taken. The collection bucket was then tipped into module 1 of the wetland. This method of greywater collection allowed a representative sample to be taken from all greywater generating activities over a 20-26 hour period.

Most test parameters were completed on-site. Any parameters not tested on-site were tested within 24 hours of the sample being taken. For tests off-site the sample jars were filled to the top so no air was trapped inside and stored below 4°C.

4.4.2. Test Parameters

The full list of parameters that were tested in the monitoring of the case study wetland's influent and effluent are summarised in Table 4.2 and their testing methods are described below.

Dissolved Oxygen was conducted on-site using a WTX-Oxy96 microprocessor DO meter. This was calibrated prior to each test.

pH was measured on-site using a pH320 microprocessor. This was calibrated to pH 4 and pH 7 standards prior to use. Temperature was measured using the inbuilt thermometer on the pH meter.

Conductivity was measured on-site using a LF320 microprocessor conductivity meter. This was calibrated monthly with 720 :S/cm and 6700 :S/cm standards.

Turbidity was measured on-site using the Absorptometric Method with a Hach DR2000 Visible Spectrophotometer. This method measures the optical property of a water sample that results from the scattering and absorbing of light by the particulate matter present.

Dissolved Reactive Phosphorous (orthophosphate) was measured on-site using the Amino Acid Method with a Hach DR2000 Visible Spectrophotometer. This method comprises two steps (Hach 1989: 39). The first step involves the reaction of orthophosphate with molybdate in acid solution to form a phosphomolybdate complex. The second step involves reducing the phosphomolybdate complex with an amino acid to a molybdenum blue compound, which was then analysed spectophotometrically.

Nitrite was measured on-site using the Diazotization Method with a Hach DR2000 Visible Spectrophotometer. This method involves the reaction of nitrite ions with sulfanilic acid to form an intermediate diazonium salt (Hach 1989: 31). This reacts with chromotropic acid to form a red-orange complex, which is then analysed spectophotometrically.

Nitrate was measured on-site using the Cadmium Reduction Method with a Hach DR2000 Visible Spectrophotometer. In this method cadmium metal is used to reduce nitrates to nitrites (Hach 1989: 30). Next, the nitrite ions react with sulfanalic acid to form an intermediate diazonium salt that, when coupled with gentisic acid, forms an amber coloured compound. The intensity of the amber colour is then analysed spectophotometrically to determine the nitrate concentration.

Ammonia was measured on-site using the Salicylate Method with a Hach DR2000 Visible Spectrophotometer. This method involves multiple reactions before finally yielding a green colour (Hach 1989: 28). The intensity of the green colour is then analysed spectophotometrically to determine the ammonia level.

5 Day Biochemical Oxygen Demand was tested off-site at the Government Analytical and Forensic Laboratory.

Faecal Coliform tests were conducted off-site by Aquahealth, an independent laboratory, using the membrane filtration method.

Suspended Solids were analysed in the university lab by drying and weighing glass microfibre GF/C filters before and after suction filtration of a known volume sample. Filter papers were dried thoroughly before and after filtration in a hot oven. The filters had a nominal pore size of 0.45:.

Flow Volumes were measured using a tipping bucket and magnetic counter. The tipping bucket was installed after the first gravel filter, prior to the effluent entering the wetland. The unit was made from fibreglass and had a magnetic counter fixed to a metal arm

(Plate 4.3). The magnet was attached to the centre of the bucket, so every time the bucket tipped the magnet would trigger the counter. This method allowed measurement of water volumes with minimal disturbance to the flow of greywater. Each side of the tipping bucket was calibrated and averaged 2.38 litres and 2.44 litres. Hence every two clicks of the counter represented a liquid volume of 4.82 litres.

Reed Growth was recorded periodically from December 1996 to January 1998. All root and rhizome plantings were measured in modules 1 and 3. From each planting the total number of shoots and reeds were recorded along with the height of the tallest reed from each planting. Heights were measured using a tape measure from the gravel surface to uppermost node where the top leaves stemmed. During winter when senescence occurred the state of the reeds was also recorded.

Two Tracer Studies was conducted between Tuesday 24 February 1998 and Sunday 1 March 1998. Cooking salt was used as the tracer as there are no active processes operating within a wetland to decrease salt levels. To measure the levels of salt as the effluent exited the wetland a conductivity meter (H20 Multi-Probe) was used in conjunction with a data logger (HydroLab- Surveyor3). The data logger enabled conductivity levels to be recorded on a regular basis without the necessity of the author being present.

Table 4.2: Summary of Monitoring Parameters and Test Site

Test Parameters	Testing Place
Dissolved Oxygen	On-site
Turbidity	On-site
Dissolved Reactive Phosphorous (Orthophosphate)	On-site
pH	On-site
Temperature	On-site
Conductivity	On-site
Nitrate	On-site
Nitrite	On-site
Ammonia	On-site
Inflow Volumes	On-site
Reed Growth	On-site
Faecal Coliforms	External Lab
Suspended Solids	Lab
BOD 5 Day Test	External Lab
Tracer Study	On-site

4.5. Results and Discussion

4.5.1. Summary of Pollutant Removal Process

Samples of effluent entering and exiting the the case study constructed wetland were collected and analysed from March 1997 to November 1997. Table 4.3 provides a summary of the case study's operational performance for the testing period. Further descriptive data (median, standard deviation, range and number of samples) are given in Table 4.4.

General trends in the data include a reduction in BOD_5, suspended solids, faecal coliforms, turbidity, nitrite and nitrate, and a stabilising of pH. However, the wetland effluent is experiencing increases in ammonia and phosphorous.

Table 4.3: Operational Performance Summary of Case Study Constructed Wetland

Parameters	Inflow			Outflow			Removal Eff.(%)
	Min.	Max.	Ave.	Min.	Max.	Ave.	
Est. Ave. Flow per Day, L	250						
Wetland Capacity, L	1,500						
Est. Hydraulic Retention Time, days	6 days						
Conductivity (μS/cm)	157	1439	552	332	660	537	2.6
pH	6.64	10.6	8.48	7.44	8.2	7.69	-
SS (mg/L)	4.3	1382.61	395.8	2.3	187	48.5	87.8
Turbidity (FTU)	40	185	97.6	13	52	30.2	69.1
Dis. Oxygen (%)	12	80	44.4	11	69	38	-
BOD_5 (mg/L)	100	240	160.2	34	75	64.0	60.0
Faecal Coliforms (CFU/100mL)	630,000	5,000,000	2,332,500	490	52,000	18,427	99.2
Phosphorous (mg/L P)	0.8	13.4	4.4	2.77	10.2	6.7	-51.6
Nitrate- NO_2 (mg/L)	0	6.5	3.1	0.2	7.5	2.9	6.2
Nitrite- NO_3 (mg/L)	0.004	0.046	0.022	0.0	0.034	0.008	64.6
Ammonia (mg/L N)	0.02	2.9	0.9	1.02	5.2	2.2	-143.7
Water Temperature (°C)	9.5	21.4	15.2	7.1	21.6	13.7	-

One of the benefits in utilising constructed wetlands for wastewater treatment is their capability to remove and stabilise pollutants from incoming greywater, which is extremely variable in nature. The stabilisation of pollutants is highlighted in the results where the range is less extreme for the outflow than inflow for all parameters except ammonia and nitrate (Table 4.4).

Certain parameters were unable to receive extensive sampling due to limited finances. Analysis for BOD_5 and faecal coliforms were both conducted through independent laboratories and were relatively expensive. As a consequence only 10 BOD_5 tests were conducted and 11 tests for faecal coliforms (Table 4.4). Tests for other parameters also had to be rationed throughout the year to ensure that allocated reagents/equipment etc. were not used up before the end of the year.

Reeds play an important role in the transformation or elimination of pollutants in constructed wetlands (Reed et al. 1995: 185). The importance of reeds in a wetland system is highlighted from a study of SSF wetlands in Santee, USA, which found that unvegetated beds had a much poorer removal performance than vegetated beds (Table 3.1). This study also shows that fully grown reeds with deep root systems provide the most efficient removal process. Reed cuttings for the case study were planted in November 1996 and as a result no testing of the effluent was conducted with fully grown reeds and root systems. As a consequence of testing with immature reeds the results will not show the optimal performance capability of the case study. However a general treatment performance can be deduced with the expectation that treatment performance will improve as the reeds develop deep root systems.

Table 4.4: Further Descriptive Data of Testing Parameters

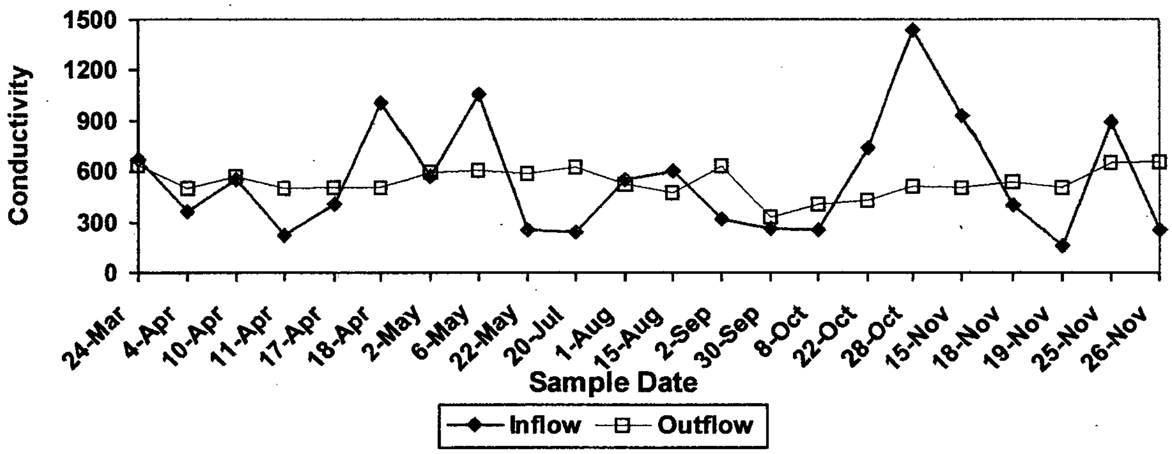
Parameters	Inflow				Outflow			
	Median	Std. Dev.	Range	Samples	Median	Std. Dev.	Range	Samples
Conductivity (µS/cm)	400	356.08	1,282	20	520.5	84.22	328	22
pH	7.83	1.52	3.98	19	7.68	0.175	0.77	21
SS (mg/L)	260	425.47	1,378	13	48	49.175	184.7	13
Turbidity (FTU)	82	44.90	145	19	29	9.862	39	21
Dis. Oxygen (%)	47	21.45	68	19	37	16.898	58	21
BOD_5 (mg/L)	165	55.59	140	5	72	17.219	41	5
Faecal Coliforms (CFU/100mL)	1,850,000	1,869,463	4,370,000	4	21,000	17,859	51510	7
Phosphorous (mg/L P)	3.7	3.00	12.55	20	7.055	2.317	7.43	22
Nitrate- NO_2 (mg/L)	3.2	1.96	6.5	19	2.8	1.666	7.3	22
Nitrite- NO_3 (mg/L)	0.018	0.01	0.042	20	0.005	0.009	0.034	22
Ammonia (mg/L N)	0.59	0.92	2.9	16	1.925	0.981	5.2	16
Water Temperature (°C)	15	3.19	11.9	20	13.75	3.404	14.5	22

4.5.2. *Detail of Parameter Testing*

4.5.2.1. Conductivity

As can be seen from the graph in Figure 4.1 the wetland has created a much more stable/constant reading of conductivity than the greywater inflow. The range for outflow when compared to inflow is much narrower, 328 µS/cm and 1,282 µS/cm respectively. However, the average removal efficiency through the wetland is minimal (2.6%) producing an outflow average of 537 µS/cm. Minimal removal is expected as there are no direct processes at work within the wetland to remove or decrease conductivity levels.

Figure 4.1: Conductivity Results- Inflow and Outflow



4.5.2.2. Dissolved Oxygen

Inflow: Dissolved oxygen levels for inflow ranged from 12% to 80% with an average of 44%. Inflow dissolved oxygen levels are dependent upon the greywater source and the period of time that the sample remains within the collection bucket. Inflow dissolved oxygen levels are not very reliable because prior to reaching the sample bucket the influent receives aeration as the greywater travels through 25 metres of PVC piping and one rough gravel filter. This aeration process would initially raise dissolved oxygen levels within the sample bucket. However, as long as a sample remains within the collection bucket dissolved oxygen levels will be lowered as the relatively high levels of BOD and suspended solids would cause rapid oxygen depletion.

Outflow: Dissolved oxygen levels are usually quite low in SSF constructed wetlands as anaerobic and anoxic conditions are dominant (Marshall 1995: 103). Results from Marshall's study (1995: 103) of on-site constructed wetlands confirm low or non-existent dissolved oxygen levels, two systems involved in the study measured 0 mg/L and a third system recorded only 0.5 mg/L. A final rough gravel filter was designed into the case study in order to overcome low levels of dissolved oxygen as effluent exited the system. The final filter succeeded in its task as the average dissolved oxygen level for outflow was 38%.

4.5.2.3. Faecal Coliforms

Faecal coliform tests were conducted by an independent laboratory and incurred a cost, as a result this study's budget could only afford 11 tests (4 inflow, 7 outflow).

Inflow: Faecal coliform levels in the greywater ranged from 6.3×10^5 CFU/100mL to 5.0×10^6 CFU/100mL with an average of 2.3×10^6 CFU/100mL. The average inflow level falls within the faecal coliform levels for combined greywater from an average household, 1.8×10^4 CFU/100mL to 8×10^6 CFU/100mL (Jeppesen and Solley 1994: 19). However, the inflow values may be overstated by a factor of 10 to 100 as microbial populations have the potential to increase in numbers when stored for up to 48 hours (Rose et al. 1991, in Jeppesen and Solley 1994: 21). The inflow samples were collected and stored in a 55 litre rubbish bin over a 20-26 hour period in order to gain a reprentative sample of household activities over the period of one day. This storage time may have increased the population density of faecal coliforms within the sample bucket, potentially overstating the recorded inflow value for faecal coliforms.

Outflow: Faecal coliform levels in the wetland outflow have been quite variable, ranging from 490 CFU/100mL to 52,000 CFU/100mL with an average of 18,427 CFU/100mL. The highest reading of 52,000 CFU/100mL is more than double any of the other readings and occurred on 22 October 1997, a morning where three loads of washing entered the wetland before a sample was taken. Such an excessive hydraulic load greatly reduces the hydraulic retention time within the wetland by pushing the wastewater through the system a lot more quickly, decreasing the system's treatment capacity. The buffering capacity is an important factor that must be designed into constructed wetlands and is discussed further in section 4.6.1.

As can be seen from Table 4.5 only one outflow reading was below 1,000 CFU/100mL. This is not satisfactory and a much better performance is required before any system operating at these levels is approved for domestic use, unless they are used in conjunction with absorption trenches or the effluent receives some form of disinfection. It is expected that treatment levels will improve as the reeds and their root/rhizome systems grow but to what level is unknown. The safest option would be to increase the hydraulic retention time of the wetland by a few more days to enhance the chances of reducing population numbers through natural die-off, predation, filtration, sedimentaion or biological deactivation (Kadlec and Knight 1996: 535, Griggs and Koosterman 1988: 11).

Table 4.5: Faecal Coliform Levels

Sample Date	Inflow Reading (CFU/100mL)	Outflow Reading (CFU/100mL)
2/9/97	1,800,000	490
30/9/97	5,000,000	25,000
17/10/97	-	3,900
22/10/97	630,000	52,000
28/10/97	1,900,000	4,600
19/11/97	-	22,000
25/11/97	-	21,000

4.5.2.4. BOD

BOD_5 tests were conducted by an independent laboratory and incurred a cost; as a result the study's budget could only afford 10 tests (5 inflow, 5 outflow).

Inflow: BOD_5 results are shown in Table 4.6. The greywater inflow ranges from 100 mg/L to 240 mg/L with an average of 160 mg/L.

Outflow: On average the wetland removed 60% of inflow BOD_5, producing an effluent with an average of 64 mg/L and a range of 34 mg/L to 75 mg/L. Reed et al. (1995: 187) report that BOD_5 levels of less than 20 mg/L are consistently achieved in established North American wetlands. Established wetlands have deep root penetration into the substrate and are therefore capable of transfering greater levels of oxygen throughout the substrate. Root penetration and efficient BOD_5 removal are highlighted in Table 3.1, where SSF wetlands planted with deep rooted species achieve higher removal rates. A possible reason that the case study wetland is not achieving levels of 20 mg/L may be due to the lack of established reeds with deeply developed root systems. As the reeds mature and the roots systems grow deeper BOD_5 levels should improve.

Table 4.6: BOD_5 Test Results

Sample Date	Inflow Reading (mg/L)	Outflow Reading (mg/L)
6/5/97	116	75
2/9/97	180	74
30/9/97	240	65
28/10/97	100	34
26/11/97	165	72

4.5.2.5. Nitrogen Nutrients

As discussed in Chapter 3, nitrogen enters a wetland system in a variety of forms, the major four compounds are organic nitrogen, ammonia, nitrite and nitrate. Unfortunetly during this study organic nitrogen was not measured so a detailed analysis of nitrogen removal is not possible. However, an insight into the nitrogen cycle within the wetland is possible by examining the changes in the three other nitrogen compounds (ammonia, nitrite and nitrate).

As can be seen from Table 4.3, ammonia is increasing through the wetland by 144%. This rise in ammonia can be explained by two factors, organic nitrogen changing to ammonia and the lack of aerobic conditions within the substrate to promote nitrification.

Organic nitrogen entering a wetland is usually associated with particulate matter such as organic wastewater solids (Reed et al. 1995: 191). The initial removal of organic nitrogen is usually quite rapid due to filtration and sedimentation of the solids, which then undergo decomposition or mineralisation and release ammonia into the water column. This process is responsible for the increase in ammonia levels through the wetland. Unfortunately the major mechanism for ammonia removal within a constructed wetland, biological nitrification, does not appear to be operating sufficiently within the case study. Nitrifying microorganisms require oxygen to operate effectively; therefore, aerobic conditions must be present within the substrate before nitrification will occur. As the case study wetland is still young without well established reeds and root systems there is insufficient oxygen entering the substrate to support the nitrifying organisms in the conversion of ammonia to nitrite and nitrate. This relationship is illustrated in table 3.1, where the depth of the root penetration in the SSF wetlands is compared with the removal efficiency of ammonia. The data in this table show that *Scirpus* had root penetration to the floor of the 76 cm reed bed and achieved 94% removal of the applied nitrogen, *Phragmites* with roots to 60 cm achieved 78% removal and *Typha* with roots to 30 cm removed 28% of the applied nitrogen.

The final stage in the nitrogen cycle is the denitrification process where nitrite and nitrate are converted to nitrogen gas, nitrous oxide or nitric oxide. Nitrite is being removed quite effectively from the wetland at a rate of 65% to produce an average outflow value of 0.008 mg/L. The efficiency of nitrate removal is much lower at 6% with an average outflow value of 2.9 mg/L. Reed et al. (1995: 195) state that the availability of an adequate carbon source tends to be the most dominate factor in biological denitrification of nitrate. Major sources of carbon in SSF wetlands are from organics present in the wastewater or naturally present in the wetland. There is a lack of naturally occurring carbon within the wetland due to the young age of the reeds, which do not have any decaying plant litter to release a supply carbon into the water column necessary for denitrification. While carbon sources within the greywater are being removed throughout the wetland (60% reduction in BOD_5 and an 88% reduction in suspended solids) there should still be a sufficient quantity of carbon present to aid in the denitrification process. Reed et al. (1995: 195) estimate that 5-9 g of BOD are required to denitrify 1 g of NO_3-N (nitrate). The average BOD_5 value for the wetland outflow is 64 mg/L, which should be able to denitrify 7.1 mg/L of nitrate, well above the average inflow and outflow values.

This suggests that there is another factor in the relatively low removal efficiency for nitrate. Effluent leaving the wetland passes through a free draining gravel filter that would provide adequate aerobic conditions for nitrifying microorganisms to convert some ammonia to nitrate in the last stage of the treatment process. As this is the last stage of the process before sampling there is no time for denitrification to occur, which may raise nitrate levels between the effluent entering and exiting the final gravel filter.

It can be assumed that with the 144% increase in ammonia, from the inlet through to the outlet, that the case study wetland is removing an unknown percentage of organic nitrogen by converting it to ammonia. It is at this stage that the process of nitrification is not optimised as there is not enough oxygen to support the nitrifying bacteria.

Reed et al. (1995: 191) state that the potentential for nitrogen removal may take several years to develop in a wetland system as "it may require at least two or three growing seasons for the plants, root systems, litter layer, soils [substrate], and benthic materials to reach equilibrium." It is therefore reasonable to expect nitrogen removal to improve as the case study wetland ages.

4.5.2.6. Phosphorous

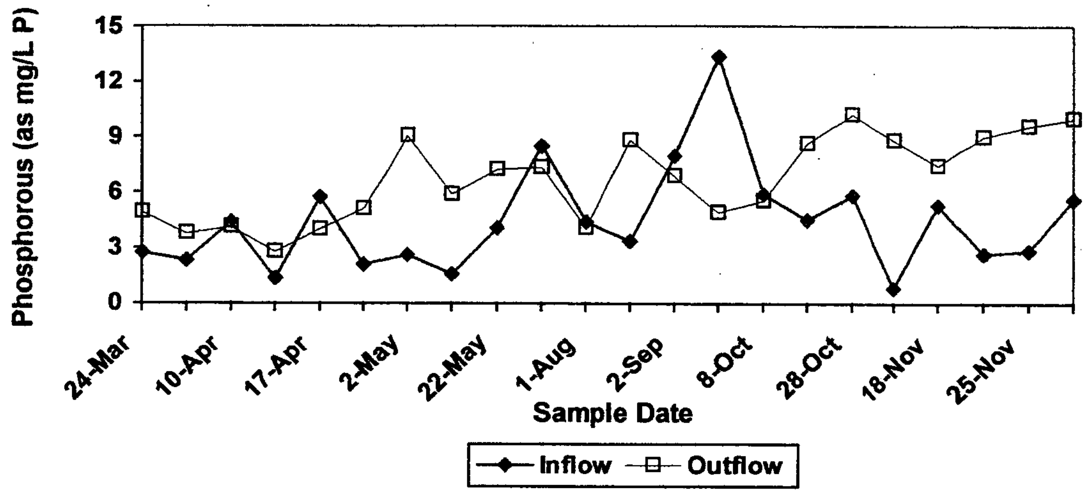
Inflow: Phosphorous readings for inflow were at medium levels, averaging 4.4 mg/L with a range of 0.8 mg/L to 13.4 mg/L. Phosphorous readings are graphed in Figure 4.2 and show erratic patterns for both inflow and outflow.

Outflow: As discussed in Chapter 2 phosphorous is a difficult pollutant to remove in any treatment technology. SSF wetlands remove phosphorous through plant uptake and substrate adsorption at the contact interface (Reed et al. 1995: 196). However, there are few recognised long-term phosphorous removal mechanisms as the substrate has only a limited capacity for adsorption and once wetland plants mature, phosphorous uptake becomes minimal. Also once wetland plants mature, decomposition of plant litter releases some of that phosphorous back into the wetland.

Unfortunately neither of these phosphorous removal processes appear to be working as there is a phosphorous gain of 52% through the wetland producing an average value of 6.7 mg/L. This may be due to phosphorous leaching into the water column from the substrate. The gravel (blue metal) to be used for the substrate arrived at the site unwashed, containing silt and other fine gravelly particulate matter. The author attempted to wash the gravel before placing it into the wetland but found that to wash it thoroughly without the correct equipment would have wasted an excessive amount of water. It may be possible that the gravel and fine silt and particulate matter are slowly leaching phosphorous into the water column, thereby increasing phosphorous levels as the wastewater passes through the wetland.

If phosphorous removal is a major objective of a constructed wetland then it is necessary to use a substrate with the correct chemical and physical properties to promote the adsorption of phosphorous (Kadlec and Knight 1996: 451). Examples of substrate with these properties are iron and aluminum rich materials, limestone media and specially prepared clays.

Figure 4.2: Phosphorous Levels- Inflow and Outflow

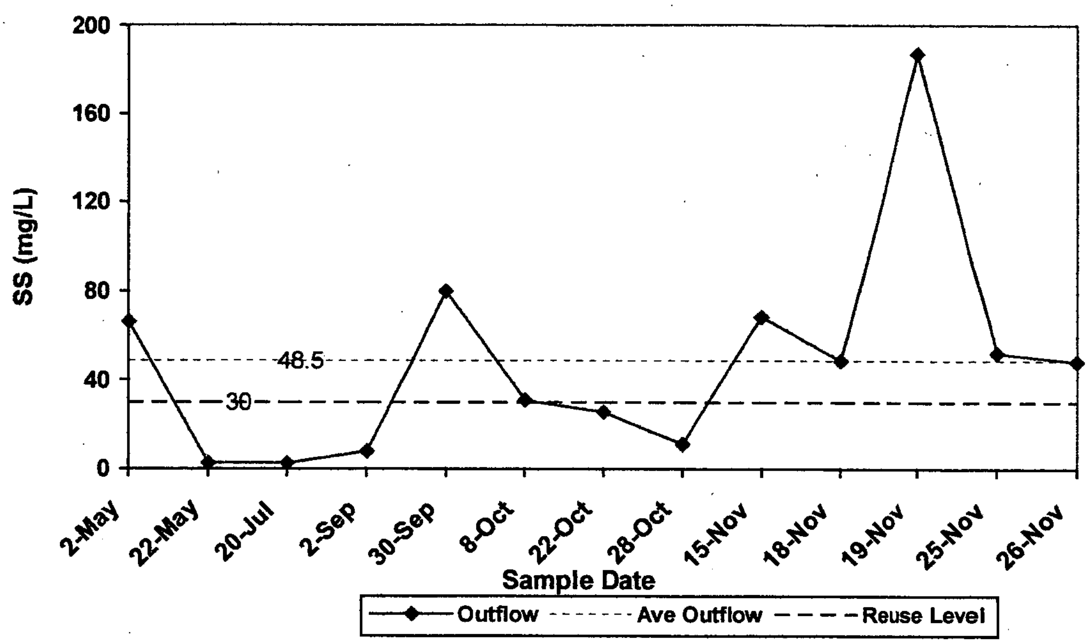


4.5.2.7. Suspended Solids

Inflow: Suspended solid levels in the greywater were very variable with a range of 4 mg/L to 1059 mg/L and an average of 472 mg/L. These levels are high and it is possible to reduce them dramatically by installing a greasetrap. A system that used a greasetrap to pretreat domestic greywater before it entered an on-site constructed wetland achieved much lower levels of suspended solids. The inflow from this system averaged 79 mg/L with a range of 45 mg/L to 140 mg/L (Marshall 1995: 103). Before this greasetrap was installed suspended solid levels exiting an undersized greasetrap were high, ranging from 542 mg/L to 976 mg/L. The installation of a greasetrap may also help reduce the increase in ammonia levels that occur through the case study wetland as suspended solids are partly responsible for the introduction of organic nitrogen into the wetland. With less suspended solids entering the wetland there will be a decrease in conversion levels of organic nitrogen to ammonia, thereby reducing ammonia readings at the outlet.

Outflow: Suspended solid levels decrease through the wetland to an average of 48.5 mg/L, a removal efficiency of 88%. Outflow results for suspended solids are shown in Figure 4.3 along with the outflow average and level required for re-use. The range in outflow results is 2.3 mg/L to 187 mg/L. The peak of 187 mg/L on the 19 November 1997 is over twice the value of any other reading and may be due to three loads of washing entering the wetland that morning. As discussed above in section 4.5.2.3 an excessive hydraulic load into the wetland would greatly reduce the HRT and therefore decrease the system's treatment capacity. The buffering capacity of the system is discussed further in section 4.6.1.

Figure 4.3: Suspended Solids- Outflow



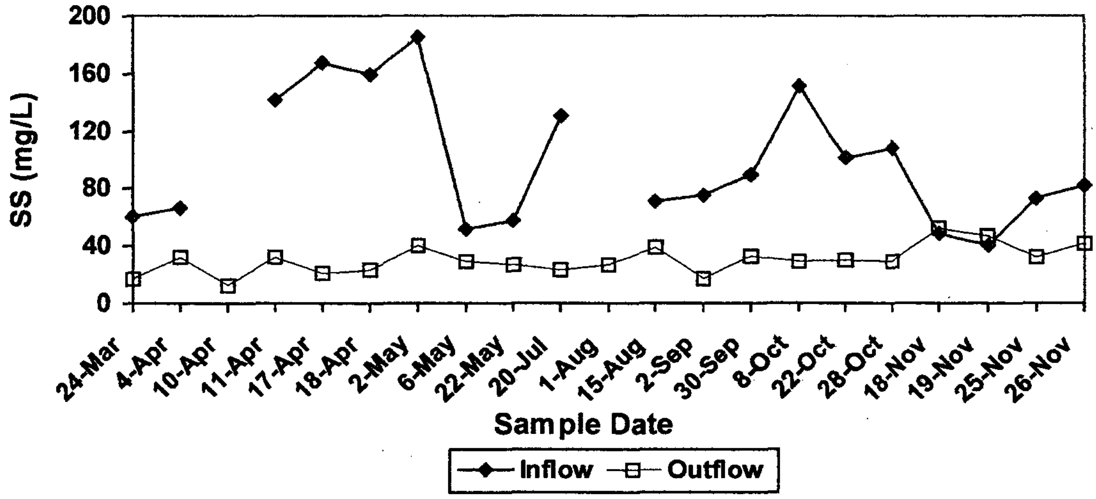
In module 1, where the inflow enters the wetland, there is evidence of solids accumulating on the surface of the substrate. As discussed in Chapter 3 solids have the potential to cause clogging within SSF wetlands, resulting in short-circuiting of the system or overflow of wastewater. To date clogging of the substrate has not been a problem within the case study and it appears unlikely to be a major concern in the near future for three reasons. Firstly, the settling of solids on the substrate surface in Module 1 appears to have reached an equilibrium. The substrate surface of module 1 has a huge amount of visible biological activity, numerous invertebrates ranging from 1mm to 8mm, when compared with the other modules. This biological activity appears to be breaking down the solids of clogging potential as fast as they are entering the wetland. Secondly, the substrate is of adequate size and shape to allow sufficient hydraulic conductivity and drainage pores/holes. Lastly, the stems of the reeds at the substrate surface provide access holes for the wastewater to drain into the substrate.

4.5.2.8. Turbidity

<u>Inflow</u>: The average tubidity level of the greywater inflow was high, 98 FTU, with a range of 40 FTU to 185 FTU. The level of suspended solids has a direct effect upon turbidity levels so the use of a greasetrap for pretreatment of inflow would decrease the turbidity of the wetland influent.

Outflow: As can be seen in Figure 4.4 the outflow turbidity levels are lower and much less variable than the inflow levels. The constructed wetland achieves a removal efficiency of 69% to reach an average level of 30 FTU with a range of 13 FTU to 52 FTU.

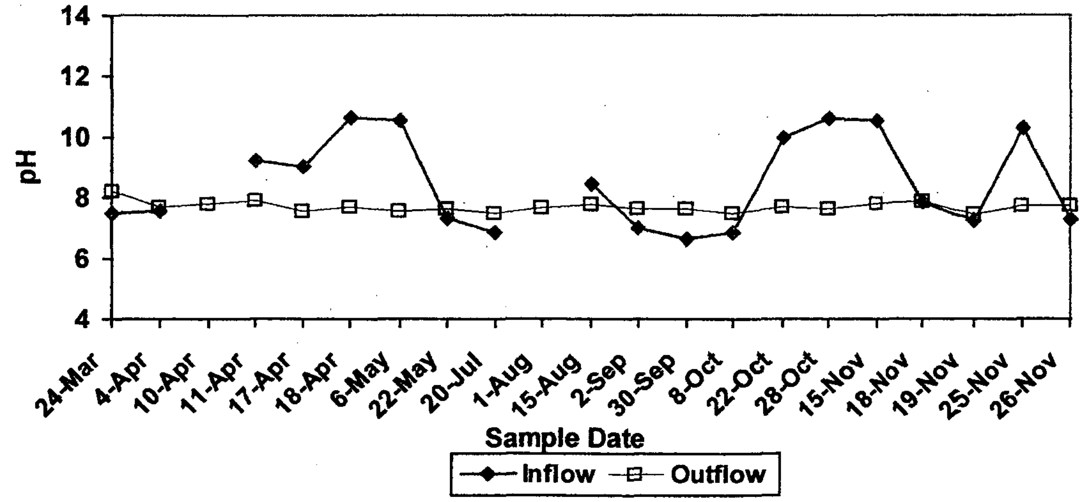
Figure 4.4: Turbidity Results



4.5.2.9. pH

The constructed wetland stablised the variable nature of the greywater influent to an average pH of 7.7 with a range of 7.44 to 8.2 (Figure 4.5). The average pH of the greywater entering the wetland was 8.5 with a range of 6.6 to 10.6.

Figure 4.5: pH Results



4.5.2.10. Temperature

Water temperatures in the wetland reflect the daily air temperatures at the site. Inflow temperatures ranged from 9.5 °C to 21.4 °C with an average of 15.2 °C. These temperatures were influenced by the greywater source (e.g. warm shower water) and the period of time that the sample remained in the collection bucket.

The outflow ranges from 7.1 °C to 21.6 °C with an average of 15.2 °C. The outflow departs the wetland from the base of module 5 and runs up one metre length of tubing to a free draining gravel filter. Therefore the outflow temperature is very dependent upon the ambient air temperature and the period of time that it remains in the tubing. Due to these factors the outflow temperature had a larger range than if the outflow exited directly from the wetland to the filter.

4.5.2.11. Seasonality

Colder conditions through winter trigger senescence in *Phragmites australis* and also slow the activity of microorganisms within a constructed wetland. Both of these factors have the potential to decrease a wetland's treatment capacity throughout the colder periods of the year. However, minimal seasonal variance was found within medium-large-scale Danish wetlands (Kadlec and Knight 1996: 735) and small-scale on-site wetlands operating in the more extreme environs of Norway (Jenssen et al. 1997: 248). Jenssen et al. stated that to minimise the effects of Norway's freezing winters two factors should be considered when designing wetlands, insulation of the wetland and a greater hydraulic retention time. In the Norwegian study Jenssen et al. found that the biological processes of nitrification and denitrification were still occurring in an aerated pond at temperatures between 0°C and 1.5°C.

It has been hard to judge whether or not seasonality has had a dramatic impact upon pollutant removal efficiency of the case study wetland for two reasons. Firstly, only a limited number of tests were conducted through the winter period due to financial constraints. Secondly, the reeds are not fully grown so there is no historical data from this wetland to compare across the seasons while the wetland is operating at optimal capacity. It is expected that there will be minimal seasonal variance with wetlands operating in Tasmania if they are designed along the lines of the Norwegian systems, which experience much harsher winter conditions than Tasmania. Aerobic organisms will also be able to survive within the substrate as the ability of the *Phragmites australis* to transfer oxygen to their roots is not severely affected by senscence (Marshall 1995: 100; Armstrong and Armstrong 1990: 532).

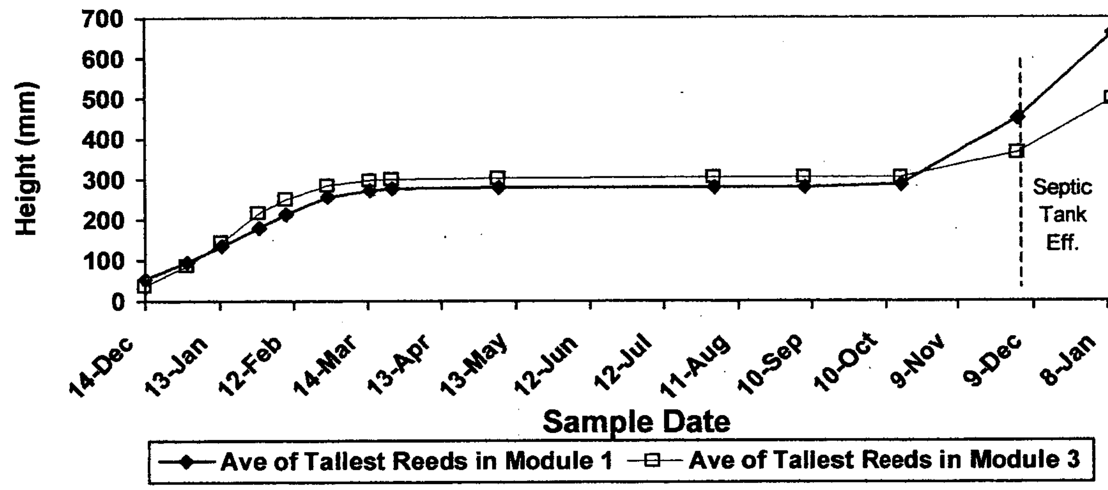
4.5.3. Reed Growth

Clumps/rhizomes of *Phragmites australis* were planted at the end of November 1997. Twenty-six per cent of the clumps failed to survive the transplanting process and on the surviving ones shoots appeared within 4 weeks. Modules 1 and 3 were chosen as the sample for reed growth measurements. The average height of the tallest reed for each clump in modules 1 and 3 is graphed for the year in Figure 4.6. Figure 4.7 graphs the total number of plants and shoots for module 1 and 3 throughout the year.

Reeds and shoot production grew consistently through to April 1997 before the colder weather set in and the reeds began to go dormant. At this time the reeds stopped growing and started to brown off (Plate 4.4). By mid May the reeds had stopped growing and were in senescence with no green aerial parts. At this time the average height of the tallest reed in each clump was 279 mm and 328 mm for modules 1 and 3 respectively. In May there was a total of 126 reeds and shoots in Module 1 with an average of 8.4 per clump, Module 2 had a total of 76 reeds and shoots with an average of 7.6 per clump.

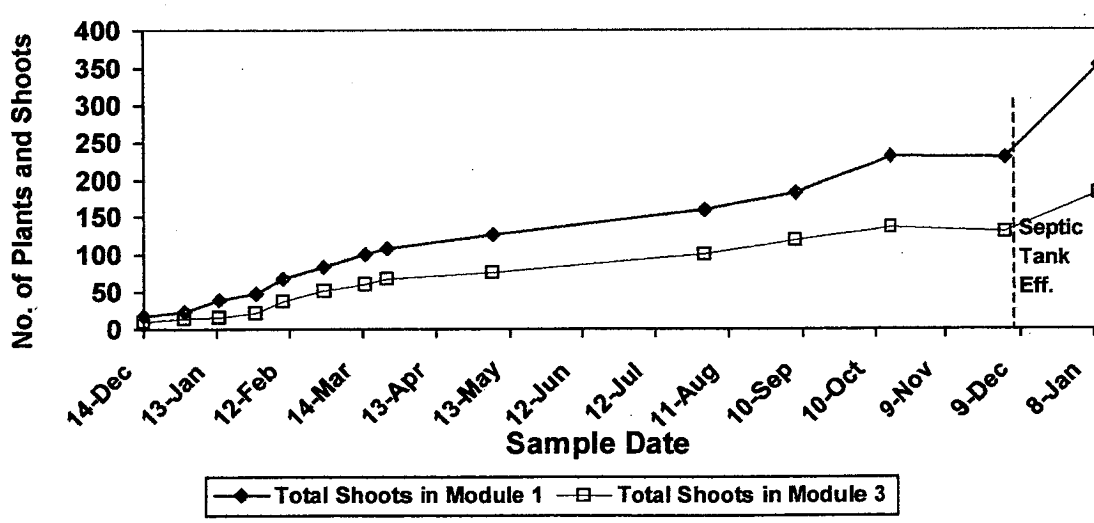
Through winter and the begining of spring the reeds remained dormant. Around the middle of September many of the dormant reeds began to turn green at their tips. Figure 4.6 is a little misleading as the growth lines do not start to really take off again until October; however there was a lot of activity as many of the young shoots were growing well and getting close to the height of the dormant reeds before this date.

Figure 4.6: Average Height of Tallest Reeds



The rhizomes of *Phragmites australis* do not go dormant but continue to grow throughout winter (Hocking et al. 1983: 124). Figure 4.7 confirms this as it shows an increase throughout winter in the total number of plants and shoots for modules 1 and 3. Through winter shoots did emerge but had no or minimal rates of growth, staying very close to the substrate for protection against the cold until spring. The dip in plant and shoot numbers in Figure 4.7 between November and December is because 12%-16% of the dormant reeds did not survive the winter and were no longer counted in the totals.

Figure 4.7 Number of Plants and Shoots Per Wetland Module



As discussed in Chapter 3 it has been recommended to mix greywater and blackwater together to ensure that the reeds receive an adequate supply of nutrients and trace elements that are in limited supply in greywater (Marshall 1995: 89; Mitchell 1995:

903). Marshall (1997: 37) states that for wetlands receiving only greywater the best reed to use is *Typha spp.* On 5 December 1997 septic tank outflow was diverted into module 1 of the case study wetland with the aim of observing if there were any changes in the growth of the reeds. On the 10 January 1998 measurements and observations were taken for modules 1 and 3. The visible difference was very dramatic with all modules showing an increase in reed height and shoot emergence, especially in module 1 where the difference in 5 weeks was the most noticeable. Module 1 had become a thick stand of reeds with the tallest reaching 790 mm and an average tallest reed per clump of 661 mm (increasing from an average of 449 mm on 3/12/97). The average number of reeds and shoots per clump had risen from 15.3 per clump to 25.2 per clump over the five week period. The average in module 3 had risen from 13.1 reeds and shoots per clump to 18.2. The average of the tallest reed per clump in module 3 rose from 402 mm to 546 mm over the same period. The reed growth over the five week period of receiving septic tank effluent is shown in Plate 4.5 and 4.6. Plate 4.5 shows modules 1 and 2 prior to the wetland receiving septic tank effluent and Plate 4.6 shows the same modules after 5 weeks of receiving septic tank effluent.

Weeds within the modules were only a problem where the substrate was more than 5-10 mm higher than the water level. If the substrate was below this level the weeds were not a problem as their roots were constantly wet and unable to survive due to the anaerobic conditions. However, once the reeds were well estabilished they out-competed any weedy areas in the wetland as they blocked out most of the direct sunlight that reaches the substrate.

It is also interesting to note that three tomato plants are growing well on the side of module 1. These plants were self established from seed travelling from the kitchen into the wetland. This is a concern as it suggest that the preliminary filters are allowing relatively large quantities of suspended solids through into the system.

4.5.4. Flow Volumes

Since the installation of the flow meter on 1 August 1997 to the final reading on 10 January 1998 the case study wetland received a total of 51,752 litres of greywater from the household. There are only two permanent occupants living in the house, which means that each is generating 160 litres of greywater per day. This is approximately 40 litres per day greater than the Australian average of 117 litres per day (refer to Chapter 2). No water conservation practices are followed by the occupants of the household. If they were more aware of water use within the house and implemented water saving practices and devices they should be able to decrease water consumption to below the Australian average.

In designing the case study wetland it was estimated that the household would generate 250 litres per day, 125 litres per person per day. At this level the wetland has a design hydraulic retention time of 6 days, assuming no short-circuiting of the system (refer section 4.3.3). But because the actual flow level is nearly 320 litres per day the hydraulic retention time drops to 4.7 days, reducing the period of treatment for the greywater by 31 hours.

Daily flow volumes are extremely variable depending upon the household activities for the day, for example washing days generate a larger volume of greywater. This example is highlighted in the recording of flow volumes where readings were taken in consecutive days. Between 18 and 19 November three loads of washing were done on the morning of the 19th contributing to the total of 774 litres generated for the day (387 L/person/day), while between 25 and 26 November no washing was done and a reading of 206 litres were generated for the day (103 L/person/day).

Due to a limited budget only one flow meter could be purchased and as a result only inflow volumes were recorded. It is expected that the ouflow volume would be slightly less than the inflow volume due to evapotranspiration losses.

4.5.5. Hydraulic Tracer Study

Over a six-day period two hydraulic tracer studies were conducted through the case study wetland using cooking salt. The tracer studies were undertaken to provide an indication of the hydraulic retention time for greywater within the wetland. The first tracer study was started on Tuesday 24 March 1998 and was completed on Saturday 28 March at 3:30 PM. The second tracer study started on Saturday 28 March at 3:40 pm and was completed the next day at 3:10 pm. Figure 4.8 shows the tracer curve for both tests, with data being logged every hour from Wednesday 25 March at 10:10 am to the end of the tracer study on the Sunday. The second tracer study is graphed in Figure 4.9, this tracer curve represents data being logged every ten minutes for the duration of the second test.

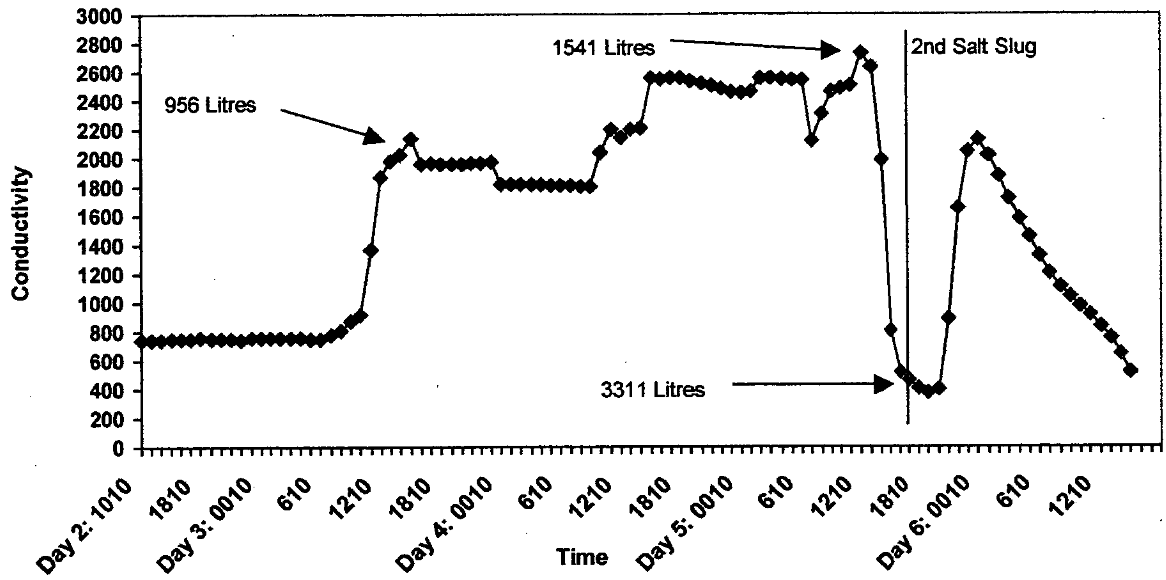
As discussed above in section 4.3.3, the design hydraulic retention time within the wetland is six days. However, the occupants of the house are generating more greywater per day than estimated and as a result the design hydraulic retention time falls to 4.7 days (refer to 4.5.4 Flow Volumes). The objective of the tracer study is to determine whether or not there is short-circuiting of the greywater within the wetland. If no short-circuiting is occurring the salt slug should show up on the tracer curve after 4.7 days or when 1500 litres of greywater has passed through the wetland, which is the wetland's water holding capacity.

On the tracer curve for the first salt slug the conductivity levels begin to rise from midday on Day 3 (Figure 4.8). In a seven hour period conductivity levels rose from 805 μS/cm to 2139 μS/cm. The retention time to the first peak of 2139 μS/cm is only 2.6 days. Two extra people were staying in the house at this time raising the total number of occupants in the house to four people, increasing the volume of greywater generated each day. With a greater volume of greywater passing through the wetland the retention time will decrease so it is more applicable to look at the volume of water that has passed through the wetland than the actual period of days. At the first peak only 956 litres of greywater have entered the wetland, 64% of the wetlands total capacity. This indicates that there is a short-circuiting of the wetland by over one-third, decreasing the actual retention time of the household and its two occupants to 3.1 days. Six days is recommended for constructed wetlands in temperate climates (Marshall 1997: 36) and consequently the three days that the greywater is experiencing in the case study wetland is not an adequate period to provide sufficient treatment of the pollutants. The greywater is only receiving half the recommended treatment period.

The highest peak (2724 μS/cm) on the tracer curve appeared at midday on Day 5 after 1541 litres had passed through the wetland. This indicates that while some greywater is short-circuiting the wetland the peak or highest concentration is travelling through the wetland at the correct pace.

As the occupants had left on a four-day holiday by mid-morning of Day 5 a hose was left running in order to push the remaining salt through the wetland. As a result there is a very sharp decline in conductivity levels after the peak of 2724 μS/cm.

Figure 4.8: Hydraulic Tracer Study Curve
(also showing volume of greywater passing through the system at various stages)



The second slug of salt entered the wetland once conductivity levels had reduced to normal greywater levels again (400-600 µS/cm). The second tracer study was conducted in a different manner from the first, as there were no occupants within the house generating greywater. In order to push the salt slug through the wetland it was necessary to leave a hose with water running throughout the night and into the next day. Once the salt was placed into the wetland a bath with 175 litres was drained, and thereafter the hose was left running throughout the night delivering 121 litres per hour until 12:50 p.m. the next day when the tap was opened up to deliver 331 litres per hour. The second tracer curve is shown continuing on from the first curve in Figure 4.8 where data have been logged every hour. Figure 4.9 shows the second tracer study in more detail with data being logged every 10 minutes.

As water was left running into the wetland throughout the night the peak of the second salt slug passes through the wetland in a much shorter time. The peak from the second salt slug is 2129 µS/cm, 600 µS/cm less than the peak in the first tracer study. This difference can be explained by the second salt slug being pushed through the wetland by fresh water rather than greywater, as in the first tracer study. Fresh water has a lower conductivity reading than greywater. Fresh water at the site had a conductivity level of 47 µS/cm while greywater at the outlet average 537 µS/cm during the testing.

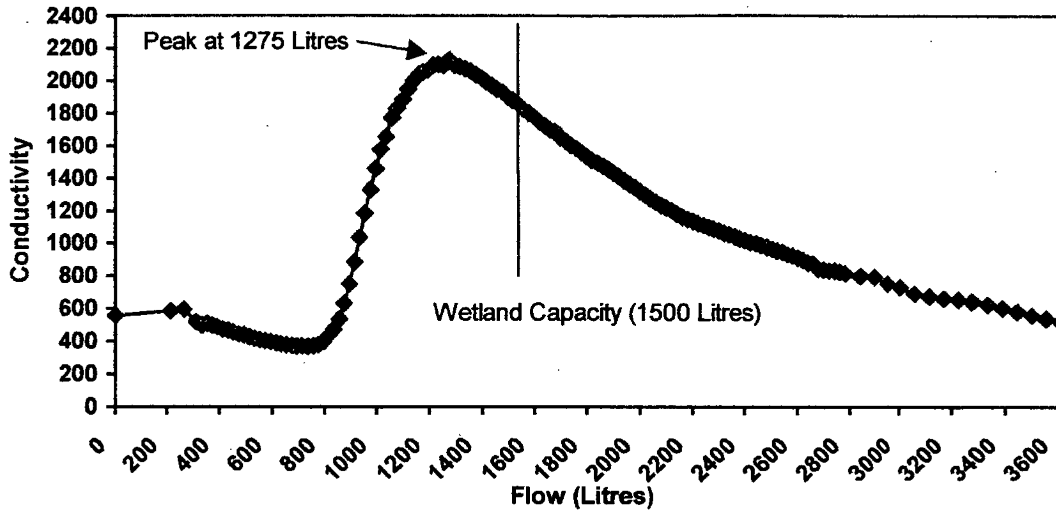
It is interesting to note the difference between the patterns in the first and second tracer studies. The tracer curve for the first study is more irregular than the second curve, with small decreases in conductivity levels as one travels along the curve to the peak. This can be explained by the stop-start nature of greywater generation within a household; certain periods of the day produce greywater while others periods produce none or minimal amounts. The stop-start nature of household activities generating greywater appears more beneficial to the hydraulic flow within the wetland. When greywater stops flowing it settles and disperses in layers across the wetland modules. A continuous flow appears to cause paths of less resistance from the inlet to the outlet of each module, thereby pushing the salt slug through the wetland at a faster rate.

On the second tracer curve (Figure 4.9) conductivity levels begin to rise after 850 litres and peaked at 1275 litres. This means that the salt started to reach the wetland outlet after passing through only 56% of the wetland, short-circuiting the wetland by 44%. The peak of 2129 :S/cm passes through 85% of the wetland, a short-circuiting of 15%. The peak of the first tracer curve passed through at around 1500 litres, the wetland's wastewater holding capacity, showing that the stop-start nature of greywater generation is beneficial to the hydraulic flow within the wetland.

There also appears to be a dispersion of salt behind the main flow. In both the first and second tracer studies conductivity levels did not get below 600 µS/cm until over 3000 litres had passed through the system. This signifies dead pockets within the wetland, resulting in some pollutants receiving twice the required treatment.

The problem of short-circuiting must be addressed within the case study wetland in order to improve the overall operational performance. It is essential that the greywater stays within the wetland for a minimum of 6 days to ensure that the pollutants, especially pathogens and other adverse microorganisms, receive adequate treatment. This issue is further addressed below in section 4.6.1.

Figure 4.9: Hydraulic Tracer Curve for Second Salt Test



The reeds within modules 1 and 2 were adversely affected by the high salt concentrations added to the wetland for the tracer studies. The leaves at the top of the reeds in these modules appeared to suffer a 'burning' effect, as they dried out and faded in colour. The rest of the reeds' leaves were fine and it looks as if the plants as a whole would survive. The reeds in the modules 3, 4 and 5 suffered no visible adverse effects.

4.5.6. Maintenance Requirements

The maintenance requirements of the wetland were minimal. Once the wetland was operational the only maintenance that was required was the removal of weeds from problem areas and the cleaning of the first gravel filter. Once the reeds had grown to a certain height (600 – 700 mm) weeding was no longer required as the reeds out competed the weeds. The gravel in the first filter was cleaned/flushed out three times in a 14-month period. The first gravel filter received fresh greywater from the household and was responsible for removing solids before it entered the wetland. Over time solids built up on the gravel surface, increasing the potential of clogging the filter. In order to make cleaning of the filter easier, mesh has been placed onto the surface of the gravel. Cleaning of the filter now entails the removal, hosing down and replacement of the mesh. As the solids collected in the filter may contain faecal coliforms they should be disposed of in a compost heap or buried to minimise any potential health risks. The second gravel filter treating effluent leaving the wetland has experienced no accumulation of solids on the gravel surface and therefore has not required any cleaning or changing of the gravel.

Plate 4.5: Modules 1 and 2 Prior to Receiving Septic Tank Effluent

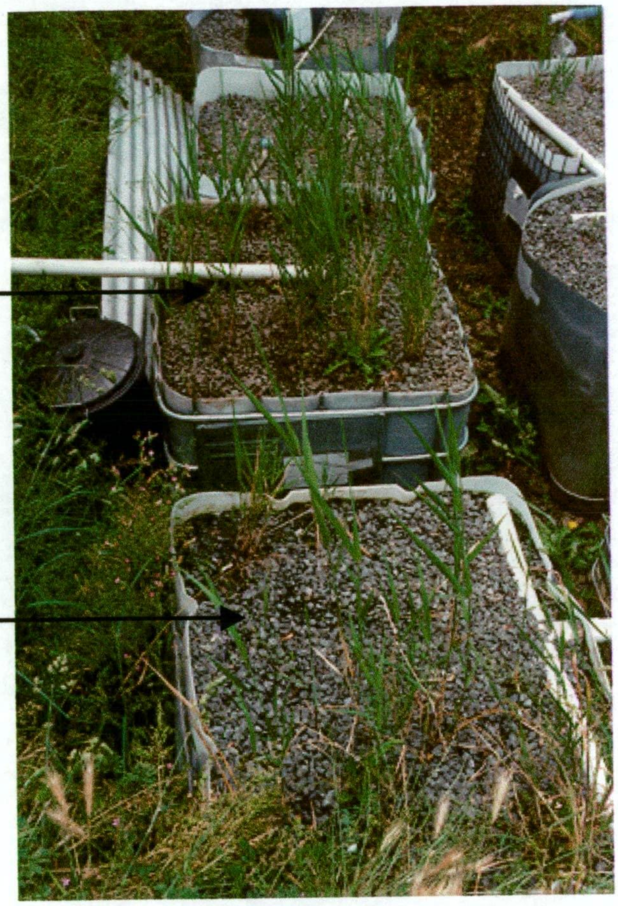


Plate 4.6: Modules 1 and 2 After Receiving Septic Tank Effluent

4.6. Design Improvements

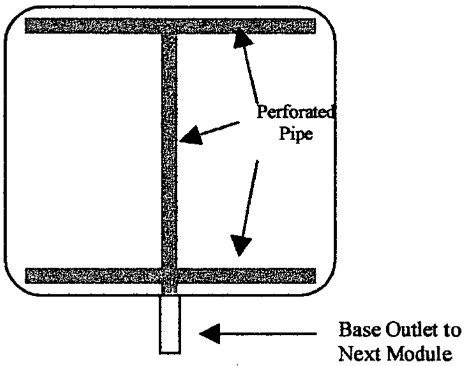
Throughout this study the author has seen areas that could improve the operational performance of the case study wetland or, alternatively, be designed into future constructed wetlands. The areas of improvement or enhancement are listed below.

4.6.1. Buffer Capacity- Increasing Hydraulic Retention Time

In designing a constructed wetland it is very important to calculate inflow volumes carefully to ensure a sufficient hydraulic retention time. Equally important is the provision of a buffer capacity to allow for unexpected loading rates, such as parties or increases in daily activities. An example of increased levels of hydraulic loading were provided in the results section where effluent samples were taken for faecal coliform and suspended solids testing. These samples were taken after three loads of washing were done the same morning. The extra hydraulic loading appeared to be responsible for raising the levels of these parameters to over twice that of any other reading. The rise in washing activity increased the hydraulic load into the wetland, resulting in a decrease of the hydraulic retention time and consequently decreasing the period of treatment.

Analysis of the hydraulic tracer study found that greywater entering the wetland is short-circuiting through the system (refer section 4.5.5). Short-circuiting decreases the period of time that greywater undergoes the treatment processes occurring within the wetland, thereby diminishing the optimal performance of the wetland by not maximising the potential treatment capacity. To prevent short-circuiting a wetland must be designed and constructed to allow a plug-flow through the modules, without any 'dead' pockets. To diminish the effects of short-circuiting within the case study wetland a better mechanism is required for a more even distribution and collection of greywater entering and exiting the individual modules. The current method for collecting greywater at the base of modules 1, 3 and 5 consists of a network of perforated pipes, shown in Figure 4.10. Once the greywater is collected at the base of these modules it simply flows into the outlet pipe and into the next module (upflow modules 2 and 4). In these modules there is no mechanism to provide for an even distribution of the greywater at the base. A network of piping, like that in Figure 4.10, would allow for a more even distribution of greywater across the base of these modules. At the top of modules 2 and 4 greywater is currently collected from only one side of the module, creating 'dead' or 'slow' pockets of greywater along the other three sides. To overcome these dead or slow pockets of greywater it is recommended to place a network of perforated pipes (like that in Figure 4.10) at the substrate surface, allowing a more even collection of greywater across the top of the modules.

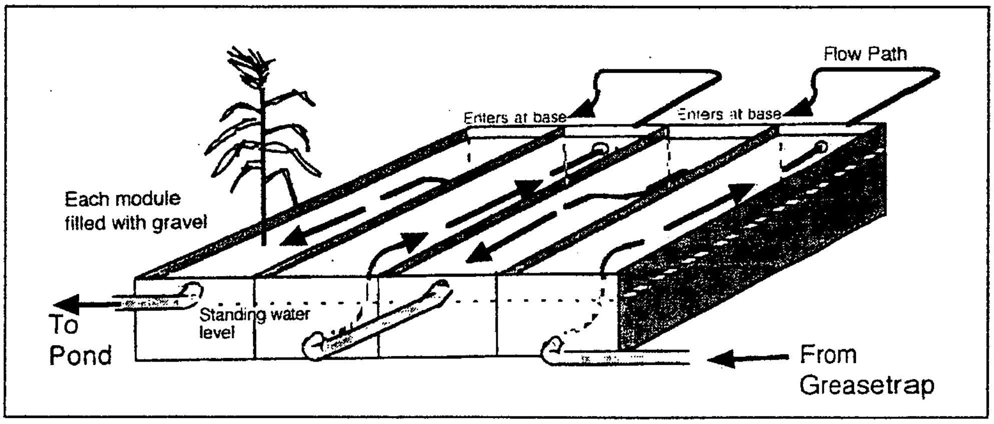
Figure 4.10: Greywater Collection at Base of Modules 1, 3 and 5



It would be expected that with the addition of these piping networks, at positions throughout the wetland discussed above, that the extent of greywater short-circuiting through the wetland would decrease but to what extent it is unknown.

An alternative design that appears to provide a reliable plug-flow of wastewater through a wetland is a modular horizontal SSF system, Figure 4.11. Marshall (1995: 126) conducted a hydraulic tracer study on such a systems and found that the first appearance of the lithium tracer at the wetland outlet appeared after 5.4 days; the ideal plug-flow for this wetland systems was 5.5 days. This means that the wastewater was only short-circuiting the wetland by 0.1 of a day.

Figure 4.11: Horizontal SSF Constructed Wetland Design



4.6.2. Substrate

Gravel to be used for substrate should be washed prior to delivery. Unwashed gravel contains silt, crushed rock and other particulate matter that has the potential to release nutrients into the water column.

Another consideration concerning substrate is the level of phosphorous removal that the wetland is required to achieve. If phosphorous removal is one of the main objectives of a constructed wetland then it is essential to use a substrate with the correct chemical and physical properties to promote the adsorption of phosphorous (Kadlec and Knight 1996: 451). Examples of substrate with a high phosphorous adsorption capacity are iron and aluminium rich materials, limestone media and specially prepared clays.

4.6.3. Pre and Post Effluent Treatment

Unfortunately the lack of funds did not allow the installation of pre and post effluent treatment to the level desired. If the funds were available a large greasetrap would have been installed for pre-treatment and a sand filter and a holding pond for post-treatment. Installation of these forms of treatment would have improved the overall operation performance of the case study wetland system.

It is possible to decrease certain pollutants entering into a wetland by providing pretreatment of wastewater. The case study wetland had a basic gravel filter that removed larger solids from the wastewater but still allowed oils and smaller solids to enter into the wetland. As discussed above in section 4.5.2.5, greasetraps can provide an effective method for decreasing pollutants entering a wetland. This was highlighted in Marshall's study (1995: 103) where an appropriately sized greasetrap greatly reduced the levels of suspended solids in greywater prior to it entering the wetland. By installing a greasetrap the pollutant levels entering the wetland will decrease, which should theoretically improve pollutant levels exiting the system. The greasetrap must be of an adequate size to cope with daily flows and single flow activities from baths and washing machines.

As seen in the results above the wetland effluent contained levels of faecal coliforms and BOD_5 that were still relatively high. The current form of post wetland treatment is a basic gravel filter. The installation of an appropriately sized sand filter may be an option in decreasing the final levels of these parameters in the effluent. As wastewater seeps through the filtration media of a sand filter it receives physical and biological treatment causing a reduction in pollutant levels (Lombardo 1982: 46). The effectiveness of sand filters treating wastewater is highlighted in Table 4.7, which shows the results of a pilot sand filter trialled at a sewage lagoon in Campbell Town, Tasmania, to polish secondary effluent (DELM and DPIF 1996: 90). Table 4.7 shows a significant reduction (>90%) in BOD, biological indicators and non-filterable residue (NFR). The performance of the Campbell Town sand filter illustrates how very effective intermittent, medium flow sand filters are. The introduction of a sand filter as a form of post effluent treatment might polish the case study wetland's effluent to much more acceptable levels.

Table 4.7: Summary Performance of Pilot Sand Filter at Campbell Town

Parameter	**Influent Average**	**Effluent Average**	**Removal Efficiency %**
BOD (mg/L)	60	5	92
NFR (mg/L)	78	4	95
Total Coliforms (No./100mL)	2.60E+06	2429	99.9
E.coli (No./100mL)	4.04E+03	54	98.7

Source: DELM and DPIF 1996: 91

However, sand filters can require a high level of maintenance depending upon the quality of the effluent and hydraulic loading. To give an indication the sand filter operating at Campbell Town (discussed above) received a hydraulic load of 1 m^3/m^2.day and required cleaning every 4 – 6 days of operation (DELM and DPIF 1996: 91). Such levels of maintenance would be too demanding for the majority of on-site systems owners. The introduction of a sand filter and greasetrap will also add to the installation costs of the overall system.

To maximise the benefits of a wetland system the addition of some form of storage facility that allowed for the re-use of the wetland effluent would be needed. The wetland effluent could either be held in a pond system or a storage tank, from which the effluent could be pumped out when needed. The most logical use for the effluent would be for outdoor use on the garden, which currently accounts for 42% of water use in an average Australian household (White 1994 in Marshall 1995: 4).

4.6.4. Insulation and Aesthetics

During testing the wetland modules in the case study remained open and cleared to provide easy access for observation and monitoring of the overall system. It is recommended in cooler climates that constructed wetlands be insulated to provide a more constant performance by protecting against seasonal temperature variations

(Jenssen et al. 1997: 248). Once this study is completed it is the intention of the author to surround the wetland modules with soil in order to insulate the system in colder months and to provide a more aesthetically pleasing form of wastewater treatment than is present at the moment.

4.6.5. Root Depth

In order to encourage deeper and faster root penetration of the reeds into the substrate it is recommended to install a mechanism that allows the operator of the wetland to lower the water level within the substrate. Lowering the water level encourages the reeds to send their roots deeper into the substrate in search of the water. This mechanism would allow a wetland to reach its optimal treatment performance in a shorter period of time. Such a mechanism would be difficult and costly to design into the case study wetland.

A water lowering mechanism is not essential because root depth into the substrate will increase as the wetland plants mature.

4.6.6. Sufficient Nutrients and Trace elements

As discussed in Chapter 3 and presented in the results above, greywater alone does not provide sufficient nutrients and trace elements for the rapid growth of *Phragmites australis*. However, Marshall (1997: 37) does state that certain reed species are more suited to greywater conditions than others, such as *Typha spp.* As *Phragmites australis* does not perform as well in greywater wetlands the case study wetland should either receive effluent from the septic tank or have the present reeds removed and replaced with a species that performs better under greywater conditions.

4.7. Further Research

The case study has provided an insight into many areas of on-site constructed wetlands treating domestic greywater within Tasmania. It is evident from the results discussed above that further research and development is required to gain greater knowledge of on-site wetland systems, in order to maximise their wastewater treatment capabilities. These areas of research and development are discussed below.

Further research is required into constructed wetland design. It is vital that wetlands be designed to optimise their treatment capacity by retaining the wastewater within the system for the maximum period of time. A wetland should be designed to allow a plug-flow and eliminate or minimise short-circuiting of the treatment processes.

The effect of pre and post wetland treatment on final water quality needs to be assessed. The introduction of a greasetrap for pre-treatment of household wastewater and a sand filter for polishing of wetland effluent may dramatically improve the water quality of the final effluent. If this is the case then local governments may be more willing to approve constructed wetlands with these forms of pre and post treatment. The maintenance requirements of these forms of treatment should also be evaluated.

More research needs to be done on different wetland plant/reed species. Certain species may provide deeper root penetration, release greater amounts of oxygen into the rhizosphere or cope with greywater only wetlands better than other species.

Research of mature on-site wetlands that span a couple of years are required to assess the seasonal influences on the performance of systems operating in Tasmania.

Re-use of wetland effluent will provide environmental and economic benefits to owners of on-site constructed wetlands. Efficient methods for re-use of wetland effluent need to be investigated in order to maximise these benefits.

Chapter 5. Council Contact

As seen in the chapters 3 and 4, constructed wetlands are a relatively new technology that have a wide scope for use in the treatment of pollutants in water. Constructed wetlands are capable of treating a huge variety of pollutants and can be adapted to any scale, including small-scale on-site wastewater treatment. For constructed wetlands to become an accepted form of on-site wastewater treatment within communities the support of local government (local councils) is vital.

In this chapter, it is aimed to assess the views of local government towards the current forms of on-site wastewater treatment and whether they have knowledge and are receptive to new technologies of treatment. The results of open and informal interviews with five councils are discussed below along with a brief outline of local governments' role regarding on-site wastewater treatment. This chapter does not intend to be an in-depth analysis of local government and on-site wastewater treatment but rather to provide a brief overview of their role and discuss general trends and thoughts that arose from the interview.

5.1. Local Government and On-site Wastewater Treatment

Local government plays three roles with regards to on-site wastewater treatment. Firstly, councils are responsible for the approval of all new on-site wastewater treatment systems, ensuring that they are in accordance with the Tasmanian Plumbing Act. An on-site system must operate to the required levels to minimise health and environmental risks. Therefore, local government must ensure that all systems are of correct design and dimensions for each particular site and household, and that the system is installed and connected properly. The most common types of on-site systems, septic tanks, AWTS and composting toilets, are accredited under the Plumbing Act. Systems that are not accredited, such as constructed wetlands, require a special connection permit where conditions usually apply.

Secondly, local government must investigate any complaints regarding on-site systems. The council must ensure that these systems are operating adequately and are not posing a public health risk or polluting land and/or water bodies. If such problems are occurring the local government must make sure that the owners repair their on-site system to a level where they are operating satisfactorily.

Lastly, local government holds a great deal of knowledge and experience regarding on-site treatment and they are able to provide valuable advice/consultations to members of the municipality. This advice centres on recommending the most appropriate technology and design for each site's characteristics, soil types and household composition. With the knowledge and experience that the councils possess they have the potential to play a greater role as educators within their municipalities. As seen in Chapter 3, many on-site systems fail due to the home owners lack of knowledge regarding operating and maintaining their treatment system. Councils could therefore play a vital role in educating the users of on-site wastewater treatment systems and alleviate some of the failures associated with operation and maintenance procedures.

However, a reality of local councils is that resources are very tight. The lack of resources not only may affect a council's ability to mount an effective education campaign, it also restricts the quantity and quality of monitoring of on-site wastewater treatment systems and limits the amount of current knowledge that they can obtain regarding developments in on-site wastewater treatment technology.

5.2. Interview Objectives and Structure

The objectives of the council interviews were:

- To provide an insight into current on-site wastewater treatment systems used in Tasmania today and to assess the councils' satisfaction with their performance levels;
- To establish the level of knowledge that councils possess of alternative on-site treatment system;
- To determine whether or not the councils are receptive towards new/alternative on-site wastewater treatment systems; and
- To see what the councils perceive as barriers towards the installation and use of these new systems.

The interview consisted of 16 questions and was done over the phone in an informal conversational manner. The five local governments involved with the interview were the Brighton, Clarence, Derwent, Glenorchy and Huon Councils, all in south-east Tasmania. In each case an Environmental Health Officer was interviewed.

5.3. Results and Discussion

The author found that all the councils were experiencing similar problems with the established systems and that they were all enthusiastic for other technologies which, provided they performed to adequate levels, could provide an alternative choice to on-site wastewater treatment. Any other significant comments raised by the environmental health officers will also be discussed.

The 16 question can be split into 3 subject areas; current systems, constructed wetlands, and composting toilets.

5.3.1. Current Systems

As expected, the main form of on-site wastewater treatment was the septic tank followed by the aerated wastewater treatment system. All of the five councils have had, and are still encountering, problems with the performance of septic tanks within their municipalities. Due to the lack of resources and the logistics involved councils are unable to monitor actively the performance of on-site wastewater treatment systems throughout their municipality. However, they do have an obligation to investigate complaints relating to these systems.

Complaints regarding septic tanks problems usually arise from neighbours who are concerned with foul odours and water pooling or running onto their property. It is reasonable to assume that the number of septic tank failures is greater than the councils are aware of because many failures will not be reported, as some rural properties have no close neighbours. The factors that the councils cited as most common in septic tank failure are: soil types that are either clay based and have poor drainage or too sandy and permeable (5 councils); poor system design (3 councils); and the owners' lack of knowledge regarding operation and maintenance (3 councils). It is interesting that two of the three councils which expressed the view that a lack of owners' knowledge was a contributing factor in septic tank failures do not produce brochures outlining the requirements needed in maintaining and operating a septic tank. The three other councils provide a brochure detailing this information when home owners apply for septic tank approvals. Both of the councils that did not produce brochures with this information had plans to do so in the near future. The Brighton Council, which is not currently producing any information, has plans to not only give the information to new approvals of septic tanks but also to new purchasers of houses with septics already installed. The council saw this as important because they are getting many buyers from urban areas moving to rural areas who have had no experience with septic tanks at all. The brochure would inform them of their obligations and responsibilities for owning and maintaining a septic tank.

As stated above, the councils have trouble monitoring all on-site wastewater treatment systems within their municipality and are therefore unaware of many systems that are not performing adequately and may be posing a health and environmental risk. To overcome this problem the Derwent Council has generated an idea that will enable them to keep a closer eye on all new on-site wastewater treatment systems in their municipality. They propose that each year owners of on-site treatment systems must send in a certificate, signed by a qualified person (e.g. plumber, health officer), to the council evaluating the state and performance of their system. All properties that have not sent in a certificate will also be checked and they plan to run random checks on properties to ensure that certificates sent in are are correct. They believe that this approach will enable them to monitor the state of on-site systems within their municipality in a much more efficient manner. The Derwent Council estimates that this method of monitoring will cost the owners between $40-$50 per annum.

The second main form of on-site treatment is the aerated wastewater treatment systems (AWTS). Two councils were happy with the performance levels of AWTS and the other three councils expressed some concern regarding their performance. The main concerns were with regards to turbidity and faecal coliforms counts in the AWTS effluent being above the required level. High faecal coliform levels are of particular concern as the effluent from AWTS is re-used for irrigation around the garden, and the higher the faecal coliform levels the higher the potential health risk. In order to maintain a high quality effluent AWTS's require relatively high levels of maintenance. It is now standard practice for contractors to monitor and maintain AWTS on a quarterly basis to ensure that the systems are operating at required levels. The contractors costs range from $450-$550 per annum. The Clarence Council stated that they had mixed results regarding AWTS's performance, some brands performing to the required levels and some performing poorly. They were not prepared to comment on which AWTS brands performed better than others. All AWTS systems are packaged with information regarding proper maintenance and operating procedures, so while owners are not directly responsible for maintenance they are aware of the systems' needs.

The other types of on-site wastewater treatment systems in operation in these municipalities were composting toilets (refer section 5.3.3 below) and a lagoon system treating wastewater from a tavern. The Brighton Council is currently investigating the use of sand filters to treat greywater and the quality of effluent associated with them.

5.3.2. Constructed Wetlands

None of the councils had constructed wetlands operating within their municipality to treat wastewater on-site but two of the councils have been approached regarding the use of such systems. The Derwent Council has been approached about the use of a constructed wetland system for the treatment greywater for a single household and they are keen to trial and monitor this wetland. The Clarence Council was also approached regarding the use of constructed wetlands in the treatment of wastewater from some proposed holiday units.

As constructed wetlands are a relatively new form of wastewater treatment, the level of knowledge possessed by the Environmental Health Officers regarding constructed wetlands and their potential for on-site wastewater treatment was low. The extent of their knowledge centred on a couple of general articles that they had come across. This may suggest that councils in Tasmania do not have good access to resources/information regarding developments in standard systems and new technologies for on-site wastewater treatment. Therefore, local councils will be unable to provide up-to-date information to members of the municipality seeking advice, resulting in the unnecessary reoccurrence of many old problems, and solutions to problems not being applied as effectively and at a greater cost. Lack of current information and knowledge may mean that new technologies, such as constructed wetlands, stay unknown or applied in circumstances for which they are not designed, which will deter councils from using these new forms of treatment when they may be the best option in certain circumstances.

The main barriers that the councils thought may pose a problem to the acceptance of constructed wetlands in Tasmania were with regard to their performance levels, Tasmania's cooler climate and the lack of information found within the state. As performance levels are a major barrier for constructed wetlands it must be proven that they are capable of working just as well, or better than, the current forms of treatment before any level of acceptance will be granted. Four of the councils expressed a willingness to trial constructed wetlands within their municipality if they were approached. This is a promising and progressive attitude towards assessing the potential of constructed wetlands operating in Tasmania.

5.3.3. Composting Toilets

Four councils have or have had composting toilets operating within their municipalities and all were happy with their performance. As composting toilets have been operating throughout the state and have been approved in many municipalities the councils had a greater knowledge of them than they did of constructed wetlands. In the Glenorchy Council area a composting toilet had been removed by the new occupants when the house was sold. The new owners went back to the old systems as they did not want the 'fuss' associated with composting toilets, they saw that the level of maintenance and the 'storage' of human faeces on-site as disadvantages. These views are barriers to the wide-spread acceptance of composting toilets. Another barrier that was raised by the councils was the cost of composting toilets.

A concern for two councils was the changing ownership of properties with a composting toilet, as seen in the case above. They believe that in many cases the new owners would not be as motivated as those installing the initial system and would not be aware of the maintenance and operating procedures involved with composting toilets. With a lack of this knowledge the composting process is likely to fail and the health risks associated with composting toilets will rise dramatically.

5.4. Summary

Four of the councils showed a keen interest in the potential use of constructed wetlands for on-site wastewater treatment and asked for further information, as their knowledge is minimal. These councils were also willing to trial constructed wetlands within their municipality in order to gain a greater understanding of the mechanics and performance of wetlands. Provided it can be shown that constructed wetlands can operate to required levels there is scope for constructed wetlands to operate in Tasmania because the current systems in use are failing in some circumstances. Septic tanks are failing due inappropriate site conditions, poor design, and/or lack of user knowledge regarding operation and maintenance procedures. AWTS are also failing in certain situations and have the added disadvantages of high running and maintenance costs. As constructed wetlands are robust, have minimal running and maintenance costs, and are independent of a site's soil characteristics they may be able to provide an additional alternative to people requiring on-site wastewater treatment.

Composting toilets have been around a lot longer than constructed wetlands in Tasmania and consequently the environmental health officers have greater knowledge about them and appear to have a reasonable degree of confidence regarding their performance. The councils also have the potential to play a greater role in educating the users of on-site wastewater treatment systems.

Chapter 6. Conclusion

The aim of this study was to determine whether constructed wetlands are a viable alternative to the on-site treatment of domestic greywater in temperate climates. The review of literature in Chapter 2 shows that in many circumstances the current and accepted forms of on-site treatment are not treating wastewater, black or greywater, to sufficient levels in order to eliminate or minimise the associated potential environmental and health risks. The failure of current on-site systems was confirmed when interviewing local governments within Tasmania; all those contacted have had problems with the current systems (refer Chapter 5). It therefore appears that there is an opening for alternative forms of wastewater treatment to enter the market, provided they are capable of treating household wastewater to adequate levels.

As seen in Chapter 3, constructed wetlands are a relatively new technology that is gaining acceptance around the world. The discussion of results in this chapter shows that constructed wetlands are capable of consistently treating wastewater to required levels. While most research has been conducted on medium to large-scale systems there is a growing interest in the use of constructed wetlands on a smaller scale. This chapter also highlighted the success of small-scale systems in treating domestic wastewater in the cold climates of Norway and Austria. Small-scale wetlands can be constructed at a reasonable cost, when compared to current forms of on-site treatment, and allow for the re-use of effluent.

The case study wetland provided a valuable insight into the design, construction, operation, reed growth, cost, performance and flow dynamics of small-scale on-site systems operating in Tasmania. From the case study it was found that pollutant removal processes are occurring within the wetland but not to sufficient levels at this early stage. The design improvements discussed in Chapter 4 would enhance the case study's operational performance but to what level is not known. Probably the most significant factors affecting the wetland's performance were the young age and growth patterns of the reeds and the short-circuiting of the wetland.

As reeds age their root systems penetrate deeper into the substrate, increasing the wastewater/rootzone contact area within the wetland, which should improve the overall treatment performance. While the reeds, *Phragmites australis*, were growing in the case study wetland their growth was not as vigorous as expected. The introduction of septic tank effluent generated a large growth spurt of the reeds. A wetland system relies on sufficient nutrients and trace elements within the water column to generate vigorous and healthy reed growth. It would appear that greywater alone does not possess sufficient quantities of nutrients and trace elements to sustain vigorous growth for *Phragmites australis*. However, other wetland species may grow well in systems that receive greywater only; more research is needed in this area. The root systems of the reeds play a vital role in the treatment processes by providing aerobic microorganisms with attachment sites and life sustaining oxygen within the substrate.

Within the case study it was discovered that greywater was passing through the wetland in a much quicker period than expected. This reduction in hydraulic retention time was due to two reasons. The first was the production of greywater by the household's occupants that was in excess of estimates used in design calculations. The second reason was the internal hydraulic flow. The hydraulic tracer study showed that greywater flowed more directly from the inlet to the outlet within each module. Short-circuiting of the wetland results in greywater not remaining within the wetland for a sufficient time period, thereby reducing the treatment processes at work. To eliminate or minimise short-circuiting it is imperative to design and construct a system that removes dead pockets and encourages plug-flow through the wetland.

While knowledge of constructed wetlands within local governments around Tasmania is very limited the interviewed environmental health officers showed a great deal of interest. The environmental health officers suggested that they might be willing to trial wetlands within their municipalities. Such trials would provide further knowledge and data into the use of on-site constructed wetlands within the State, ultimately helping small-scale systems progress to a stage where they consistently achieve desired water quality levels. Trials within the State would also raise awareness of constructed wetlands and their potential to provide an alternative form of on-site wastewater treatment.

In conclusion, constructed wetlands have the potential to become a viable alternative to the on-site treatment of wastewater in temperate climates. Constructed wetlands are capable of consistently providing a reliable form of wastewater treatment at a reasonable cost and allow for the re-use of the effluent. However, more research needs to be conducted, within and outside Tasmania, to develop small-scale wetlands to a stage where they consistently treat wastewater to required levels and to raise awareness and confidence in such systems.

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